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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

USDA ARS Walnut Gulch Experimental Watershed

1953–2003

Abstract

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This report comprises abstra
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This report represents state-of-the-art research in watersheds. The content includes reviews of watershed research programs conducted by the U.S. Department of Agriculture's Agricultural Research Service and Forest Service, U.S. Geological Survey, U.S. Bureau of Land Management, U.S. Environmental Protection Agency, and National Science Foundation Consortium of Universities for the Advancement of Hydrological Sciences, Inc., as well as recent research on watershed-scale topics such as hydrology, erosion, economics, instrumentation, ecology, sociology, and fire.

Keywords: watershed, watershed management, hydrology, water quality, water quantity, runoff, sediment loss, sediment transport, sociology, ecology, ecosystem, erosion, economics, instrumentation

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Preface

Watersheds are the primary planning unit being used for resource management, as well as the natural unit for research studies on surface water hydrology and water quality. The First Interagency Conference on Research in the Watersheds (ICRW) was conceived by a group of federal agencies (ARS, USGS, USFS, BLM, EPA, NRCS and NSF) to bring together researchers working at the watershed scale and stakeholders working on watersheds to bridge the gap between the two groups. The intent was to have researchers present what they are doing now, what more they could do, and how the research is being applied for management purposes. In turn, the watershed stakeholders would articulate how current research is helping them and the additional steps that are needed to help them make better decisions about managing their watersheds. The meeting organizers specifically designed this conference to be a one-stop shopping point for stakeholders to get a comprehensive overview of watershed research by all of the major federal agencies conducting watershed research.

To achieve these goals, the format of ICRW was unique. First, the conference was located at a small facility within the San Pedro River Watershed in the town of Benson, Arizona. All meals were catered on site to encourage informal interaction between researchers and stakeholders. Second, the first plenary session consisted of invited speakers from federal agencies to review their watershed research programs and from the Upper San Pedro Partnership to present an example of research scientists and decision makers working together. This set the stage for the nearly 150 volunteered presentations of research on watershed-scale topics such as hydrology, erosion, water quality, economics, instrumentation, ecology, sociology, remote sensing and fire. Third, the ICRW was scheduled to coincide with the 50th anniversary of the renowned USDA ARS Walnut Gulch Experimental Watershed (WGEW) near Tombstone, Arizona. To celebrate 50 years of research at this experimental watershed, a half-day field tour and evening BBQ at the watershed were scheduled.

This book contains the proceedings of the First Interagency Conference on Research in the Watersheds in Benson, Arizona, October 27–30, 2003. The proceedings begins with six invited papers describing the current and future watershed research programs of ARS, USGS, USFS, BLM, EPA, and CUAHSI. Following these, the proceedings provides an overview of watershed research organized by the topics of erosion, hydrology, watershed modeling, watershed networks and data management, water quality and quantity, ecology, integrated management and integrating science with watershed decision making.

The organizing committee would like to express its gratitude to BLM, ARS, and EPA for their financial support, and to ARS for publishing this volume. We also express appreciation to all the presenters who contributed to these proceedings and to all participants who made the conference a success. Plans are underway to hold the Second Interagency Conference on Research in the Watersheds in 2008.

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Overview of Agency Watershed Research Programs

The USDA-Agricultural Research Service Watershed Research Program

Mark A. Weltz, Dale A. Bucks

Abstract

Water quantity and quality issues have increasingly become the focus of attention of United States citizens, private and public organizations, and units of government striving to meet competing demands while protecting the environment and public health. Sound agricultural management practices are required to ensure success in maintaining a healthy and productive land and water base that sustains local communities, food and fiber production, and also protects and restores critical natural systems. The central mission of the USDA-Agricultural Research Service's (ARS) Watershed Research Program is to address challenges and solve problems that confront American agriculture enterprises. The ARS accomplishes this mission by using the scientific method to improve our understanding of basic hydrologic processes. ARS and its collaborators use this knowledge to develop new methodologies and technologies to mitigate deleterious effects of floods and droughts, reduce soil erosion and sedimentation on our farms and within our streams and lakes, improve water quality, and enhance water supply and availability. The ARS watershed network is a set of geographically distributed experimental watersheds that has been operational for more than 70 years and is the most comprehensive watershed networks of its kind in the world. The watershed facilities serve as outdoor laboratories that provide an essential research capacity for conducting basic long-term, high-risk field research. The watershed network and its associated historical database from 23 states provide the only means to evaluate the long-term impacts and benefits of implementing agricultural practices on water quality and water availability, documenting effects of global change, and developing new instrumentation and decision support systems to enhance the economic and environmental sustainability of agriculture. More than 140 ARS subwatersheds and related facilities, ranging in size from 0.2 hectares to over 600 km^2, are currently operated from 17 research facilities within the continental United States.

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Introduction and History

The ARS Watershed Network (Figure 1) can be broadly characterized as an *intensive* network where some sets of geographically distributed watersheds are observed and studied in great detail. In an intensive network, numerous observations and dense instrumentation nets are concentrated in relatively small watersheds to support investigations for specific hydrologic process understanding. This is in contrast to an *extensive* network which collects information over a much larger area, at lower instrumentation resolution, for broad interpretation by providing regional "index" information (Neff, 1965).

The ARS Experimental Watershed Program grew out of depression era efforts by the Civil Conservation Corps (CCC) and the Soil Conservation Service (SCS). Kelly and Glymph (1965) described the early history of the watershed program, including research associated with the 1930s conservation motto "stop the water where it falls." The research focused on merits of upstream watershed conservation to reduce runoff and erosion. It was geared to studying on-site problems and concentrated on field-sized watersheds up to roughly 10 hectares and, to a large extent, utilized paired watershed analyses. In the mid-1930s, major research stations were established in Coshocton, OH; Hastings, NE; Riesel, TX, and Watkinsville, GA to examine fields and watersheds up to several hundred hectares in size. Research addressed on-site effects of tillage and management practices, and plot and lysimeter studies were incorporated at some sites. Many of these experimental watersheds were transferred to the newly formed USDA-ARS in 1954.

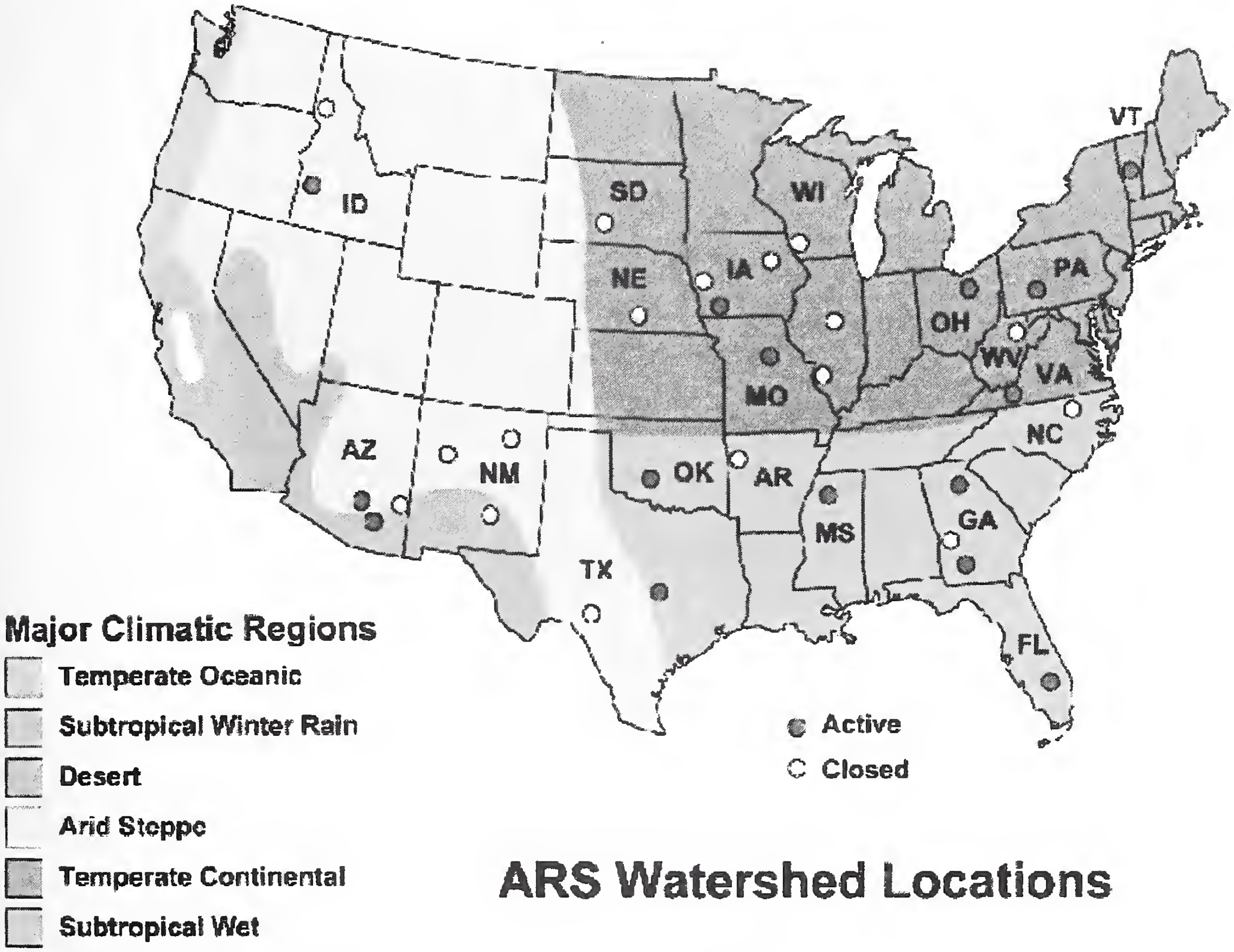

Figure1. Locations of the historical and active Agricultural Research Service experimental watersheds.

There was early recognition of the scaling problems in transferring knowledge from small to larger watersheds (Harrold and Stephens 1965). This problem and growing concern of downstream, off-site impacts of upstream watershed practices resulted in establishment of a subset of larger ARS experimental watersheds associated with new watershed research centers in a number of hydroclimatic regions in compliance with U.S. Senate Document 59 (Great Plains, Northeast, Northwest, Southeast and Southwest Watershed Research Centers in Chickasha, OK; State College, PA; Boise, ID; Tifton, GA; and Tucson, AZ; respectively). The goal in establishing the watershed research centers was to select a representative basin and establish satellite basins, which were less well instrumented, to extend the data and findings from the primary watershed center. Nested watersheds and unit source areas on major soil types were included in the watershed designs to investigate scale effects.

The Current Network

Seventeen locations within the contiguous United States are currently collecting a variety of abiotic and biotic data at 140 subwatershed nested within the larger ARS watersheds. Watersheds are currently being added to the network to address water quality and turf management issues in Iowa, Indiana, Ohio, and New York. Data from these watersheds will be available in the near future. The ARS watersheds represent numerous diverse land uses and agricultural practices and cover a wide range of hydroclimatic conditions. The diversity of observations made at these watersheds is a reflection of the diversity in dominant hydroclimatic processes across locations and evolving research objectives. As research objectives have changed to address problems such as water quality (e.g., biotic, chemical, pathogen, sediment) and global change, instrumentation and observations have been added to the basic rainfall-runoff observation infrastructure. An important component of the network is the ARS

Hydraulics Engineering Unit located in Stillwater, Oklahoma which has provided critical expertise and facilities in the development of flood-control and hydraulic structures and runoff measurement devices deployed in many of the watersheds. ARS also conducts hydraulic engineering research on the design and safety issues related to earthen dam flood control structures in support of Public laws' PL-534 and PL-566 at Stillwater, OK. Greater detail on individual ARS watersheds can be found at: http://www.nwrc.ars.usda.gov/watershed/.

Data Availability

The Agricultural Research Service (ARS) is a research organization. Data collected from the ARS Watershed Network should be considered experimental data. While much of the original instrumentation, installation and data processing procedures for basic rainfall, runoff and meteorological data was guided by Handbook 224 (Brakensiek et al. 1979 - revised from 1962), data collection has evolved at individual locations to address regional research needs. ARS watershed data have not historically been collected and reviewed under a national standard set of guidelines and procedures such as those employed by the USGS. Instruments, parameters observed, and data reduction procedures vary from watershed to watershed. A description of data acquisition programs and an assessment of the quality of collected data at many of the experimental watersheds is described in USDA (1982) and at: http://www.nwrc.ars.usda.gov/watershed/.

ARS does not have a mandate to monitor and distribute data collected at its experimental watersheds, availability of data from the watersheds also varies by individual ARS watershed location. Based on data compiled and maintained by Jane Thurman at the Hydrology and Remote Sensing Laboratory in Beltsville, Maryland, as of January 1, 1991, ARS had operated over 600 watersheds in its history. A rainfall-runoff database is available from in the Hydrology and Remote Sensing Laboratory for 333 of these watersheds. In addition, a historical climate database for the United States and the Cligen weather generator develop by Dr. Arlin Nicks is available at: http://hydrolab.arsusda.gov/wdc/arswater.html. About 16,600 station years of data are stored there from watersheds ranging from 0.2 hectares to 12,400 km^2. After 1990, the HRSL no longer archived data but has provided links back to the individual ARS watershed locations. These locations are making a concerted effort to make the ARS Experimental Watershed data more readily accessible and to provide additional types of data (soils, vegetation maps, geology in standard geographic information system formats, etc.) available through a web enabled search and retrieval system but progress varies due to resource constraints. It is anticipated that a prototype system that is currently being developed will be available in late 2004. Furthermore, we exploring methods to link the ARS databases with databases maintained by the U.S. Forest Service and its watershed network (Figure 2) and the Natural Resources Conservation Service Soil Climate Analysis Network (SCAN) (Figure 3) as a means to efficiently access both historical and real-time data on hydrologically important data. Those interested in working with ARS scientists and with ARS Watershed data should contact the Research Leader at that watershed location or can contact Dr. Mark Weltz, National Program Leader for Hydrology and Remote Sensing (maw@ars.usda.gov) for information about the Watershed Program as a whole.

Collaboration / Cooperation

The ARS Experimental Watersheds have been magnets for interagency, and university collaborative research. For instance, these outdoor laboratories have been invaluable for validating remote sensing satellites, aircraft-based instrumentation, and development of retrieval and prediction algorithms that are used by NASA for estimating a variety of biotic and abiotic parameters and conditions.

Several noted examples include interagency interdisciplinary hydrologic, atmospheric and remote sensing experimental campaigns:

I. Mahantogo'90
(Mahantango Watershed, PA)
II. Monsoon'90; Walnut Gulch'92
(Walnut Gulch Watershed, AZ)
III. Washita'92
(Little Washita Watershed, OK)
IV. SGP (South Great Plains, '97, '98, '99)
(Little Washita Watershed, OK)
V. DEC (Demonstration Erosion Control)
(Goodwin Creek Watershed, MS)
VI. SALSA'97-'99
(Walnut Gulch/San Pedro Watersheds, AZ)
VII. SMEX 2001-2002 (Walnut Creek Watershed, IA)
VIII. SMEX 2003
(Little River Watershed, GA)

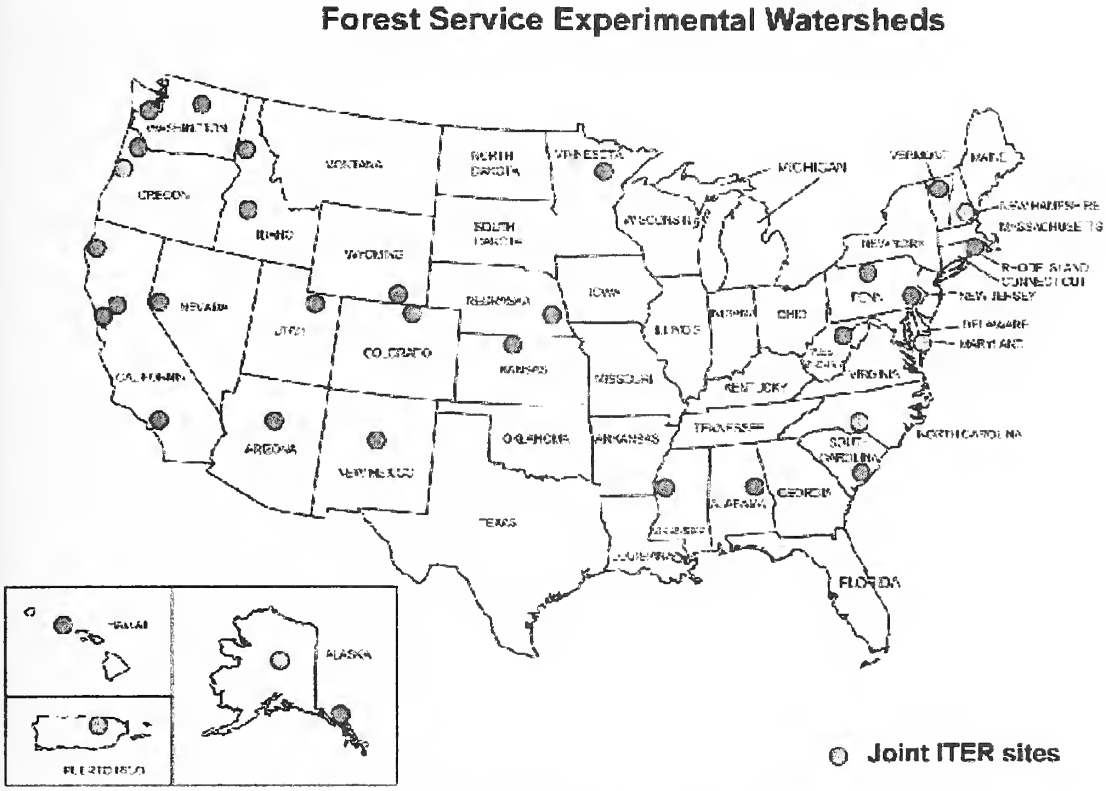

Figure 2. Locations of the US Forest Service experimental watersheds.

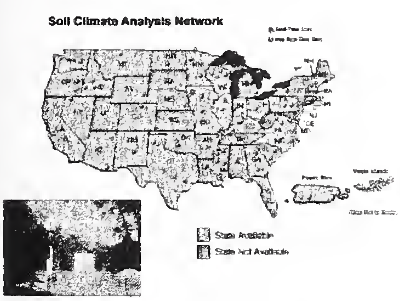

Figure 3. Locations of the Natural Resources Conservation Service Soil Climate Analysis Network many of which are co-located on the Agricultural Research Service experimental watersheds.

The ARS Experimental Watersheds are managed as outdoor laboratories. We stress collaboration with federal, state, and non government agencies in our management of theses facilities to broaden the types of observations being made and to leverage the existing watershed infrastructure to address the challenging questions facing the Nation today. Examples of ongoing collaboration:

- USDA-NRCS Soil Climate Analysis Network (SCAN) http://www.wcc.nrcs.usda.gov/scan/
- UNESCO/WMO Hydrology for the Environment, Life and Policy (HELP) catchments http://www.unesco.org/water/ihp/help
- Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUASHI)
- DOE AmeriFlux
- NOAA Surface Radiation Network (SURFRAD) in the Continental United States
- DOE Atmospheric Radiation Measurement (ARM/CART) site in Oklahoma

A new Joint U.S.-China Center for Soil and Water Conservation and Environmental Protection, was dedicated on May 2, 2002. This center is located on the campus of the Northwest Sci-Tech University of Agriculture and Forestry (NWSUAF) in Yangling, China. The U.S. counterpart is located at the University of Arizona and the Southwest Watershed Research Center in Tucson, Arizona. ARS scientists are working collaboratively with their Chinese and University counterparts to develop new methods to reduce soil erosion and sedimentation that will benefit both countries in managing soil and water resources at both the field and watershed scale.

Major Accomplishments

Development of innovative instrumentation

ARS watersheds have pioneered the testing and development of stream flow instrumentation including the drop-box weir for high-energy, high-bedload systems, supercritical flumes for arid regions, and small-scale runoff flumes. Stream sampling methods for water quality such as the Coshocton Wheel, traversing slot sediment samplers, and widely used in-stream samplers have also come from ARS watersheds. Other advances include state-of-the-art hydro-meteorological field sensors, watershed-wide telemetry, archival equipment and systems, the dual-gage precipitation measurement system, load cell precipitation gage, radar and acoustics technology to measure sediment transport, snow pillow and advanced snow sensors and programmable, variable rate, rainfall simulators.

Development and testing of remote sensing technologies and applications

Pioneering research in both the theory and application of remote sensing to the use of microwave remote sensing of soil moisture has been conducted by ARS personnel at the ARS watersheds. Results are currently being implemented by both NASA and the Japanese space agency. Large scale soil moisture observations may contribute to major breakthroughs for hydraulic modeling, crop yield forecasting, drought assessment, irrigation management and the ability to detect and model land surface response in climate change studies. In addition, long term acquisition of complimentary remote sensing imagery supported by ground and atmospheric measurements at several ARS watersheds are used as long-term validation for both NASA and European Space agency sensors.

Improvement in agricultural water quality

Nutrients and herbicides related to farming practices have been detected in shallow groundwater and agricultural runoff in many parts of the country. ARS watershed research has led to: (i) buffer system designs composed of grasses and trees that can be used to assimilate nitrogen and phosphorus from both surface water and shallow groundwater and reduce off-site impacts of animal feeding operations, (ii) nitrogen management practices, using the ARS-developed Late Spring Nitrate Test, which have demonstrated reduced nitrate pollution levels, (iii) the development of the Soil and Water Assessment Tool (SWAT) model, which has been applied extensively for policy planning and in developing best management practice alternatives, and (iv) the quantification of water quality impacts of brush control herbicides picloram and clopyralid, which were shown to dissipate quickly in the soil and to be undetectable in surface runoff or subsurface flow.

Studies in ARS watersheds were instrumental in obtaining approval of these herbicides for public use.

Rainfall frequency analyses

Analyses of ARS dense rain gauge networks were utilized to modify NOAA National Atlases of rainfall frequency which is utilized to develop design storm characteristics for flood control maps and prevention activities.

Development of hydrologic and natural resource management models

ARS watershed research and data have been critical to the development and validation of natural resource models too numerous to mention in this report in detail (ANAGNPS, CONCEPTS, CREAMS, Curve Number, GLEAMS, EPIC, KINEROS, REMM, RUSLE2, SRM, SWAT, and WEPP). An example of an ARS model that has had tremendous impact is the Universal Soil Loss Equation (USLE) model. The USLE and its replacements the Revised Universal Soil Loss Equation (RUSLE) and RUSLE2 erosion prediction tools are the most widely utilized field scale erosion prediction tools in use around the world today. The USLE model was recently recognized for its outstanding impact on sustaining agriculture production around the world by reducing soil loss by the American Society of Agricultural Engineering. The ARS-developed KINEROS model was utilized by a consulting firm and resulted in construction saving of over $16 million on a series of dams on the Au Sable River in Michigan. More recently, the Simulator for Water Resources in Rural Basins (SWRRB) model and the Soil Water Assessment Tool (SWAT) model have been used by many federal and state agencies to evaluate USDA conservation program effectiveness and the economic and environmental impacts/benefits derived from implementing conservation practices.

Hydraulic structure design

The Natural Resources Conservation Service (NRCS) has used ARS developed procedures for design and construction of more than 800,000 km (500,000 mi) of vegetated channels. The design procedure is listed as one of the top five outstanding agricultural engineering achievements of the 20th century by the American Society of Agricultural Engineering. These and other design criteria are available on the SITES 2000: Water Resources Site Analysis CD from ARS. This expert system is helping NRCS and local sponsors of earthen dam flood control structures design urgently need safety upgrades to the 11,000 structures that have been constructed across the United States. ARS in association with the Oklahoma Conservation Commission has also developed a video that describes the benefits of these small hydraulic structures that explains the importance of maintenance and repair of the structures.

Future Program Direction

The ARS Watershed Program and its Experimental Watersheds provide exceptional **"outdoor laboratories"** to develop knowledge that addresses societal water resource issues in real world settings. The stability of these research platforms, with a high-quality knowledge base and observational infrastructure makes them ideal facilities for collaborative research to investigate the hydrologic cycle and potential changes to it across a wide range of hydro-climatic conditions. There is no comparable network of experimental agricultural watersheds in the world. For a marginal increase in resources, many of these ARS Watersheds can integrate additional observations of critical state, flux and biogeochemical variables to become independent verification and validation watersheds for addressing questions on availability and reliability of clean water and address issues related to Total Maximum Daily Loads (TMDL) and global climate changes that are confronting our Nation. In addition, a number of ARS Experimental Watersheds could potentially partner with the National Environmental Observation Network (NEON) that the National Science Foundation is attempting to implement across the nation (http://www.nsf.gov/bio/neon/start.htm).

NEON is envisioned as a network of networks, a system of environmental research facilities and state of the art instrumentation for studying the environment that will enable integrative research on the nature and pace of biological change at local, regional and continental scales. The ARS watershed network could collaborate with the NSF sponsored NEON partners to develop advanced technologies and measurement capabilities to document the impact and benefits that agricultural production systems have on all factors that affect the structure and function of natural and managed ecosystems and the ability of these systems too sustainable deliver clean and safe surface and ground water supplies to the public.

A second critical role of experimental watershed data in the quest for hydrologic scientific understanding was clearly stated in the 2001 NRC report *Envisioning the Agenda for Water Resources Research in the Twenty-First Century.* The report states the following: "Intensifying water scarcity cannot be successfully addressed in the absence of reliable data about the quantity and quality of water over time and at different locations. The end-of-century trend of investing fewer and fewer dollars in data-gathering efforts...will need to be reversed if availability is to be adequately characterized." In *A Plan for a New Science Initiative on the Global Water Cycle*, Hornberger et al. (2001) emphasized that "beyond the need to collect new data, existing long-term records must be archived and preserved carefully, and observations must be continued indefinitely at sites with long high-quality records, so that patterns of temporal variability, including long-term, low-frequency fluctuations, can be identified and studied."

A third opportunity for expansion and targeting of the ARS experimental watersheds stems from the reauthorization of the 2002 Farm Bill. This act substantially increased funding for the Environmental Quality Incentives Program (EQIP), the Conservation Reserve Program (CRP) and provided continued funding for other conservation programs. Overall, Federal expenditures for conservation practices on farms and ranches in the U.S. were increased about 80 percent above the level set under the 1996 Farm Bill. While it is widely recognized that these conservation programs will save millions of acres from soil erosion, enhance water and air quality, promote wetland and wildlife habitat restoration and preservation, and conserve agricultural water use, the environmental benefits have not been previously quantified and reported at the national level. Tracking the progress of these programs in terms of the environmental benefits will allow policymakers and program managers to implement and modify existing programs and design new programs to more effectively and efficiently meet the goals of Congress.

This new multi-agency program is currently called the Conservation Effects Assessment Project (CEAP).

The goal of CEAP is to provide the farming community, the general public, OMB, legislators, and others involved with environmental policy issues an accounting of the environmental benefits obtained from conservation program expenditures. CEAP will also provide an opportunity for increased cooperation and collaboration initially among USDA agencies such as NRCS, ARS, FSA, CSREES, NASS, and ORACBA. As increased funding is provided for the program, cooperation and collaboration will continue to expand with other USDA agencies, including FS, ERS, USGS, and EPA as well as other federal and state agencies and private organizations in the development of scientifically-based practices that provide an optimum to balance environmental benefits, program costs, and food and fiber production.

Note

This material was originally presented by Dave Goodrich, Daniel Marks, Mark Seyfried, and Clarence Richardson as a poster at the December 2000 America Geophysical Union in San Francisco, CA and has been updated for this meeting. We would also like to thank Jane Thurman and all the other ARS employees in the watershed program for the work they have put in developing and maintaining the historical watershed data.

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Watershed Research and Development in the USDA Forest Service

Douglas F. Ryan

Abstract

The USDA Forest Service has a long history of watershed research and development intended to address issues facing land managers. Topics of study include development and testing of forest Best Management Practices to protect water quality, instream flow studies to evaluate effects of flow alterations on stream and riparian ecosystems, and effectiveness of measures to reduce risks to water resources after wildland fires. Forest Service research watersheds serve as sites for integrated ecosystem studies, including six that are formally part of the National Science Foundation's Long-Term Ecological Research (LTER) Network. Integration of watershed studies enables results to be applied to a variety of issues including protection and restoration of habitat for endangered fish and aquatic species, and impacts of air pollution in water quality. In the future, results of watershed studies will be synthesized more widely to address management and policy issues at larger scales of time and space.

Keywords: watershed, research, forest, management, ecosystem, network

Introduction

Among the mandates that Congress gave the USDA Forest Service in its "Organic Act of 1897" was insuring "favorable conditions of water flows" from the federal lands that the agency administers. To comply with this mandate, the Forest Service began studies of the effects of land management on water quantity and quality early in the 20th century. Although methods and techniques have evolved greatly in the intervening years, the agency's interest and research in this topic continues to the present.

The objective of the Forest Service watershed research and development program is to provide a scientific basis for decisions by land managers and policy makers in the Forest Service and other federal, state, and tribal agencies, and technical assistance to managers of private lands regarding the effects of land management on water resources.

Funding for Forest Service research and development on watershed processes was $16 million in fiscal year 2002. This program is substantially an in-house effort that is carried out in collaboration with other federal agency and university scientists. A related agency program of research on fish and aquatic habitats that was funded at $7.5 million in fiscal year 2002, but that program is not the primary focus of this paper.

Program Description

Consistent with the agency's water resources mandate, a long-running focus of research in the Forest Service has been the development and testing of Best Management Practices to protect water quality during land management including practices such as timber harvesting and road building and maintenance. These studies continue to be relevant as new management practices are devised in response to new technology, increased demands to protect the environment and changing resource conditions. For example, while construction of forest roads was a major focus of research in the 1980s, over the past decade a need has emerged for environmentally acceptable methods to decommission roads as national

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forests have come under pressure to reduce the size of their road networks. The research emphasis on roads has shifted accordingly to include studies of practices to remove roads from service and reduce the impacts of existing roads. Studies of land use effects are often classic, paired-watershed experiments in which land use on one study watershed is deliberately manipulated while a similar watershed is maintained as an unaltered reference.

Many parts of the country are experiencing an increase in competing demands for scarce water resources. Some water users, such as urban, agricultural, and industrial activities, withdraw water from streams, while others, such as instream recreation, hydroelectric power generation, and protection of aquatic and riparian habitats for fish and wildlife, require flows within stream channels. Managers and policy makers in the Forest Service and other agencies often interact with states, tribes and other agencies in decisions on how to allocate water that flows across federal land among such competing uses. An important question for land managers involved in these decisions is how much water needs to remain within the stream for "in-stream" uses? The Forest Service has research devoted to developing a scientific basis for estimating maintenance flows to protect aquatic and riparian ecosystems in streams that have their flow regime altered by water withdrawals or dam operations. These studies examine how changing stream flow affects geomorphology, sediment transport, and aquatic and riparian habitats. Studies also include assessing the economic impacts of altered stream flows on instream recreation. The Forest Service has established a technology transfer team dedicated to making the results of these studies readily available to legal teams negotiating water rights adjudications, and agency participants in Federal Energy Regulatory Commission proceedings on hydropower relicensing involving streams and rivers on national forests.

Watershed research examines the effects on water quality and aquatic habitat of processes occurring on hill slopes, in streamside areas, and in channels. This work is coordinated with research on the habitat requirements of threatened and endangered fish, aquatic organisms and wildlife in streams and streamside zones. Studies include the relationships between hydrology and geomorphology and important ecological components such as streamside vegetation, landslides and debris flows, and large woody debris that are often critical to fish and wildlife habitat. Results of these studies have been incorporated into management and policy actions that are being taken to protect and restore fish and aquatic species at risk of extinction such as the native salmon species in the Pacific Northwest.

Forest Service scientists conduct research on the effects on water resources of wildland fire and the management actions to reduce the consequences of wildland fires. Severe wildland fires can consume surface litter, and some soils can become water-repellent after their surface has been burned. As a result of these and other changes after fires, severe erosion, floods and damaging mud flows may occur immediately downstream after wildland fires. Such extreme events can pose a risk to life and property for nearby communities even after the fires are out. A science synthesis found few studies of treatments to reduce the risk to downstream assets (Robichaud et al. 2000). As a result, agency scientists have initiated investigations of the effectiveness of post-fire treatments. Preliminary results show that existing treatment techniques, such as straw wattles and log erosion barriers, can reduce the risk of erosion for low intensity rain events, but not for high intensity events. Research is continuing to develop, test and monitor new techniques to protect the public from post-fire risks. Models has been developed that incorporate research results to permit managers to evaluate post fire erosion hazard and decide which actions best reduce the risk.

A long term interest of Forest Service research has been effects of land management on water quality. An enduring focus has been the linkage between land disturbance and erosion, sediment production and deposition. Steam chemistry has been studied in relation to processes such as biogeochemistry and effects of atmospheric deposition. One of the earliest studies on effects of acid deposition, for example, was conducted at Hubbard Brook Experimental Forest that contains Forest Service study watersheds. Recent Forest Service work in Southern California has found impacts on soils and water chemistry from atmospheric deposition of pollution-derived nitrogen compounds. Responding to the Safe Drinking Water Act's requirements for states to conduct Source Water Assessments, the Forest Service published a synthesis of effects of forest management on drinking water quality (Dissmeyer 2000). This study pointed out that little information was available on the relationship between dispersed recreation and the presence of human pathogens in surface waters. To fill this gap, Forest Service scientists formed a collaboration with the USDA Agricultural Research Service to investigate better means of detecting and studying pathogens in streams.

Forest Service experimental watersheds often serve as the sites for long-term, integrated, ecosystem studies. The Forest Service manages six Long Term Ecological Research (LTER) sites in collaboration with National Science Foundation, and numerous agency and university cooperators. These sites are Hubbard Brook Experimental Forest in New Hampshire, Coweeta Hydrologic Laboratory in North Carolina, H. J. Andrews Experimental Forest in Oregon, Luquillo Experimental Forest in Puerto Rico, Bonanza Creek Experimental Forest in Alaska, and the Baltimore Ecosystem Study in Maryland. Data from these intensively-studied sites are often used to develop and validate models of ecological processes because measurements of many interrelated environmental variables have been done over long time periods at these sites. Most of these sites were host to ecosystem-level investigations even before the NSF formally established the LTER Network in 1980. An exception is the Baltimore Ecosystem Study that was started in 1997 to investigate how a city functions as an ecosystem. Although the Forest Service does not manage the land in this urban area in the same sense that it does at most of its forested research sites, watersheds still comprise the basic unit of this urban study in ways the are similar in many respects to investigations that were traditionally conducted in forests.

Future Directions

Although an historic strength of Forest Service Research has been its focus on intensively studied watersheds, a goal for the future is to develop better ways of linking studies into networks that permit research results to be synthesized to apply at larger spatial and temporal scales. A joint effort is underway between the Forest Service's H. J. Andrews Experimental Forest and the NSF's LTER Network to develop a web-based network to provide public access to long-term hydrology data from Forest Service research watersheds and LTER sites. A search engine has been developed that permits users to find and download long-term data and metadata on steam discharge and meteorology. Twenty-three Forest Service-related experimental watersheds and additional LTER sites are in the process of linking their long-term stream flow and meteorological data and metadata to this search engine. This new resource, called "HydroDB," can be visited on the web at: http://www.fsl.orst.edu/climhy/hydrodb/index.htm. Graphical and statistical tools to display and analyze these data are being developed. Future plans include adding the capability for practical uses of these data for applications such as improving road culvert design for fish passage. Addition of other types of data such as sediment transport and stream and precipitation chemistry is also anticipated. This approach has the potential serve as a model that could be applied to many types of interrelated environmental observations. Developing this tool has been an interagency enterprise from its inception, and we welcome participation in this effort by other agencies and institutions with long-term studies. This new tool is a first step toward drawing upon the agency's investment in intensive, long-term studies to make their results more widely available for synthesis and application to larger-scale problems in the future.

Summary

Forest Service watershed research and development has long been part of the agency's response to its original Congressional mandate to protect water resources as an integral part of managing national forests. The agency has a long-running history of watershed studies designed to investigate the effects of land use practices on water quantity and quality. Over many decades these studies have evolved as public values with respect to land use have changed. An early emphasis on developing and testing Best Management Practices for timber operations continues but has refocused on new and emerging management practices. Instream flow studies seek to provide tools to help managers to evaluate flows needed to maintain viable stream and riparian ecosystems where water withdrawals and flow modifications occur on federal land. Studies of effects of wildland fire focus on estimating the effectiveness of practices to reduce post-fire flood risk and sediment flows and other post-fire remediation. Watershed studies have been incorporated into investigations of factors that are critical for protecting and restoring habitat for fish, aquatic and riparian species in danger of extinction. Forest Service watersheds have become sites of integrated ecosystem studies with six formally participating in the LTER Network. In the future Forest Service watershed research will focus on developing better ways to synthesize the agency's long-term watershed studies to address emerging problems for decisions makers at larger temporal and spatial scales.

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Watershed Research in the Bureau of Land Management

Daniel P. Muller

Abstract

The Bureau of Land Management (BLM) has been involved with research on watersheds for decades. Research related to processes that affect precipitation infiltration and runoff, soil transport, vegetation health, and overall sustainability of watersheds is accomplished cooperatively with other Federal agencies and State institutions. The purpose of this paper is to review BLM's approaches to conducting watershed research on public lands, with an emphasis on future research direction.

Keywords: science strategy, science catalogues, watershed unit, watershed scales

Introduction

BLM's management role

BLM manages over 261 million acres of public lands as mandated by the Federal Land Policy and Management Act (FLPMA) of 1976. BLM's mission is to sustain the health, diversity, and productivity of the public lands for the use and enjoyment of present and future generations. These lands are generally open to a number of uses, such as recreational opportunities, commercial activities, scientific and educational activities, transportation systems, and conservation initiatives (e.g., wildlife habitat management). Portions of these lands are designated as special management areas, which include wilderness areas, wild and scenic rivers, national monuments, and national conservation areas. Issues arise due to competing and more concentrated uses of public lands and resources. Land and resource management issues related to the use of public lands are increasing due to population growth and expanding recreational expectations. Credible science-based information is essential in determining which combinations of uses by location will best meet the present and future needs of the American people.

BLM's scientific role

FLPMA often refers to science and implies the need for scientific information to manage the public lands. Consistent with this law, BLM developed a national Science Strategy in 2000 with three primary objectives: to delineate the role of science in BLM decision making; to establish a clear process for identifying science needs and priorities; and to provide a mechanism for communicating those needs (Bureau of Land Management 2000). This process ensures that science needs are reflected in BLM's planning and budget documents.

In accordance with the Science Strategy, management issues are documented in catalogues that highlight science needs at both the national and regional levels. Once issues are identified, BLM scopes the subjects with science providers and begins to develop science-related funding proposals through the BLM budget planning system. Funding priorities are based on approved projects and the availability of appropriated funds. The results of research and other scientific investigations are used in planning and decision making, including agency strategic planning, policy formation, land use planning, and specific activity-level planning.

BLM works with its science partners to assist in addressing land and resource management issues. A significant component of this process is communicating national and regional science needs to science partners, such as the Cooperative Ecosystem

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Studies Units (CESU 2002). Communication and feedback are key elements in BLM's efforts to identify and prioritize its science needs. Additionally, effective communications with partners is needed to ensure that results are applied to land management activities and decisions.

Watershed Research

BLM acknowledges a long history of cooperation with the Agricultural Research Service (ARS). BLM's rangelands equal about 165 million acres (U.S. Department of the Interior 2003) or approximately 63 percent of the total public lands managed by the agency. Riparian and wetland areas, which intersect both forests and rangelands, account for another 23 million acres. These figures provide an appreciation for the steady demand by BLM for rangeland research, and why the agency has depended on and utilized ARS published results in its land management processes. Research sites located in watersheds, such as Walnut Gulch (Arizona) and Reynolds Creek (Idaho), have provided data, models, and interpretations related to precipitation effectiveness for plant growth, soil moisture availability to plants, effects of vegetation management on runoff in streams, and various natural and land use stresses on vegetation production.

The watershed unit is used by BLM to assess resource conditions and evaluate compliance with rangeland health standards (U.S. Department of the Interior 2001) and forest health protocols. Resource conditions are typically measured using indicators of health, such as those reported by Prichard (1993), Pellant et al. (2000), The Heinz Center (2002), and the 2003 Sustainable Rangelands Roundtable First Approximation Report (Sustainable Rangelands Roundtable 2003). Various scales of watershed units (e.g., sub-watershed, watershed, basin) must be considered, as not all indicators of health are appropriately assessed at any one level. In other words, it may not be possible or feasible to measure conditions within sub-watersheds and simply aggregate those values for determining conditions at more coarse scales. In view of research needs, the land management agencies will continue to depend on studies reflecting diverse watershed scales.

A review of current and recently published research being cooperatively sponsored between BLM and ARS was completed for this paper (U.S. Department of Agriculture 2003a, U.S. Department of Agriculture 2003b). A list of selected themes of interest to BLM that reflects ongoing watershed-related research projects in ARS appears in Table 1.

Table 1. Selected research interests at BLM and ARS.

Theme	Process	Management Implication
Ecology	Vegetation sustainability	Decision support systems
	Wildfire	Fuels management
	Hillslope erosion	Vegetation management
	Invasive weed production	Weed control
	Plant transpiration	Water conservation
	Plant nutrition	Vegetation management
Hydrology	Sediment transport	Water quality compliance
	Ephemeral streamflow	Runoff conservation
		Watershed stability
Climatology	Climate change	Water storage implications
		Vegetation management
	Weather simulation	Grazing-related management

Future Research

Institutional considerations

The Department of the Interior is in the final stages of developing its strategic plan for 2003–2008 (U.S. Department of the Interior 2003), as required by the Government Performance and Results Act. This law, and the related strategic goals that agencies are required to develop, mandates that expenditures and accomplishments be in accordance with agency-specific approved plans. Funding for research and scientific development will need to support the goals and outcomes set forth in the Department's strategy.

Changing demographics and related landscapes may be considered another significant force affecting research priorities. Issues related to tradeoffs between traditional commodities (e.g., grazing, timber harvesting, mineral extraction) and protection of the environment is being compounded by ever increasing population centers in the West and demands for recreation on the public lands. Within wildland areas surrounding western U.S. towns and cities, land management agencies are dealing with the need to manage numerous competing uses (e.g., off-road vehicles, camping, hiking), while protecting the ecological integrity of these lands. Research and other scientific studies will necessarily give attention to these types of impacts as the agencies continue to deal with urban expansion and its effects on the environment.

BLM expects to see moderate levels of funding available for scientific investigations as it continues to implement the science strategy and work through its science providers. With the goal of understanding the condition of public lands, the BLM Applications of Science Initiative solicits needs from field offices and selects high-priority proposals to develop fiscal year requests for funds. Funding in 2003 was added to the BLM budget, and consequently, 23 projects were funded. The Joint Fire Science Program (2003) provides additional funding to BLM in support of development of information and tools dealing with wildland fire issues. With the establishment and management of national conservation areas and national monuments, BLM continues to prepare management plans that include the identification of science opportunities and research needs.

There is a long history in the Federal government of cooperation between science providers and land managers. Future relationships between these entities should recognize two factors in developing research topics and using research results. First, all parties need to work diligently at jointly recognizing broad research topics of concern and establishing research priorities related to land management issues. Specific project proposals should be developed only after affected land managers and scientists have agreed on the problem to be addressed and related study methods to ensure a useable outcome. In particular, research planning and strategies at the national and regional levels need to focus on broader land and resource management issues and avoid extensive discussions on detailed project proposals. The resulting agreements or guidelines will allow agencies with varying roles to communicate more effectively and be more competitive in obtaining precious funds for conducting critically important research.

Second, BLM is seeing more successful research partnerships as agencies and interdisciplinary teams interact more frequently throughout the projects. From the time a resource management problem is identified, researched, and resolved, the interaction between scientists and land managers must be a continual process. Research partnerships can be positively influenced by land managers who are given ample opportunity to express their concerns and demonstrate resource management problems. Upon initiation of studies, all interests should meet frequently to review progress and ensure that objectives are either being met, or in some cases, being redefined. It is arguable that the most critical part of research efforts is the application of results to aid in solving land and resource management issues. Close interaction between the researchers and land managers in transferring research results can be realized through continued coordination (e.g., interagency, academia), such as training sessions, development of technical guidance, on-the-ground demonstrations, and peer involvement in developing management alternatives.

Research needs

Research needs are initiated through BLM's Budget Planning System (BPS). In 2004, highlights of the BPS include continued development of systems to monitor and understand changing resource conditions (e.g., remotely sensed data, GIS models) and techniques for restoring plants and watersheds damaged by invasive, non-native plants and major wildfires (U.S. Department of the Interior 2003). Due to the recent establishment of national monuments and national conservation areas managed by BLM, and the scientific values acknowledged in the establishment of these areas, scientifically creditable baseline information is needed to develop land use plans to manage resource values. Scientific investigations will be initiated or continued in order to provide information related to energy and mineral resources development in relation to the protection of hydrological resources and a variety of habitat concerns.

BLM is committed to identifying the agency's priority research needs through its science catalogues as a means of conveying these needs to the CESU affiliates and other science partners. Based on the author's review of CESU-related watershed research, general areas of interest appear to be in subjects such as managing biological resources, controlling the effects of abandoned mine land pollution on land and water, developing sustainable recreation environments, and predicting the impacts of produced waters from oil and gas development. Relative to these areas of interest and the numerous specific studies listed in the catalogues, research related to managing biological resources is worth noting. Successful management of biological resources is often expressed with vegetation attributes, which reflect soil, climate, and moisture conditions. For BLM, biological resource priorities include wildfire impacts and post-fire rehabilitation, invasive weeds inventory and control, rare and endangered species assessments and management, critical habitat assessment and improvement, and forest and range plant community sustainability.

Summary

In managing over 261million acres of public lands, BLM continues to be an active participant in watershed and rangeland research, enabling more science-based land management decisions. In issuing its Science Strategy, the agency has set forth clear direction regarding its process for identifying national and regional science needs and working cooperatively in conducting studies and research. One example is BLM's long history of working with the ARS on watershed and rangeland issues, ensuring that the results of ecological, hydrological, and climatological studies are applied to land management situations.

In reference to future research in BLM, five institutional considerations were noted. First, in accordance with the Government Performance and Results Act and related requirements to report planned accomplishments, BLM is increasingly accountable for its expenditures, including funds invested in research. Second, the once open western rangelands traditionally used for production of goods are now under heavy pressure by expanding population areas to provide resources for recreation and wilderness. Third, an important consideration related to the Federal budget is BLM's appropriations, which are expected to provide only moderate levels of funds for research and the need to be extremely targeted in funding the highest priority research. Fourth is the need for effective communications between science providers and land managers to ensure the wise investment of public funds and full utilization of research results. And, fifth is the need for science and management partnerships to be more interactive throughout the study process and in applying results.

The highest priority research needs in watershed and rangeland environments are those that will facilitate a better understanding of changing resource conditions and techniques to restore plants and watersheds damaged by invasive weeds and wildfire. Resource management priorities include studies related to wildfire and invasive weed impacts and control, critical habitat assessment and improvement, and plant community sustainability. BLM will continue to convey its priority issues and research needs through its science catalogues to CESU affiliates and other science partners.

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The CUAHSI Plan for a Network of Hydrologic Observatories

Lawrence Band, Marshall Moss, Fred Ogden

Abstract

The Consortium of Universities for the Advancement of Hydrologic Science (CUAHSI) was founded in 2001 to address and facilitate the study of critical interdisciplinary science issues identified by the hydrologic community (www.cuahsi.org). These include predictability of hydrologic processes, scaling principles and critical interfaces with atmospheric, ecological, biogeochemical and societal processes. A major impediment to this study is the lack of an adequate observational base, and the infrastructure required to acquire this base in watersheds of sufficient size. Hydrologic Observatories (HO) are one of the center pieces of the CUAHSI initiative and are conceived to directly address this gap. They are conceived as major research facilities that will be available to the full hydrologic community to facilitate comprehensive, cross-disciplinary and multi-scale measurements necessary to address the current and next generation of critical hydrologic science and management issues. Some of the topical aspects of the HO mission have been implemented in the form of a subset of the more hydrologically oriented LTER sites, the ARS and WEB experimental watersheds, and through the long term monitoring program of the USGS. However, these efforts have either not been at the scales envisioned for the HO (e.g. 10^4-10^5 km^2), have not emphasized long term measurement, or have not addressed the development of a comprehensive, multidisciplinary information base designed to address the driving scientific and management questions posed by CUAHSI. However, the HO can be designed to build on and integrate these ongoing efforts by identifying larger basins that contain longer term LTER, ARS or WEB sites, or implement more comprehensive, multidisciplinary data collection and experimentation schemes in monitored basins.

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Keywords: CUAHSI, Hydrologic Observatories, hydrologic infrastructure

Introduction

Environmental change affects hydrologic systems in ways that cannot be predicted with current knowledge and information, and that pose threats to the future of water supply, water quality, ecosystem health, sustainable use and human society. Environmental changes occurring on scales from local to global include land cover change, climate change, engineered modifications of the hydrologic cycle (e.g., large-scale water transfer or flowpath modification), alteration of biogeochemical cycles, and loss of biodiversity. These changes are superimposed on a background of natural variability, challenging our ability to forecast hydrologic events or differentiate human caused changes. One of CUAHSI's goals is to address cross-scale and interdisciplinary questions that need to be answered so society can anticipate and be prepared to deal with changes in hydrologic systems. A significantly increased investment in hydrologic observation and synthesis is needed to answer key scientific questions that cannot currently be answered with our existing research framework and infrastructure.

A national system (network) of HOs are proposed to provide spatially and temporally coordinated interdisciplinary data and infrastructure platforms for research at a new, higher level of integration. Furthermore, in light of continuous and accelerating environmental change in Earth systems, all of the HOs are to collect long-term observations on both a minimum common set of core variables, as well as state and flux variables that may be more locally or regionally important (e.g. permafrost extent, stormwater infrastructure) required to develop mass budgets in water, sediment, and biogeochemicals and trends in these budgets. Long-term observation allows separation of directional trends from year-to-year variability, discovery of thresholds or extreme events that result in surprises, and incorporates a larger number of extreme events. Equally important is spatial nesting that will allow scientific questions to be answered at multiple scales but also, importantly, investigations of scaling phenomena.

The HO network eventually will comprise a set of instrumented watersheds, spatially distributed to cover representative hydrologic regions or ecoregions. Coordination among nodes within the network will be facilitated with the CUAHSI Measurement Technology facility, the Hydrological Information System, and a national hydrologic synthesis center (www.cuahsi.org). Finally, all observatories will implement education and outreach activities and maintain close interaction with local and regional stakeholders.

The HO must be maintained as a community effort, and it will collaborate and coordinate with other appropriate observation networks (including existing hydrologic monitoring). In summary, the HO network will be a sophisticated research infrastructure to support state-of-the-art research in hydrologic sciences and related disciplines that address fundamental and critical water resource problems.

Major new infrastructure needs to be developed in support of this agenda and operated as national facilities to support hydrologic data collection, analysis and synthesis in a manner not feasible with current infrastructure. All data and information generated will be available to the full scientific community. The observatories will also be available for use to the full community as a research facility by competitive proposal and will include:

1. Hardware: state-of-the-art and developing field and laboratory instrumentation that can provide measurement, monitoring and analysis of basic stores, fluxes and transformations of water and associated biogeochemical constituents within large watersheds across a spatial/temporal scale spectrum, as well as tracer and isotopic characterization of water source and residence times. This will be coordinated with the CUAHSI Measurement Technology facility.
2. Spatial data infrastructure – coordinated with the CUAHSI Hydrologic Information System (HIS): characterization of the structure of watersheds including high resolution topography, channel network pattern and morphology, soil, aquifer, land use/land cover, and socioeconomic information, as well as facilities to analyze and correlate data of multiple space and time scales and to integrate, manage and disseminate information,
3. The interdisciplinary environment necessary to achieve an intellectual synthesis, coordinated with the CUAHSI Hydrologic Synthesis Center.

The CUAHSI framework

Recent NSF and NRC white papers on the future of hydrology outline a set of science initiatives and questions focusing on variability and predictability in the hydrologic cycle at multiple scales, the coupling of hydrologic processes with atmospheric and ecosystem processes, and interactions of hydrologic systems with human societal activities (NRC 1991, NRC 1999, Gupta et al. 2000, NRC 2001a, USGCRP 2001, NRC 2001b). CUAHSI has built on these points to formulate a major research theme: to develop predictive understanding of the storage, flux, and transformation of water, sediment, and associated chemical and microbiological constituents under both natural and human-altered conditions. One of the primary points made by the NRC reports and that contributed to the formation of CUAHSI was that research in hydrologic science is observation limited and that the universities are unduly constrained by a paucity of infrastructure to support that research. The CUAHSI Science Plan calls for the establishment of a network of Hydrologic Observatories (HO) for the development of critical observational bases to facilitate formulation and testing of hypotheses posed by the interdisciplinary hydrologic science research community. As the next level of hydrologic research that has been identified as necessary to make significant progress on pressing scientific and societal questions is **integrative** the HO network must incorporate the following four major themes outlined in the CUAHSI Science Plan:

- **Coupling** of the water cycle across disciplinary boundaries with ecosystems and atmospheric, biologic, geologic, and social processes
- **Cross scale** relations, from the level of soil pedons to mesoscale atmospheric systems
- **Hydrologic interface** behavior, (Land surface and atmosphere, land surface and groundwater, surface water and groundwater, surface water and land surface, land-atmosphere, saturated-unsaturated zone)
- **Predictability** of hydrologic processes (e.g. flood magnitude, drought onset and duration, contaminant transport and arrival times)

A combination of longer term, lower density monitoring and short term, intensive field campaigns will need to be designed to support the over-arching science questions that unite the CUAHSI infrastructure initiatives. To address large-scale water and material balances within the natural organization imposed on the land surface by river and stream systems, each hydrologic observatory will be developed over a regionally significant river basin. These watersheds will be large enough to address spatial and temporal variability in regional land surface conditions and the feedback and coupling between the land surface and the atmosphere, on the order of 10^4 to 10^5 km^2. This is considered to be the minimum area within which mesoscale features and land/atmosphere feedbacks may be discerned and tracked. In addition, the recent launch of the Earth Observing System (EOS) generation of satellites include significant global scale monitoring systems with resolutions of 1 km or lower. The scale of the HO will be sufficient to observe and track satellite derived land surface state and flux variables (such as those generated by the MODIS TERRA and AQUA systems), as well as correlate these products with HO generated observations, as well as supporting research in the assimilation of remotely sensed information into regional watershed models.

Each HO will be located in a distinct hydroclimatologic setting, and instrumented with a nested sampling design capable of addressing the order of magnitude variations in the length scales of key processes. These will range from nested observations through the depth of the rooting zone to characterize available soil water dynamics and soil water controls on ecosystem and biogeochemical processes, to regional rain radar, and stream gauge networks capable of generating the information required to study the scaling behavior of flood peaks or low flow patterns.

Instrumentation and facilities in each site needs to be planned to support broad, community defined science questions, as well as to provide base infrastructure required by more specific, individual or group based projects. These latter programs will range from individual Principal Investigator led projects, through the type of large scale intensive field campaigns the scientific community has engaged in (e.g. BOREAS, FIFE, LBA), potentially involving hundreds of scientists. The presence of the HO will facilitate and encourage these types of investigations by making available both basin scale information and instrumentation that would otherwise be unavailable to individual or small groups of researchers, and by providing the context and observational background in large watersheds that is currently missing. Close coordination with the CUAHSI Hydrologic Information System (HIS), the Measurement Technology Facility and Hydrologic Synthesis Center will be necessary to build a network capability that both ties together the individual HO into a continental observing and experimentation system, as well as wide distribution of information generated within the HO to the full hydrologic community.

HO Design

Ideally, there would be between ten and twenty observatories in the network phased in over a period of time to allow adaptive growth and evolution of the network. This scale of observation would allow detection of trends, positive and negative correlations between regional trends, and continental-scale hypothesis testing. A common conceptual framework is derived from using a specific set of science questions distilled from the CUAHSI science agenda as design drivers. The driver questions have societal and scientific relevance ranging from local to continental and global levels and the nested network design will permit synthesis of the knowledge derived across this scale range.

The set of program drivers that will guide infrastructure and sampling design are drawn from the major scientific questions posed by the set of NRC reports and refined into CUAHSI themes. Program drivers are not exclusive questions the HO will focus on, but will span the hydrologic continuum and promote integrated intellectual and technical approaches to information generation. In this sense they are meant to be sufficiently broad to be representative of the types of cross disciplinary and cross boundary questions the community has outlined and are likely to pose, noting that the function of these drivers is to guide major infrastructure development. These drivers include, but are not limited to:

1. **Land-surface/atmosphere.** Does water cycling within a basin contribute significantly to the precipitation that falls in the basin, and do these feedbacks intensify wet and dry periods?
2. **Land-surface/groundwater**. How do atmospheric and surficial processes control groundwater recharge and how can this knowledge be used to develop quantitative estimates of recharge at the scale of thousands of square kilometers?
3. **Groundwater/surface water**. How can the exchange of water between the regional aquifer, alluvial aquifer and surface water be quantified and its residence time in each domain estimated, as these properties control many biogeochemical properties and influence aquatic ecosystems?
4. **Hydrologic extremes.** How do human modifications of the local hydrologic system (both directly and indirectly by changing the land surface) influence the likelihood and intensity of drought and floods relative to global climatic phenomena such as ENSO?
5. **Land use effects on biogeochemistry**. How does land cover and use influence the loading, transport and transformation of biogeochemicals in large watersheds?

The design is being led by a team of scientists with expertise in each of the relevant areas of hydrologic science inherent in the design questions. The prototype will frame and resolve design decisions that are required for full implementation, including the administrative and management infrastructure to support professional staff, cooperative agreements with existing experimental watersheds and data collection agencies, and university investigators.

The design team members are soliciting input from the general research community as needed. After the completion of the prototype study, competitive proposal design grant applications will be solicited from existing and developing research consortia, in order for teams to take the prototype design and apply its procedures in other settings with the anticipation of competing for one of the first awards to implement a fully operational hydrologic observatory.

CUAHSI is presently developing a conceptual prototype design on the 14,000 km^2 Neuse River basin in North Carolina. For a set of the drivers, a larger area surrounding the basin may need to be utilized in order to approach sufficient process length scales (e.g. land-atmosphere interactions and precipitation recycling). For the Neuse, the likely area to expand to would be the Pamlico-Albemarle Sound drainage area, which is a NAQWA basin. Other drivers may require smaller, more intensively studied catchments within the watershed.

The Neuse River was chosen for the development of the conceptual design of a hydrologic observatory because of: (1) the leadership and support of the North Carolina Water Resources Research Institute (NCWRRI) and the highly capable participation of many scientists from the hydrologic science community in the Research Triangle; (2) the robust suite of both historic and ongoing hydrologic data collection efforts (by Federal, State, and university scientists), and (3) the fact that the Neuse River Basin contains a variety of conditions that are found in many other regions of the country and is not dominated by one particular set of topographic, pedologic, and climatologic conditions. As mentioned above, the Neuse is located within an existing NAQWA focus watershed and contains a set of smaller, more intensively instrumented catchments (although no LTER, ARS or WEB sites). Ongoing monitoring funded by federal and state agencies is geared towards the study of flood hazard and point and nonpoint source nitrogen loading and export from the basin into the Pamlico-Albemarle Sound complex, the second largest estuary in the country and a vital fish nursery for the Atlantic.

A national design team with strengths in different specialties has been assembled to take primary responsibility for the prototype design, with support from a set of other university and government scientists. The primary design team includes:

- Dr. Ken Reckhow, Duke University and NC-WRRI (water quality); Team Leader
- Dr. Chris Duffy, Penn State (hydroclimatology, saturated-unsaturated zone interactions)
- Dr. Jay Famiglietti, UC-Irvine (atmosphere-land interactions)
- Dr. David Genereux, NC State (groundwater-surface-water interaction)
- Dr. John Helly, UC San Diego (information systems)
- Dr. Witold Krajewski, Iowa (hydrometeorology)
- Dr. Dianne McKnight, Colorado-Boulder (biogeochemistry)
- Dr. Fred Ogden, Connecticut (floods/geomorphology)
- Dr. Bridget Scanlon, University of Texas, TBEG, (infiltration/vadose zone hydrology)
- Dr. Len Shabman, RFF (economics/social science)

It is important to note that the prototype design in the Neuse River Basin is envisioned as a design exercise, and does not indicate that an HO will be established in this basin. A set of other sites around the country will be considered for the implementation phase through competitive proposals.

Interactions with Existing and Proposed Monitoring Programs

The nested observation systems would be planned to operate within the framework of the existing USGS surface water monitoring network, which will provide multi-annual and decadal records for regional watersheds. Sparser sampling has been operated for groundwater levels and sediment, nutrient and contaminant transport in surface and subsurface systems for watersheds of similar size to envisioned HO, with notable programs including the National Water Quality Assessment (NAWQA) (http://water.usgs.gov/nawqa) and the National Stream Quality Accounting Network (NASQAN) (http://water.usgs.gov/nasqan). It is important to note that while the USGS monitoring network will provide important context and potential baseline information to develop the HO, it is designed primarily for purposes of resource characterization. Therefore, the existing USGS surface and groundwater monitoring, while providing significant scientific data resources, are also not (by themselves) designed to respond to the major scientific initiatives envisioned. A similar assessment holds for existing NOAA-NWS monitoring networks.

On the other end of the spectrum, the nested sampling requirement suggests that initiation of an HO as a larger watershed containing an existing experimental watershed site would provide significant leverage for achieving cross-scale synthesis. Some of these sites have been operated for decades, and are often instrumented with high spatial density for measurement of multiple components of the hydrologic cycle. An important component of the HO network will be to develop and improve cooperation among the field and research programs of various agencies and with the academic community. However, most of these sites are well below 100 km^2, such that they would be very useful to incorporate into an HO, but are not of sufficient scale to meet the needs of the science questions posed.

Proposal Design Grants

Lack of experience within the community in operation of these facilities at the envisioned nested space and time scales, with the coordination of observations and experimental measurement between disciplines, and the lack of baseline information for many candidate watersheds suggests six month proposal design grants. While this information may exist for a set of watersheds at the envisioned scales, the proposal design grants would facilitate other proposals and allow better refinement of the operational plans for these facilities, to assemble available data and specify what new monitoring needs to be initiated. This would provide the ability to integrate baseline information for a larger set of watersheds, including land cover, stream flow, groundwater levels and atmospheric information, as well as coordinate appropriate partnerships between universities, government agencies, NGOs and community groups and to define regional versions of the basic science drivers.

Considering science questions focussed on water stores and fluxes, coupled with energy, sediment, nutrient balance and transport at multiple scales, important questions that require resolution are what instrumentation and sampling densities may need to implemented to measure specific state and flux variables in the presence of significant, multiple scale heterogeneity in climate, soils, land use, geomorphology and aquifer conditions. The development of full proposals with realistic cost estimates will require assessment of existing infrastructure within a watershed, estimates of additional infrastructure requirements and costs relative to stated science goals, and potential sources of cost-sharing with other agencies and communities with existing interests in the watershed.

The proposal design grants will be made available to develop the consortia required between universities, state and local government, federal agencies, NGOs and community groups, organize and explore available data and information, and develop appropriate research emphases centered on candidate watersheds.

Summary and Expected Results

The goal of the CUAHSI Hydrologic Observatory network is to provide the observational and experimental basis necessary to approach the set of multiple-scale, interdisciplinary science questions that have been identified by the hydrologic community, with sufficient flexibility to adapt and distribute developing technology in measurement and information science, as well as to identify and address emerging questions within the hydrologic sciences. The successful design and implementation of a Hydrologic Observatory network, in conjunction with the CUAHSI Hydrologic Information system and Measurement Technology Facility will provide the interdisciplinary hydrologic community with enhanced infrastructure required to make substantial progress on critical scientific and policy related questions surrounding current and future distribution, circulation and characteristics of water and its constituents within the hydrologic cycle. As the community has concluded that the current state of our science is observation limited, design and delivery of this infrastructure as community facilities is a key task that CUAHSI has undertaken to provide. The facilities will be available for use by the full hydrologic community by competitive proposal, and all information generated within the HO which makes use of CUAHSI facilities will be publicly available within a specified time period. The HO network will have direct interactions with a set of existing experimental watershed and hydrologic monitoring programs by formal arrangement, by geographic nesting of the HO either within larger monitored watersheds (e.g. NAQWA) or by containing smaller, more intensively monitored catchments (e.g. ARS, LTER). The value-added advantages of the HO network is strongly leveraged with these existing programs, and provides the cross-scale and cross disciplinary facilities currently required for the current and future generation of hydrologic questions and issues, but largely not available to the scientific community, and specifically to the university research community.

Acknowledgments

Material for this paper drew significantly from notes and draft working papers of the CUAHSI Standing Committee on Hydrologic Observatories. Individuals on that committee that contributed significantly include Marc Williams, Roger Pielke, David Goodrich, Diane McKnight, Norman Miller, Ken Potter, Ken Reckhow, Doug Kane, Bridgett Scanlon and Berry Lyon. Ken Reckhow and Rick Hooper provided useful comments on the manuscript.

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An Overview of the Strategic Content of EPA's Watershed Research Program

Lee A. Mulkey, Thomas O. Barnwell, Steven Hedtke, Rochelle Arraujo, Michael Slimak

Abstract

Introduction

The Environmental Protection Agency (EPA) has appropriately directed much attention to watersheds during its tenure as the Federal Agency charged with protection of human health and the environment. Watershed research as a vehicle to understand the interaction of the hydrologic cycle and human activities has also enjoyed a vital and active role in EPA's tenure. Specifically, EPA's Office of Research and Development (ORD) is pleased to share in a conference that highlights the role that experimental watersheds and related interagency research has played in our collective interests in informing public policy, increasing knowledge about watershed processes, and promoting stewardship of land, water, and biota – in a phrase, protecting, improving, and sustaining watersheds. The purpose of this paper is to briefly review an EPA-ORD perspective on our progress to date, to describe the EPA research agenda in this arena, and to challenge the watershed research community to address four fundamental hypotheses both as individual ideas and in an interdisciplinary manner. A list of references is provided for those interested in more detailed descriptions and the data upon which this paper is based.

EPA's interest in watersheds and water resources is manifold and flows from requirements of the Clean Water Act (CWA), the Safe Drinking Water Act (SDWA), the Resource Conservation and Recovery Act (RCRA), the Food Quality Protection Act (FQPA), the Federal Insecticide, Rodenticide, and Fungicide Act (FIFRA), the Superfund, the Clean Air Act (CAA), the National Environmental Policy Act (NEPA), and a number of other federal laws and Executive Orders that influence development of EPA regulations, policy directives, and guidance documents (e.g., Threatened and Endangered Species Act, Coastal Zone Management Act, E.O. on Invasive Species). While a detailed analysis of the cross connections between each of these directives and watershed research is no doubt useful if not enlightening to many, this paper will concentrate on the context provided by the Clean Water Act. This is convenient and useful because the goals and requirements of the Clean Water Act integrate the outcomes of many of the other individual interests and legislative requirements; watershed hydrology and water quality integrate atmospheric deposition, land-based activities, ground and surface water dynamics, and terrestrial and aquatic ecology.

Context and Conceptual Basis for Watershed Research Goals

In the interest of brevity and strategic perspective, it is useful to consolidate the context of EPA's watershed research into a few foundational concepts and program goals. Virtually all EPA programs can be characterized as having components that should: 1) assess the condition of the environment; 2) diagnose apparent problems and forecast alternative solutions; 3) assess current and future risks; and 4) develop remedies and strategies to protect and restore. ORD's watershed research programs are now organized around these components as illustrated in Figure 1. The programmatic goals for this research, expressed as desirable outcomes are as follows:

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- Condition Research – The states and tribes assess the condition of all their waters in a scientifically-defensible and representative fashion that allows aggregation and assessment of trends at multiple scales.

- Diagnosis and Forecasting Research – Federal, State and Local managers can diagnose cause and forecast future condition in a scientifically defensible fashion to more effectively protect and restore valued ecosystems.

- Protection and Restoration Research – Federal, State and Local managers can protect and restore aquatic ecosystems using scientifically defensible methods.

- Assessment Research – Federal, State and Local managers can conduct scientifically defensible assessments of current and future condition, causes of impairments, and management alternatives.

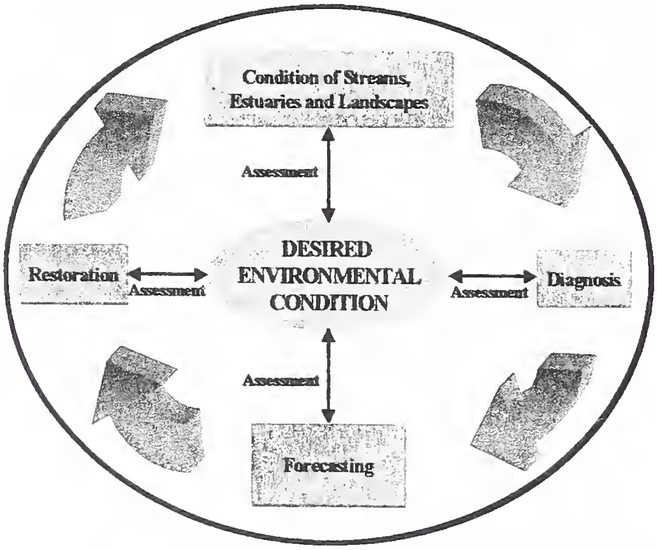

Figure 1. Conceptual approach to ORD research on watersheds.

The programmatic goals outlined above are driven by a series of research questions that must be evaluated. Again, in the interest of brevity and conciseness, these research questions are aggregated to the same level as the programmatic goals and are given below:

- Condition Research - What is the current ecosystem condition, what are the trends in condition, and what stressors appear to have been responsible for harm or deterioration?
- Diagnosis and Forecasting Research - How do biological, chemical, and physical processes affect the condition of ecosystems, and how can we most accurately diagnosis problems facing ecosystems and forecast future effects?

- Assessment Research - What are the relative risks posed to ecosystems by stressors, alone and in combination, now and in the future?

- Protection and Restoration Research - How can we most effectively reduce risks to protect ecosystems and restore them once they have become degraded?

What Have We Learned to Date and What Remains as Challenges?

The data, experiences, models, and analyses presented during this Conference will serve in part to summarize our progress to date in watershed research with particular emphasis on the role that experimental watersheds have played in those endeavors. Part of understanding the current and potential future role of such science is to survey the policy-relevant findings that should inform research planning and experimental designs. Consider the following apparent policy-relevant situation:

- over 20,000 waters identified by States as impaired due to one or more pollutants.

- a shift from point source discharges as the major source of pollutants to nonpoint sources.

- an increasing use of biological indicators and metrics as the preferred method for determining the current condition and desired water quality criteria for aquatic ecosystems.

- an increasing awareness of the importance of landscape- and watershed- scale processes and activities as determinants of water quality.

- an increasing awareness of the role of atmospheric deposition and multimedia sources as determinants of water quality.

- an increasing awareness of the role of habitat alteration as a cause of aquatic ecosystem impairment.
- an increase in human-health risks from apparent ecosystem responses to stressors, particularly pathogens.
- pressures to increase the efficiency and cost-effectiveness of watershed management implementation.
- an increase in the role of citizen stakeholders in setting watershed management goals and in implementing action programs at the local and watershed levels.
- increasing calls for more efficient, more nearly accurate models and methods, and more explicit representation of uncertainties in decision-making processes used by EPA and State Agencies.
- lack of systematic and statistically-robust evidence that best management practices (BMPs) for non-point source controls are working.
- increasing calls for outcome-based implementation and accountability.
- increasing calls for documentation of the economic benefits derived from government - funded approaches to meet Water Quality Standards and Goals.
- an increasing awareness of the role of invasive species as a cause of aquatic ecosystem impairment.
- integrated assessments for allocation of restoration resources to support water quality standards attainment within the context of socioeconomic factors.

The trends and challenges cited above drive the strategic content of the current EPA watershed research agenda. Many of these trends have been generally acknowledged by others and have shaped ongoing and previous research programs. That said, previously developed and current science and technologies are apparently not yet able to meet all the challenges for the following reasons:

- BMPs and other nonpoint source control measures have rarely been evaluated for their effectiveness in achieving improved water quality (particularly biological condition), rather only for pollutant load or concentration reduction.
- previous focus on chemical and pollutant-specific determinants of water quality does not fully address biological condition.
- the data, analysis tools, and assessment methodologies for landscape and regional scale processes are leading edge research areas not yet exploited to solve problems.
- atmospheric deposition of nutrients (e.g., nitrogen) and toxic substances (e.g., mercury) have not been integrated into watershed management science.
- biological indicators and measurements of habitat alterations, particularly related to flow and sediment, have only recently emerged as issues.
- the causes and control of increasing hazardous algal blooms (HABs), *Pfiesteria,* and pathogens are not fully known.
- ecological risk assessment guidelines, public awareness tools, and risk communication programs are largely new and rarely applied.
- free market based and economically robust risk management systems and frameworks are limited in scope and application.
- many models and decision-support tools are often cumbersome to apply, require data all too often unavailable, and fail to explicitly address uncertainty.
- guidance for setting action and management priorities to achieve outcome-based goals remains problematic.
- water quality management solutions that also lead to sustainable ecosystems and related economies are desirable; the ability to design and implement such solutions is lacking, in large part because of scientific limitations.

- economic valuation of water quality benefits cannot yet be applied to action programs and regulatory activities.

Proposed Hypotheses for Watershed Researchers

Public and natural resource managers' expectations for watershed research and operational watershed management programs are appropriately high and are increasingly interpreted and expressed via multiple disciplinary perspectives. While some perspectives are longstanding, new ones are emerging that give rise to the need for more collaborative and interdisciplinary research. Much remains to be done across the board to be sure; much can be gained by consideration of the interdisciplinary nature of some new questions. From EPA's perspective it is useful to relate all such views to the reality that water quality and water availability remain as national problems deserving continued high priority and investments. This perspective also recognizes that our collective progress in research and in progress toward improving both water quality and availability are substantial. It is also understood, and desirable, that operational watershed management programs provide adaptive learning platforms over time. Here is a set of perspectives or general hypotheses that now face watershed scientists.

Watershed management and restoration to meet local, regional, and national goals for water supply, water quality, and ecological integrity on a sustained basis are not yet achieved because:

- Monitoring hypothesis – Robust and unbiased national, regional, and watershed estimates of the condition and trends are not available to set priorities, efficiently allocate resources, and measure program effectiveness. This hypothesis calls for developing appropriate indicators for the goals, developing improved and cost-effective statistical sampling designs, and conducting assessments that provide robust statements of the magnitude and distribution of conditions.

- Biogeochemical hypothesis – Multiple processes that interact at multiple scales are not sufficiently understood and not sufficiently predictable. This hypothesis calls for continued process and experimental research that both elaborates complexities and that yields more robust and reliable models. Process and systems ecology must be included here. Controlled experiments are needed over appropriate time periods and across a wide array of site-specific conditions and scales.

- Engineering and hydrology hypothesis – Watershed management practices, structures, and technologies can be designed and implemented if the design goals and requirements are known, resources are available, and if the desired hydrologic conditions are known. This hypothesis anticipates a shift from a technology-based design approach (that is, an approach driven by the availability of technology, much like the prevailing Best Management Practice approach) to a performance-based approach (that is, designing to meet a water quality goal or standard). Implementation of this approach will also provide useful economic cost information that further drives innovative technology development.

- Economics and social science hypothesis – The benefits of achieving goals are less than the costs of meeting the goals. This hypothesis derives from the perspective that current outcomes flow from the conscious tradeoffs people make when faced with limited resources or perceptions about the choices. Economic valuation of benefits must be developed and a full array of market mechanisms and incentives must be elaborated and implemented to provide additional tradeoff options. Among such approaches include trading schemes for nonpoint and point sources.

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The following list of references, as interpreted by the authors, provide the basis for the strategic content of the EPA's Office of Research and Development (ORD) research program on watersheds and watershed-related issues. This research is conducted by intramural scientists and engineers within ORD and with academic partners via ORD's competitive grants program. ORD also partners with other federal agencies engaged in watershed research.

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Overview of the Water, Energy, Biogeochemical Budgets Program of the U.S. Geological Survey

Mary Jo Baedecker

Abstract

Small watershed studies serve as an important mechanism to understand changes in a broad range of hydrologic environments at a scale where multiple processes can be understood. The U. S. Geological Survey's (USGS) Water, Energy, and Biogeochemical Budgets (WEBB) program was designed to understand processes in small watersheds located in geographically diverse environments that represent a range of hydrologic, ecologic, and climatic conditions. Five watersheds have been the focus of long-term monitoring and research since 1991, with emphasis on changes in hydrochemistry caused by natural and anthropogenic disturbances such as atmospheric deposition, land-use change, and climate change. This paper provides a review of recent studies from the WEBB sites related to hydrologic trends and the influence of climate, biogeochemistry and chemical weathering, and geomorphology and sediment disturbances. Our program seeks to scale-up information from these intensive studies on individual watersheds to understand processes in larger systems. One of the challenges for scientists is to demonstrate transferability of their findings beyond an intensively studied watershed to other watersheds with sparse data.

Keywords: small watersheds, hydrology, biogeochemistry

Introduction

The U S. Geological Survey's (USGS) Water, Energy, and Biogeochemical Budgets (WEBB) Program was designed to understand processes in small watersheds in geographically and ecologically diverse environments that represent a range of hydrologic and climatic conditions. The five WEBB watershed sites described below have diverse precipitation and temperature ranges and, although they are located in relatively pristine environments, they are impacted to varying degrees by human activities. The selection of sites was based in part on the existence of long-term data sets and infrastructure provided by other federal agencies on which the program could build. The WEBB watersheds have been the focus of intensive monitoring and research since 1991.

Specific goals of the program are (1) to understand the coupled effects of the geologic and physiographic framework, land use, landscape characteristics, and climatic setting of a watershed on the water balance and associated generation of streamflow or fluctuations of lake levels; (2) to increase our understanding of temporal and climatic factors that affect solute input, export, and retention; and (3) to quantify the impacts of human and natural disturbances on terrestrial and aquatic ecosystems, such as the effects of land-use change on erosion and water- and soil-resource degradation, and the effects of atmospheric deposition of forest health and aquatic biota. An overarching goal is to compare responses observed among the WEBB sites, providing information about controls on ecosystem processes that allows the extension of work on these individual small watersheds to other areas and to larger regions. The purpose of this paper is to provide examples of recent USGS studies at the sites and to discuss the future challenges. More information about the

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program and the individual sites can be found at http://water.usgs.gov/ nrp/webb/

WEBB Research Watershed Sites

Below is a brief description of each site in the Program. In addition to being supported by the USGS, all of the WEBB sites have received substantial support from partnerships with other state and federal agencies and with academic institutions.

Sleepers River watershed

Located in Vermont, it is predominantly a typical northern hardwood forest developed on glacial till with about one-third of the watershed in pasture land used for dairy farming. Research in the watershed was begun by the Department of Agriculture 's Agricultural Research Service in 1959 and was supported for many years by the U.S. Army Cold Regions Research and Engineering Laboratory (CRREL).

Luquillo experimental forest

Located in Puerto Rico, it is a tropical rainforest developed on volcanic rocks. It is administered by the U.S. Forest Service and is supported by the National Science Foundation (NSF) as a Long-Term Ecological Research (LTER) site and is an UNESCO-designated International Biosphere Reserve. WEBB research is also conducted in a nearby urban and agriculturally developed watershed.

Panola Mountain watershed

Located in Georgia, it is a forested watershed in the Piedmont region, with bedrock outcrops covering a small fraction of the area. It is in a State Department of Natural Resources Conservation Park.

Loch Vale watershed

Located in Colorado, is a subalpine watershed within Rocky Mountain National Park. It is predominantly forested with bedrock outcrops on the steep valley sidewalls. It is administered by the National Park Service and is part of the interagency National Atmospheric Deposition Program/ National Trends Network. In addition, it is part of an UNESCO-designated International Biosphere Reserve.

Northern temperate lakes watershed

Located in Wisconsin, it is in a forested upland area characterized by a moderately dense distribution of lakes, typical of the low-relief, glaciated terrain in over much of the upper Midwest and southern Canada. The network of lakes in the watershed is interconnected by ground water. The watershed is supported by NSF as an LTER site and operated through the University of Wisconsin.

Current Research at the WEBB Sites

Research at these sites focuses on interactions of the geology, climate, hydrology, and ecology under the broad topics of hydrologic trends and the influence of climate, biogeochemistry and chemical weathering, and geomorphology and sediment disturbances. Long-term data collection has been a key component of the WEBB program to examine trends and the influence of climate and human intervention. Monitoring data, mass-balance budgets, and watershed models are being used to understand controls on surface- and ground-water contributions to streamflow at nested catchments, to relate the hydrology to biogeochemical cycling and climate variability, and to provide the basis for examining fluxes and mass balances of water and solutes.

Biogeochemistry and chemical weathering

Carbon and nitrogen cycling have been a major focus of the studies in the WEBB watersheds since the beginning of the program in 1991. A recent study on carbon dioxide and methane fluxes from the Loch Vale watershed demonstrated that although over the last 7000 years a subalpine wetland in the watershed has accumulated carbon, the net carbon flux during 1996 to 1998 was from the wetland to the atmosphere (Wickland et al. 2001). This may reflect a change in the primary productivity of the wetland and the study demonstrates the annual variability of carbon fluxes. Studies on nitrogen (nitrate and ammonia) at the same watershed for a 6-year period indicated that annual atmospheric deposition was in excess of the annual export of nitrogen. Concentrations of nitrate in different landscapes compared to snowpack and snowmelt for 1994 to 1995 indicated that the highest concentrations were found in water from the tundra and talus (Figure 1) (Campbell et al. 2000). The results suggest that alpine watersheds are sensitive to moderate rates of atmospheric deposition of nitrogen in part because minimal vegetation limits nitrogen assimilation by plants in the talus, which is well-connected to the surface waters. The export of nitrogen is a combination of direct flushing from atmospheric sources and from biogeochemical processes. Further studies on the delta^{15}N and delta^{18}O of nitrate confirmed that much of the nitrate exported in alpine streams is transported from the talus deposits and that nitrate from microbial nitrification is a major component of the total nitrate (Campbell et al. 2002).

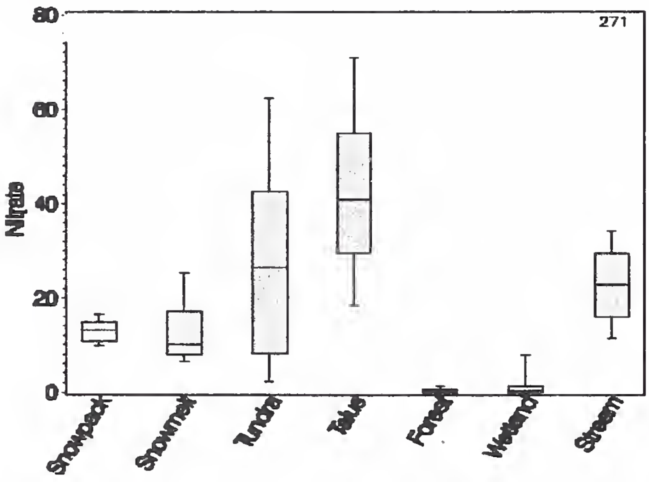

Figure 1. Distribution of nitrate concentrations (µeq L^{-1}) in atmospheric deposition (snow and snowmelt), surface water (tundra rivulets and sub-basin streams), and ground water (talus, forest, and wetlands). Boxes represent 25th, 50th (median), and 75th percentiles of population; whiskers represent 10th and 90th percentiles. [Modified from Campbell et al. 2002].

With the interest in mercury in the environment stemming from atmospheric deposition, an investigation at the Sleepers River watershed was expanded to include monitoring for mercury along with dissolved organic carbon (DOC) and nitrate during the 2000 snowmelt period. The mercury concentrations strongly correlated with the DOC concentrations throughout the snowmelt period despite the large difference in concentrations (Figure 2). These data suggest that either the mercury and DOC share a common source or the mercury, through association with the DOC, is not removed by the soils (Shanley et al. 2002b). The nitrate peaked in streamflow during the initial snowmelt and then progressively was depleted. A high mercury concentration in the particulate phase was occasionally observed (up to 16 ng/L) compared to an event maximum of 2 ng/L dissolved mercury. These results indicate that mercury fluxes in watersheds may be controlled by highly episodic mobilization during snowmelt periods and that the contribution from particulate sources may be significant.

In weathering experiments run for five years, data from flow-through columns with fresh granitoids from the Loch Vale, Panola, and Luquillo watersheds indicated significant and selective temperature effects on rates of chemical weathering. Effluent Ca, Mg, and Sr concentrations did not exhibit positive correlations with temperature, which is likely due to competing processes such as nutrient cycling and acidification (White et al. 1999). The result that temperature does not significantly impact natural silicate weathering rates has implications for atmospheric CO_2 drawdown by weathering of silicates.

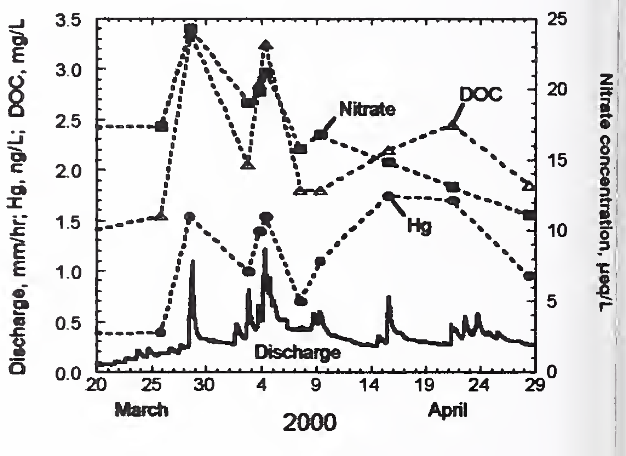

Figure 2. Discharge in Sleepers River Stream B during the 2000 snowmelt period, with concentrations of dissolved Hg, DOC, and nitrate. Note the diurnal snowmelt peaks and larger rain-on-snow event peaks, and the emphasis on high-flow sampling. [Modified from Shanley et al. 2002b].

Hydrology, geomorphology, and sediment disturbances

The WEBB data have had very practical uses and have contributed to the fundamental understanding of how watersheds function. For example, in Puerto Rico, one hundred years of hydrologic data from small watersheds were scaled-up to characterize drought and water resources throughout the island. Anomalously low rainfall in the 1990s had an economic impact that was more severe than previous droughts. Changes in the socioeconomic conditions appear to have caused the difference between the recent droughts and historic droughts. Although there were droughts recorded in the last hundred years, the population, public- supply withdrawal, and per capita consumption of water have been increasing and reservoir capacity has been declining over this interval (Larsen 2000).

One goal in using mixing models to describe the hydrologic response is to understand the sources and flow paths of water in small catchments. A detailed study at the Panola Mountain WEBB site during a three-day storm event quantified the contributions of geographic sources of stormflow to stream runoff—outcrop runoff, hillslope runoff, and riparian ground-water runoff (Burns et al. 2001). Riparian ground-water runoff dominated stream runoff on the ascending and receding limbs of the stream hydrograph, whereas runoff from an outcrop dominated stream runoff during peak-flow conditions. Controls on the contributions and the variability of the sources of snowmelt water to streamflow were examined at the Sleepers River site. In 1993 the meltwater input to the stream ranged from 41 to 74 percent, whereas in 1994 the input ranged from 30 to 36 percent (Shanley et al. 2002a).

The Northern Temperate Lakes site has the flattest landscape of the 5 WEBB sites and is a region where the dominant surficial deposits are highly permeable glacial outwash sands. The ground-water system is interconnected and can influence the water budget, nutrient budget and acid buffering capacity of the lake systems. Simulation of lake/ground-water interactions with mathematical models is essential to understand the hydrologic system. Recent work has extended the modeling approaches by formulating an analytic-element lake package for use with ground-water models, such as MODFLOW (Hunt et al. 2003).

In a comparison of four watersheds in Puerto Rico, an examination of land-use conversion to agriculture and landslides in a small montane watershed outside the forested area demonstrated an extreme case of anthropogenic disturbance in the humid tropics. In the watershed, 94 percent of the original forest cover had been eliminated by the 1940s. Based on field surveys and examination of aerial photographs, there were more than 2000 landslides identified over the last 180 years. The landslides were attributed to highly weathered bedrock, episodic heavy rainfall, and intense land-use practices (Larsen and Roman 2001). The study estimated the amount of mass wasting and suggested that the problem will continue for years despite efforts to reforest hillslopes.

Comparative studies and modeling

Long-term data from the WEBB sites are currently being used to compare and contrast results from these five sites and from other watersheds. Comparative mass-balance budgets that describe the inputs and outputs to the watersheds and the changes in storage for the five sites are being developed. These results will lead to a better understanding of sediment transport and of watershed processing of solutes. Watershed modeling is a recent component of the WEBB studies. At the Panola site, a Dynamic TOPMODEL (a rainfall-runoff model) was used to predict streamflow and catchment responses (Peters et al. 2003). A watershed model is being tested and applied to identify how specific hydroclimatic conditions result in variations in net fluxes of conservative and reactive solutes. This model was built in the Modular Modeling System (Leavesley et al. 1998) and simulates precipitation and temperature distribution, canopy interception, snowpack processes, evapotranspiration, and streamflow-generation mechanisms. New modules, built on the PHREEQC geochemical engine (Parkhurst and Appelo 1999), are being developed and incorporated to simulate geochemical processes and solute fluxes. A first step describing relations between hydrologic conditions and net solute fluxes across the diverse hydroclimatic regimes represented by the WEBB watersheds is presented in this volume (Webb et al. 2003).

Challenges for Future Studies in Small Watersheds

Long-term data from small watersheds are needed to understand the interaction of physical, chemical, and biological processes at various temporal and spatial scales. Small watersheds can be instrumented for detailed experiments and data collection and are small enough to study inputs, outputs, and apply mass-balance and modeling approaches. The future challenge is for scientists to engage in more comparative studies and to demonstrate how data and interpretations from one small watershed can be transferred to watersheds with less data or to larger systems. What are the key components that can lead to the transfer of information from an intensively studied watershed to one with sparse data?

Environmental policy and management decisions, such as those related to freshwater availability, greenhouse gases, atmospheric deposition, nutrient enrichment, and biodiversity, need a scientific underpinning to be successful. Although understanding processes such as weathering, sediment transport, and geochemistry in small watersheds is important, it is equally important to understand how such processes vary between watersheds and how the information and techniques can be scaled-up. A challenge for future work at small watersheds is the growing trend to devote resources to short-term investigations rather than examine detailed processes that require long-term monitoring. By demonstrating a wider application of techniques and methods used in pristine watersheds to environments impacted by humans, the need for long-term data collection and monitoring activities can be demonstrated.

New methods of investigation are continually being developed in small watershed studies. The application of a variety of new isotope techniques (for example, ^{35}S and $^{87}Sr/^{86}Sr$) in addition to the stable isotopes of N, O, S, H, and C can add new dimensions to understanding processes and biogeochemical cycling. Additional studies of trace metals and nutrients can provide insight as to how some of these metals enter hydrologic systems and end up in aquatic ecosystems and food chains.

To maintain the excellent watershed networks for collection of long-term data, it is important to expand the original justification for starting many of these sites from studying acidic deposition at the small watershed scale to examining the long-term effects of natural processes and of human intervention. Small watershed investigations should not be viewed as studies of one site but as part of a study of larger landscapes. Many of our studies at the USGS and at other watershed sites have started along that path.

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Technology and Watershed Planning

Watershed Planning with the Facilitator Decision Support System

Philip Heilman, Paul Lawrence

Abstract

Interest in managing watersheds is growing rapidly and many new watershed management groups are forming. Often these groups form because of the shared perception of an existing or impending problem. Once watershed groups form however, they can founder because watersheds integrate so many physical, biological, social and economic processes and information is so scarce that it is difficult to know where to start. A software tool called the Facilitator has been developed to help watershed groups develop a plan to address their problems without being paralyzed by the lack of information. The first steps in using the Facilitator are to define the group's objectives and determine what management alternatives are available to achieve those objectives. The next step is to estimate the effects of management on the objectives. Lastly, there is a graphical tool to rank alternatives based on the relative importance of the objectives. An advantage of using the Facilitator is the flexibility to use expert opinion at the initial stages to frame the problem and so determine what additional data to collect and which physical processes to should be modeled. Two examples show how watershed groups used the Facilitator to define alternatives that are then examined in more detail using watershed simulation models. The first example is a cropland watershed with a growing number of hog production units in central Iowa and the second is a primarily grazed watershed in central Queensland.

Keywords: decision support, simulation modeling, conservation planning

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An Examination of the Use of Information and Communication Technologies by Non-Governmental Organizations (NGOs) in the Chesapeake Bay Watershed

Michele Masucci

Abstract

The purpose of this paper is to examine non-governmental organizational (NGO) frameworks for information and communication technology use and management in relationship to integrated watershed management. A comparison of the information and communication technologies approaches and strategies among small, intermediate and large scale NGOs will be provided, as well as an examination of the inter-linkages within NGO networks across organizational scales and settings. NGO decision-making frameworks will be considered as based in a unique set of concerns related to integrated watershed management and associated water quality monitoring processes. In contrast to government sector planning entities, whose missions may also include water quality monitoring, NGOs typically address the watershed management concerns that help to round out government environmental protection and monitoring programs from the perspective of membership and advocacy coalitions. Often the emergence of advocated interests correlate and support the creation of new governmental policies, programs and entities as has been the case within the Chesapeake Bay watershed area – the example I will focus on as a case study. This different focus that NGOs have in comparison to other watershed management actors contributes to a unique beginning point for the use of information and communications technologies as well. This paper will discuss how a network of NGOs involved in the complex dialog existing among economic sectors, environmental quality interests, and public health concerns related to monitoring the Chesapeake Bay watershed and episodic crises that periodically emerge within specific locations across the watershed approach the use of information and communication technologies.

Keywords: NGO, Chesapeake Bay watershed, information and communications technologies

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Application of BASINS for Water Quality Assessment on the Mill Creek Watershed in Louisiana

Kansheng Wu, Y. Jun Xu, W.E. Kelso, D.A. Rutherford

Abstract

GIS applications are needed to understand hydrologic processes connected with water quality assessments on a watershed scale. In this study, we applied BASINS 3.0, Better Assessment Science Integrating Point and Nonpoint Sources developed by the U.S. EPA, to a lowland terrace watershed in southern Louisiana, in order to (1) segment the watershed and streams to select monitoring locations for a new research project on forest best management practice (BMP) effectiveness, (2) create a GIS framework in a form conducive to spatial analysis on a sub-watershed scale using a customized dataset, and (3) calibrate the Soil and Water Assessment Tool (SWAT) model to obtain reliable parameter ranges. Using the data sources and GIS extensions integrated in BASINS 3.0, DEM data, and the historical USGS peak flow data, we delineated sub-watersheds, identified stream segments and sub-watershed characteristics such as areas, vegetation, soil type, and other hydrologic parameters, and simulated stream discharge for a small sub-watershed. The simulated stream flow from the SWAT model, combined with user defined precipitation and air temperature data, were compared with observed peak flow data to calibrate localized hydrologic parameters that will be used in our study site. The study shows that the BASINS 3.0 system offers efficient modules for the watershed delineation, source data integration, and hydrologic modeling. Information from BASINS and SWAT can be an effective tool for researchers and water resource managers to predict potential impacts of land management practices on water quality and other hydrologic process.

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Keywords: watershed delineation, hydrologic modeling, BASINS, SWAT, Louisiana

Introduction

Water quality has been one of the major environmental issues across the country for over 30 years (Adams et al. 2000). Although efforts to control point source pollution since the 1970s have been moderately successful, with considerable expenditures of funds and effort from federal, state, and local agencies, controlling non-point source pollution (NPS) remains a challenging task. Best Management Practices (BMPs) have been introduced to prevent NPS from agricultural and forestry activities. Coincident with state and federal legislative efforts and the nation-wide implementation of BMPs, the importance of understanding the effects of land management practices on water quality has received increased attention in watershed management during the past decade (Brooks et al. 2003).

Effective watershed management requires a detailed understanding of hydrologic and biogeochemical processes within the watershed. The relationships among land use practices, agricultural activities and water quality parameters are both spatially and temporally complex. Mathematical models and geospatial analysis tools are often employed to investigate these relationships and identify management options.

BASINS 3.0, Better Assessment Science Integrating Point and Nonpoint Sources developed by the U.S. EPA, has been reported to be a powerful tool for spatial analysis at the watershed scale. The system is embedded within a geographic information system (GIS) that integrates a variety of spatial data, including land use, soil, vegetation, climate, and elevation that are calculated with Digital Elevation Models, and a set of modeling tools, such as the Soil and Water Assessment Tool (SWAT) developed by USDA Agricultural Research Service (Arnold et al. 1998). SWAT has been used to predict various impacts of land management on water quantity (e.g., Srinivasan and Arnold 1994, Muttiah and Wurbs 2002), sediment yield and nutrient loss (e.g., Luzio et al. 2002), and pesticide fate and transport (e.g., Brown and Hollis 1996) in a wide range of watershed scales from a few dozen hectares to thousands of square kilometers. In SWAT, spatial heterogeneity in soil and land use cover in a watershed are represented by hydrological response units (HRUs), which are based on groupings of physical and hydraulic parameters. The model estimates relevant hydrologic components including evaportranspiration, surface runoff, return flow, and ground water recharge at the delineated sub-watershed.

In this study, we applied BASINS 3.0 and SWAT to quantify the stream discharge in a small Louisiana watershed from 1968 to 1980. The project watershed was chosen because of the availability of USGS data and its proximity to the location of the BMP effectiveness study. The objectives of this research are to: (1) create a GIS framework in a form conducive to spatial analysis of the proposed BMP study watershed, (2) examine applicability of BASINS and SWAT for the Louisiana's lowland terrace watersheds, and (3) calibrate the SWAT model to obtain reliable parameter ranges for our study area. This paper presents the preliminary results of this GIS and modeling study.

Methods

Site description

The study site, located in the Mill Creek watershed, Allen Parish, central Louisiana, is characterized by a humid subtropical climate, with an annual average air temperature of 20.7 °C and annual precipitation 1558 mm. The watershed measures about 25 km long and 8.5 km wide (Figure 1), and represents a typical landscape of lowland terrace in the Gulf region. The 209.3 km^2 watershed (LDEQ, 2000) is composed primarily of commercial forest (94%) and agricultural (4%) lands. The average elevation of the northern part of the watershed is about 40-45 m, and the average elevation at the southern boundary is about 5-10 m. The dominant soils are Guyton silt loam (40.3% of total watershed) and Guyton-Messer complex (35.8%). Guyton series soils are characterized by a silt loam surface layer and a grayish loamy subsoil (USDA-SCS 1980) and poor soil drainage. The watershed was identified as impaired by LDEQ due to low dissolved oxygen concentrations.

The sub-watershed used for the hydrologic simulation in this study measured 4.7 km^2. It encompassed a first-order stream located about 8 km southeast from the proposed Mill Creek BMP study site (Figure 1). The site was similar in geomorphology, soils, and land use activities, and was the closest location that provided streamflow measurements.

Data sources

Spatial data sources used in this research include DEM, soil, land use, reach file, climate, and USGS stream flow data (gage station 08013350). Five USGS 24K DEM quads (3009202, 3009210, 3009211, 3009218, and 3009219) were used to create a single coverage for the entire study area. Soil data were obtained from the State Soil Geographic Database (STATSGO) (http://www.ftw.nrcs.usda.gov/stat_data.html). Land use cover data were obtained from USGS Land Use and Land Cover (LULC) Database (http://edc.usgs.gov/products/landcover/lulc.html), that was based primarily on manual interpretation of 1970s and 1980s aerial photography.

Climate data, including daily precipitation and daily maximum and minimum air temperature were obtained from the Southern Regional Climate Center. Precipitation data were obtained from four nearby weather stations, while air temperature data were gathered from two stations (Figure 1). Daily wind speed, humidity, and solar radiation data used for the hydrologic modeling were provided by SWAT.

Peak flow data from 1968 to 1980 from the USGS gauge station 08013350 were used to evaluate the hydrologic model. Unfortunately, no daily streamflow data was available at this station, and SWAT did not provide peak flow estimates. Despite the discrepancy between the USGS peak flow and the SWAT-simulated daily average flow, it is acceptable to compare these two datasets to obtain general ranges of SWAT model parameters, which will be used as initial values for our future BMP effectiveness study site.

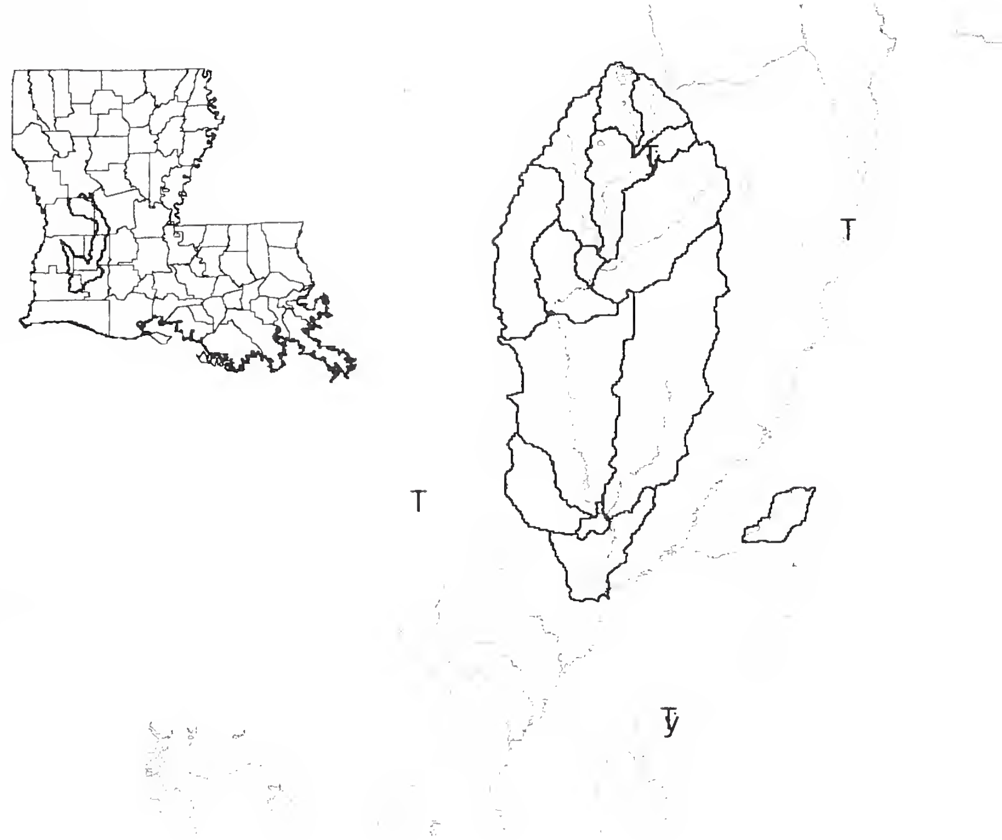

Figure 1. Delineated sub-watersheds and weather station locations

Data analysis

The Universal Transverse Mercator (UTM) projection was used for all spatial coverages in this study. Sub-watersheds were delineated with the Automatical Procedure provided by BASINS 3.0, which was used to delineate the BMP study watershed and the USGS 08013350 gauge station watershed with 24K DEM data.

The SCS Curve Number (CN) method was chosen for hydrologic simulation. A default CN2 of 35-95 was defined for the initial simulation. To calibrate CN2 the following four scenarios were used: Scenario 0: default from SWAT, 35-98; Scenario 1: 60-73; Scenario 2, an increase of 10% based on scenario 1; and Scenario 3: a decrease 10% based on scenario 1. The simulated discharge was compared with the USGS peak flow data. Regression coefficients and root mean square errors from a linear regression model were used to evaluate the modeling results. All statistical analyses were done with the SAS statistical software package (SAS 1999).

Results and Discussion

Watershed delineation and characteristics

BASINS' Automated Delineation tool delineated 14 sub-watersheds in the Mill Creek watershed (Figure 1, Table 1) with a total area of 200.4 km^2, which was about 9 km^2 larger than that (209.3 km^2) reported by the Louisiana Department of Environmental Quality (LDEQ 2000). The small watershed that was used for SWAT hydrologic modeling was also delineated with the BASINS' Automated Delineation tool. The delineated area was 4.9 km^2, which was slightly larger than the USGS-defined drainage area (4.7 km^2). The difference in the delineated areas in both cases was small (4.25% and 5.4%, respectively), suggesting that BASINS is a reliable tool for digital watershed delineation.

BASINS' automated watershed delineation provided not only sub-watershed boundaries and area, but also basic information on watershed characteristics, such as slope, stream reach length, area percentages of land use and soil types, and hydrologic response unit (HRUs). However, the size and topographic relief of a watershed appeared to affect accuracy of delineation.

Table 1. Relevant characteristics of delineated sub-watersheds in the Mill Creek watershed.

Su	Ar	Str	Sub(%)	El (m)	Lum	Apm(%)	Som*	Ars(%)	Hrus
1	6.1	5144.9	1.31	45	FRSE	30.6	LA112	79.5	12
2	5.3	4859.7	0.65	43	FRSE	60.8	LA123	74.7	6
3	12.3	9881.6	1.22	45	FRSE	85.8	LA112	57.8	8
4	14.1	9747.4	0.82	34	FRSE	85.8	LA123	65.4	7
5	1.9	2863.7	0.46	30	FEST	78.4	LA186	63.3	4
6	26.6	13747.1	0.32	34	FEST	79.8	LA186	88.8	5
7	10.5	5374.7	0.51	31	FRSE	76.5	LA123	84.1	4
8	16.7	9818.8	1.14	35	FRSE	92.3	LA112	69.3	5
9	38.8	14560.3	0.65	26	FRSE	76.6	LA123	53.2	9
10	10.2	6719.9	0.55	27	FRSE	89.1	LA123	90.4	4
11	1.4	2376.6	1.11	20	FRSE	67.8	LA114	70.8	3
12	45.3	18222.1	0.42	30	FRSE	50.3	LA186	76.2	8
13	4.9	4798.9	0.32	27	FEST	85.9	LA123	72.5	3
14	11.2	7517.4	1.02	20	FRSE	54.4	LA123	57.6	8

Notes:
Su: Sub-watershed
Ar: Areas in km^2
Str: Stream reach (m)
Sub: Sub-watershed slope (%)
El: Elevation (m)
Lum: Land use of maximum area in sub-watershed
Apm: Area percentage of maximum land use
Som*: Soil type of maximum areas in sub-watershed, see the STATSGO for detailed soil type
Ars: Area percentage of maximum soil type (%)
Hrus: Hydrologic response units
FRSE: Evergreen Forest Land
FRST: Mixed Forest Land

When a watershed is small and flat, the USGS reach file may not be appropriate for automatic delineation due to a possible large difference between DEM and reach data.

Simulation and calibration

Four scenarios with varying CN2 values were simulated. Compared to the other three scenarios, Scenario 2 with 60-73 achieved the highest R^2 (0.71) and smallest root MSE (55.5). Table 2 summarizes parameters used in the Scenario 2 simulation. A comparison of the simulated daily average streamflow with the measured peak flow showed a similar temporal pattern over the period from 1968 to 1980 (Figure 2).

For localized and site-specific application, calibrating SWAT input parameters and performance of global changes with observed data are necessary (Luzio et al. 2002). Although SWAT has 27 input parameters that can be user-adjusted, the sensitivity of each parameter is different for a specific output. When water balance components are considered, adjustment of runoff curve numbers and revap coefficients would give good correspondence (Srinivasan et al. 1998). Manoj et al. (2003) reported that curve number (CN), evaporation compensation factor (ESCO), groundwater delay time (DELAY), plant uptake factor (EPCO), revap coefficient (REVAP), and soils available water capacity (AWC) were most sensitive to stream flow. In our study, both the R^2 and root MSE changed dramatically when CN was changed from its default range of 35-98 to a more preferable range 60-73. However, an increase or decrease outside this CN range did not result in substantial changes in the model.

Table 2. Parameterization in the hydrologic modeling with SWAT.

Parameters *	Symbol	Ranges
Curve number	CN2	60-73
Groundwater revap coefficient	REVAP	0.02-0.2
Evaporation compensation factor	ESCO	0-1
Plant uptake factor	EPCO	0.01-1
Groundwater delay time(day)	DELAY	0-500

* Refer to SWAT user manual (Neitsch et al 2002).

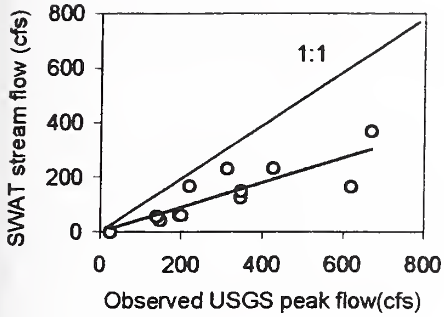

Figure 2. Comparison between simulated daily average stream flow and USGS-observed peak flow during 1968 to 1980.

Use of the SWAT model in this study required the assumption that land use and land cover did not change during the simulation period. Because land use in rural Louisiana has not changed substantially in the last few decades, this assumption is probably valid for the modeled watersheds in rural Louisiana; land use and land cover data were collected around 1978-1980. However, integration of dynamic land use data into the SWAT model could provide more reliable information, especially in those watersheds undergoing significant changes in land management practices.

Simulated hydrologic components

The simulated water balance components for 08013350 watershed (number 13 in Table 1 and Figure 1), such as precipitation, actual evapotranspiration (ET), and discharge are shown in Figure 3. Actual ET showed a clear seasonal fluctuation with the highest value (147.5 mm) in July and the lowest in December (14.9 mm), while precipitation was relatively evenly distributed over the year with a monthly range between 102 and 180 mm. Similar to actual ET, discharge showed a distinctive temporal trend, with the highest value in May (107.8 mm) and the lowest value in August (18.8 mm). Based on precipitation and temperature data from nearby weather stations, annual average precipitation for the study period in this watershed was 1676 mm, slightly higher than the longer-term average. Annual actual ET and annual discharge were estimated to be 868 mm and 784 mm, respectively. The 24 mm difference between the input (precipitation) and output (evapotranspirtion + discharge) may indicate a small groundwater recharge in the area.

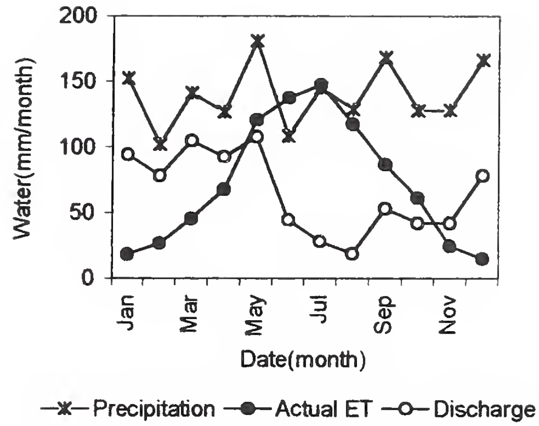

Figure 3. Seasonal trend of simulated hydrologic components.

Srinivasan et al. (1998) reported that the SWAT model may overestimate monthly streamflow during spring and summer when high spatial variability of precipitation is present, or underestimate streamflow when extreme precipitation events occur (Rosenthal et al 1995). In this study, we found a significant seasonal pattern of monthly streamflow (Figure 3) that decreased with increasing actual ET. The monthly discharge dropped rapidly in June and remained low throughout November, mainly due to the higher forest transpiration rates. The monthly average actual ET and discharge were both high in May (120.7 mm, 107.8 mm) and both low in November (24.5 mm, 41.8 mm), indicating that removal of the forests would have the greatest impacts on site hydrology during this period of time.

Summary

A GIS framework for our future BMP effectiveness study has been created with BASINS 3.0. This framework integrates all critical datasets for spatial analyses of the relationships among soil, land use, and hydrology. BASINS with the embedded hydrologic model SWAT has proven to be an effective tool for watershed delineation in the studied lowland terrace of Louisiana. In addition to watershed delineation, it provides basic information on watershed characteristics, such as area, slope, stream reach length, land use and soil types, and the corresponding percentages. However, the size and relief of a watershed appeared to affect accuracy of delineation, and cautions should be exercised when delineating a watershed that is small and with a very flat topography. The SWAT model and the calibration of CN2 have provided reasonable estimates of the hydrologic components for the studied watershed, the initial parameterization of which will be useful for the future application in our prospective BMP effectiveness assessment.

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The San Pedro River Spatial Data Archive: A Database Browser for Community-Based Environmental Protection

William G. Kepner, Darius J. Semmens, Daniel T. Heggem, Edward J. Evanson, Curtis M. Edmonds, Soren N. Scott

Abstract

It is currently possible to measure landscape change over large areas and determine trends in ecological and hydrological condition using advanced space-based technologies accompanied by geospatial data. Specifically, this process is being tested in a community-based watershed in southeast Arizona and northeast Sonora, Mexico using a system of landscape pattern measurements derived from satellite remote sensing, spatial statistics, process modeling, and geographic information systems technology. These technologies provide the basis for developing landscape composition and pattern indicators as sensitive measures of large-scale environmental change and thus may provide an effective and economical method for evaluating watershed condition related to disturbance from human and natural stresses. This project utilizes spatial data from a number of sources. The information has been modified to fit the community project area and assembled into a database browser with search functionality. We have produced all spatial data into a one-stop, easy-access product that will be useful to all others who utilize geographic information systems and could benefit from the information in regard to research, natural resource management, human-use planning, and policy development. The San Pedro Data Browser is currently available on-line via the EPA server (http://www.epa.gov/nerlesd1/land-sci/san-pedro.htm) and distributed as CD-ROMs. The purpose of the database is to disseminate available data that could be used by the stakeholder community to address environmental issues and improve environmental decision-making.

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Keywords: San Pedro River, geographic information systems, remote sensing, geospatial data, community assessment

Indicators for Assessing Watershed Conditions

B. McQuaid, S. Aschmann, C. Seybold, G. Spiller

Abstract

The USDA Natural Resources Conservation Service (NRCS) is designing the *Indicators of Watershed Conditions* web site. The purpose of the site is to provide field staffs with screening level tools aimed at watershed scale assessments. The Indicators tool component is the focal point of the site and lists and describes indicators for issues commonly addressed by natural resource managers. Another feature of the web site is the training link that provides several modules on indicators and watershed functions and processes. Field testing of the indicators in addition to continued refinement and development of the site is planned.

Keywords: watershed assessment, watershed conditions, watershed indicators

Introduction

A web site is being developed by USDA-NRCS to provide field staffs with screening level tools aimed at watershed scale assessments. The site's framework is based on a variety of issues commonly addressed by resource managers. It also provides an introduction and guidance for using indicators to address resource issues in watersheds and other large planning areas. The major component of the site is the indicator selection tool, but training, references and contacts links are also provided.
The training link provides background material on watershed functions and processes and how indicators are used to evaluate these. The references link lists references pertaining to watersheds watershed assessment, and indicators. The contacts link aids the user in selecting and evaluating indicators.

The following discussion provides an overview of the *Indicators for Assessing Watershed Conditions* web site framework including the resource issues, indicators tool, watershed assessment indicators, and the training site components.

Indicators tool and resource issues

Figure 1 depicts the framework of the Indicators tool web site component. The Indicators tool component lists and describes indicators for resource issues commonly addressed by planners. The six resource issues in Table 1 provide the framework for the web site. These issues were selected because they define a majority of watershed issues addressed by natural resource managers. To use the web site a user selects a resource issue of concern for a watershed. Under each resource issue are a selection of indicators that could be used to address that issue. The user may then elect to use this indicator or modify it to suit the attributes of the watershed.

Watershed assessment indicators

A total of 24 indicators are provided in the web site. Most of the indicators are screening tool type indicators that are designed to target sub-areas within a watershed. The design team created many of the indicators and a few are links to other indicator web sites. Some indicators pertain to more than one resource issue. For example, the indicator freshwater consumption is listed under five resource issues-quality of life, social and economic well being, habitat, water supply and availability, and sustainable food and fiber. Each indicator also has a discussion of what the indicator can tell you, the time requirement to apply the indicator, methodology, and case examples of how to apply the indicator in a watershed.

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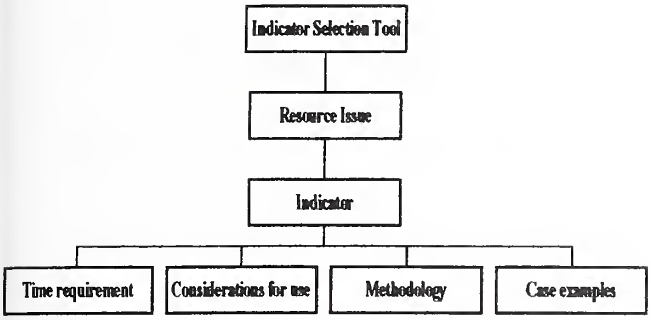

Figure 1. Flow chart depicting framework of the *Indicators for Watershed Conditions* web site.

Field testing of the indicators in addition to continued refinement and development of the site is planned. The web site is currently located under the Watershed Science Institute selection located at the USDA-NRCS homepage http://www.nrcs.usda.gov.

Training modules

The site also contains a link to ten training modules. These modules are being developed to provide the user with an overview of watershed process and functions and indicators. Topics include indicators and indicator selection, watershed processes and functions, relationship of indicators to the planning process, and strategies for using indicators within USDA NRCS, sampling and aggregation for indicator measurements, selection of indicators related to soil, air, animal/plant, and human resources.

Acknowledgments

The authors wish to thank the following design team members for their contributions to the web site: Hank Henry, Bruce Newton, Lyn Townsend, Roel Vining, Dennis Carmen, Danny Goodwin, Tom Noonan, and Carolyn Adams.

Table 1. Framework of the Indicators of Watershed Conditions web site.

Resource Issue	**Indicator**
Quality of Life	Cropland Conversion
	Flooding Potential Ratio
	Freshwater Consumption
	Percent Open Water
	Flooding
	River and Stream Density
	Index of Watershed Health Indicators
Pollutants and Contaminants	Acid Precipitation
	Clean Air Act Criteria
	Cropland Conversion
	Greenhouse Gases
	Odor
	Phosphorus Source to Sink Ratio
	Index of Watershed Health Indicators
	Wetland Change
	U.S. EPA Contaminated Sites
Social and Economic Well-Being	Average Farm Size
	Flooding Potential Ratio
	Freshwater Consumption
	Gross Farm Sales
	Household Income
	Median Non-Metropolitan
	Percent Open Water
	River and Stream Density
Sustainable Food and	Acid Precipitation
	Cropland Conversion
	Cropland Soil Quality Impairment Rating
	Flooding Potential Ratio
	Freshwater Consumption
	HEL/COVER/No-till Ratio
	Percent Open Water
	River and Stream Density
	Survey of Accelerated Erosion Features
Water Supply and	Freshwater Consumption
	Percent Open Water
	River and Stream Density
	Saltwater Intrusion
	Subsidence
	Fish and Mussels at Risk in U.S.
	Flooding Potential Ratio
Habitat	Habitat Quality
	Percent Open Water
	River and Stream Density
	Threatened and Endangered Species
	Freshwater Consumption
	Wetland Change

Using Science to Support Management Decisions in Waquoit Bay, MA

Victor B. Serveiss, Jennifer L. Bowen, David Dow, Ivan Valiela, Leela Rao

Abstract

Watershed ecological risk assessment principles were was used to organize, analyze and present scientific information to help protect the ecological resources of the Waquoit Bay watershed, in Massachusetts. Through a series of meetings with the general public and local and state managers an environmental management goals and objectives were established for the watershed. An interdisciplinary and interagency workgroup identified manmade stressors, exposure pathways, dose/response relationships, and assessment endpoints. A conceptual model was used to better clarify the pathways by which stressors such as nutrient enrichment, physical alteration of habitat, altered freshwater flow, and toxic chemicals impacted several assessment endpoints. A comparative risk analysis and an evaluation of the impacts of stressors identified nutrient enrichment as the major stressor within the watershed. This justified focusing on the assessment endpoints most impacted by nitrogen loading, eelgrass cover and scallop abundance. A nitrogen loading and an estuarine loading model were developed to link land use changes to changes in water quality. By comparing increases in nitrogen loads to losses in the area of eelgrass cover over the last 60 years, it appears that eelgrass disappears once nitrogen loads reach 20 kg ha-1 yr-1. Both the increase in nitrogen load and the decrease in eelgrass can be correlated to decreases in the annual harvest of scallops. Developed models provide the opportunity for managers to assess a variety of options to reduce nitrogen loads to their estuaries and to achieve the loads that could allow the return of eelgrass to the target area.

Keywords: watershed ecological risk assessment, multiple stressors, eutrophication, watershed management

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Experimental Watersheds

The USDA ARS Little Washita River Experimental Watershed

Michael W. Van Liew, Patrick J. Starks, John A. Daniel, Jean L. Steiner

Abstract

The Little Washita River Experimental Watershed (LWREW) located in the Red River Basin of the Southern Great Plains (SGP) near Chickasha, OK is one of USDA Agricultural Research Service's largest and best-instrumented watersheds. The LWREW drains an area of 610 square kilometers and is characterized by variations in topography, soils, and land use. The watershed is located within a steep precipitation gradient of the SGP, making it ideal for conducting research on regional mass and energy fluxes at the earth's surface. The LWREW has been operated and maintained by ARS since the 1960s and provides a wealth of field and remotely sensed data. An automated meteorological network, consisting of 42 stations, spaced at 5 km intervals, provides 5-minute measurements of rainfall, air temperature, relative humidity, solar radiation, and soil temperature. These variables as well as wind speed, wind direction, and air pressure are also monitored at three of the University of Oklahoma/Oklahoma State University Mesonet sites. Hourly measurements of soil matric potential, heat flux, and soil temperature are made at 12 soil heat and water measurement stations. Ten stream gages are located in the watershed, with three on the main stem of the river. Current research on the LWREW is designed to address problems that relate to the integrated effects of agricultural land use, land management and climate variability on surface and ground water resources and the use of remote sensing techniques to monitor and predict root zone water content and availability at regional scales. Research from these programs is expected to promote the sustainable use of water resources and to help reduce the risk associated with fluctuations in climate. Current research on the LWREW also facilitates multi-stakeholder discussions for UNESCO's HELP (Hydrology for the Environment, Life and Policy) in the Red-Arkansas River Basin.

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Keywords: experimental watershed, instrumentation, streamflow, remote sensing, ground water, watershed management

Introduction

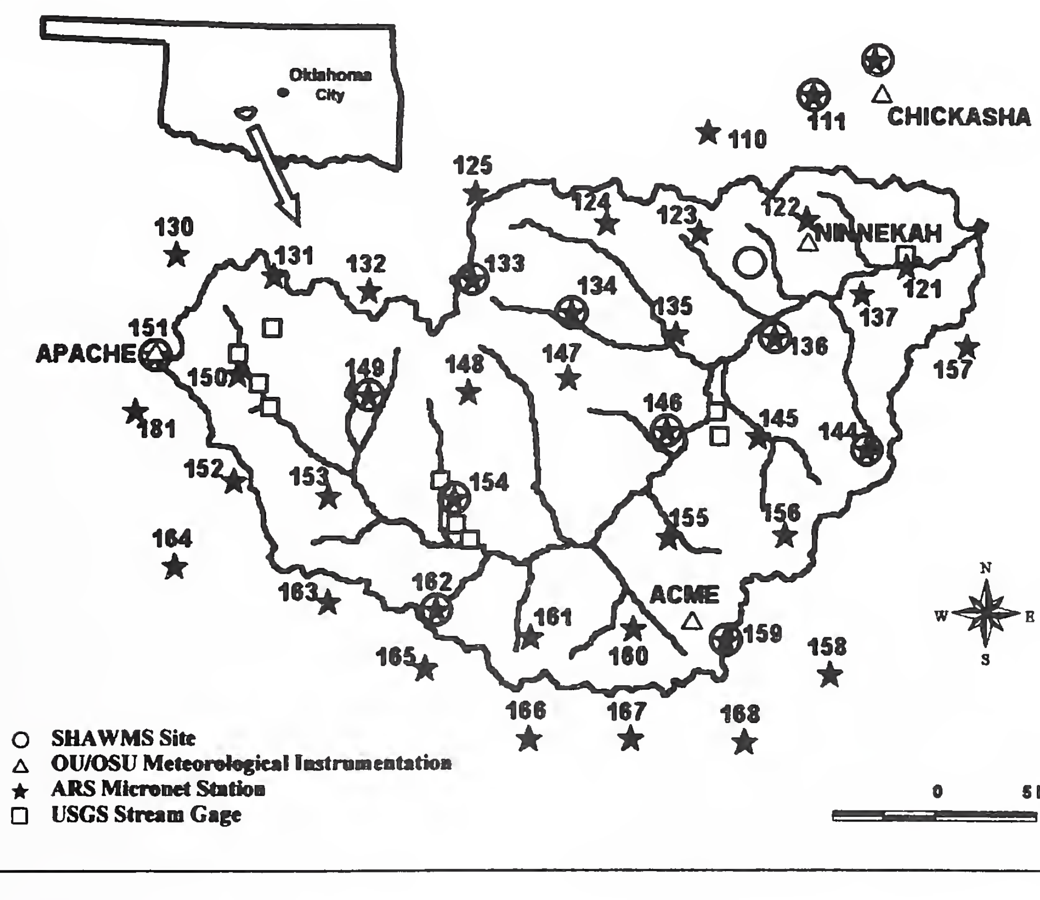
In September 1959, a report was presented to the United States Congress titled "Facility Need - Soil and Water Conservation Research," popularly referred to as Senate Document 59. As a result of this document, the Southern Great Plains Research Watershed laboratory was established in 1961. The laboratory was given the mission to assess the overall effects of flood control and watershed protection programs implemented on the Washita River Basin (WRB), which was subject to severe flooding and soil erosion. The WRB study area included what is now known as the Little Washita River Experimental Watershed (LWREW). Today the LWREW is the U.S. Department of Agriculture (USDA) Agricultural Research Service's (ARS) largest experimental watershed, and one of the best-instrumented watersheds in the world. It has been used as a Super Site in NASA Shuttle Missions and is playing a key role in both remote sensing and climate change research. In addition to a substantial volume of hydrologic data that have been collected on the watershed, the LWREW has been the site for numerous experiments related to terrestrial climate, hydrologic processes, and agricultural activities.

Test Watershed

The LWREW is located about 80 kilometers southwest of Oklahoma City, OK, and drains an area of 610 km^2 (Fig. 1). The climate in the region is sub-humid to semi-arid, with an average annual precipitation during the past 30 years of about 830 mm. The watershed is located within a steep precipitation gradient of the Southern Great Plains, making it ideal for conducting research on regional mass and energy fluxes at the earth's surface. Nearly half of the annual precipitation occurs during the months of May, June, September and October, and most of the large floods take place in the spring and fall. Mean annual temperature for the watershed is 16 degrees C. Topography of the watershed is characterized by gently to moderately rolling hills, with average elevation of about 400 m, a maximum relief of about 183 m, and a mean channel slope of about 7 m/km. Average annual runoff from the watershed is 1.2 cubic meters per second (cms) or about 62 mm.

There are 64 soil series defined for the watershed and 162 soil phases mapped within these series (Allen and Naney 1991; http://grl.ars.usda.gov/lwashita.html). Soil textures on the watershed consist of 29% silt loam, 17% loam, 41% fine sandy loam, and 13% sandy loam. Bedrock exposed in the watershed consists of Permian-age sedimentary rocks. The oldest formation in the watershed, the Chickasha Formation (sandstones, siltstones, and shales), comprises 5% of the watershed. The Dog Creek and Blaine Formations (shales interbedded with sandstones--8% of the watershed) overlie the Chickasha, which in turn are overlain by the Marlow (sandy shale--14%), Rush Springs (fine-grained sandstone and siltstone--45%), and Cloud Chief formations (irregular gypsum--17%). Although these formations dip gently to the southwest, surface runoff is generally toward the east. Alluvial deposits cover the bedrock valleys throughout the watershed and comprise approximately 11% of the total watershed area (Allen and Naney 1991).

Figure 1. Location of Instrumentation Sites on the Little Washita River Experimental Watershed.



For the most part the water bearing aquifers underlying the watershed are effluent in nature and contribute to Little Washita River streamflow. However, seepage from the river has been observed to occur along portions of the channel in the central region of the watershed.

Land use surveys have been made periodically on the LWREW since the early 1960s. Conventional surveys from aerial photographs and point sampling indicate that few changes in land use have occurred on the watershed during the past 40 years. Currently 66% of the watershed is in rangeland, 18% is under cultivation (primarily wheat and alfalfa), and the remaining 16% is in miscellaneous use (timber, farmsteads, abandoned oil sites, and urban). The watershed has numerous farm ponds located primarily in the lower portions of the watershed, and 45 Soil Conservation Service (SCS) flood-retarding structures constructed from 1969 to 1982 (Allen and Naney 1991). These structures delay and reduce peak flow, but do not significantly alter total runoff volume over a year. Irrigation systems on the LWREW are minimal, occurring primarily along the flood plain.

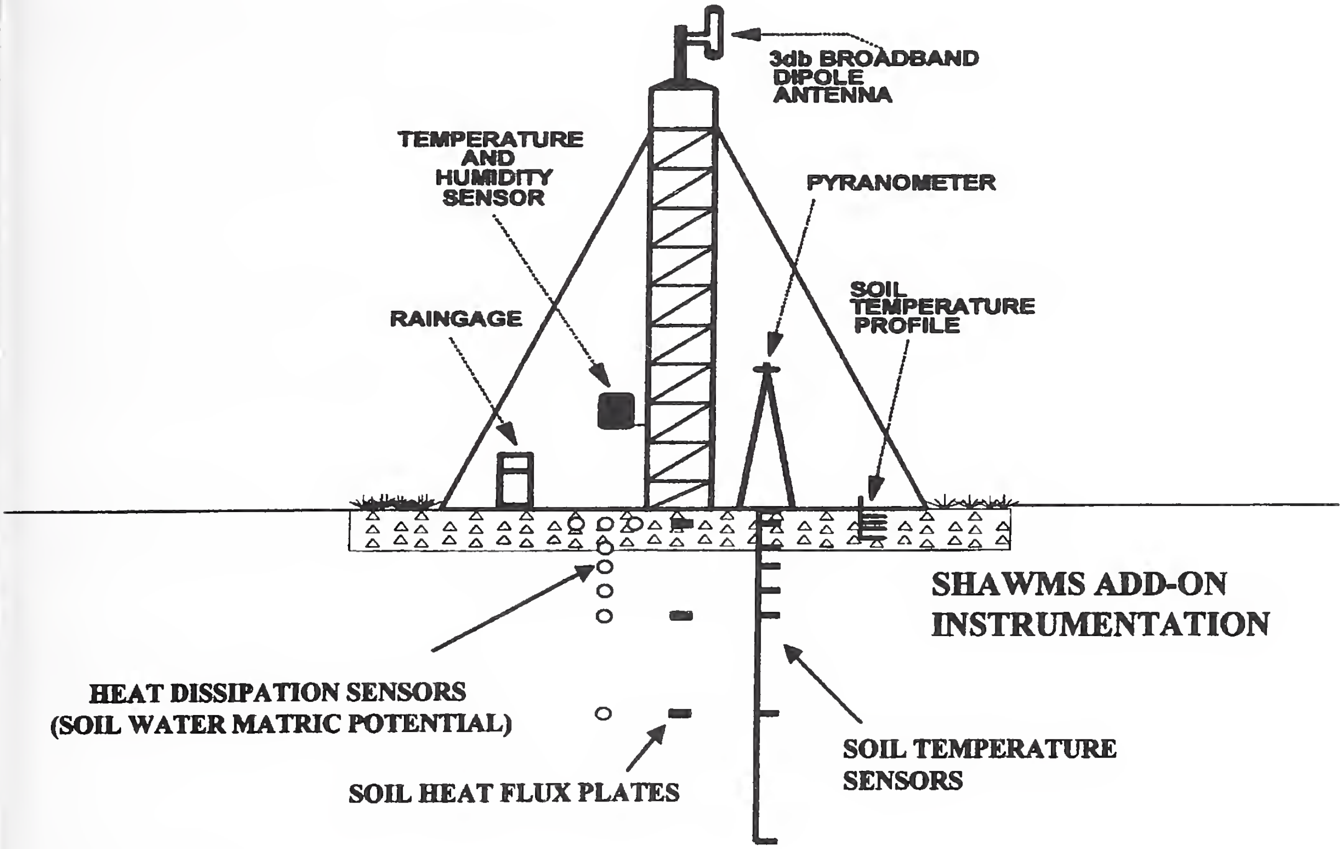
Watershed Instrumentation

In 1961 the USDA ARS began data collection on the LWREW to determine the downstream hydrologic impacts of SCS flood-retarding structures. At that time a network of 42 raingages was installed in or nearby the watershed, along with 25 ground water observation wells. In 1963 a stream gage (drainage area of 538 km^2) was installed and operated until 1985 to monitor discharge and sediment concentration. In 1978 a second stream gage was installed in the upper part of the watershed (drainage area of 160 km^2) and operated until 1985 to monitor discharge and sediment concentration. Sediment transport data for the two stream gaging sites were obtained by taking several suspended samples during each runoff event with a U.S. D-49 suspended sediment sampler, which was cable suspended from a stream gaging reel and bridge crane. Samples were also collected at both of the stream gaging sites with a U.S. PS-66A automatic, pumping type suspended sediment sampler. Pumped samples were obtained at 30-minute intervals prior to flow peaks and at 1 to 5-hour intervals after peak flows (Allen and Naney, 1991). Hydrologic data collected on the LWREW from 1961 to 1985 are archived at the USDA ARS Grazinglands Research Laboratory (GRL) in El Reno, OK and may be accessed at: http://grl.ars.usda.gov/lwashita.html. Currently the U.S. Geological Survey maintains ten streamgaging stations within the LWREW, with associated drainage areas ranging in size from 3 to 610 km^2 (Fig. 1). Streamflow data for the watershed are achieved at the USGS District office in Oklahoma City and may be accessed at: http://waterdata.usgs.gov/nwis/sw.

Today a network of 42 meteorological stations (referred to as the Micronet) is maintained by staff at the USDA ARS GRL (Fig. 1). Each Micronet station, located on a 5 km by 5 km grid, provides 5-min values of rainfall, incoming solar radiation, air temperature, relative humidity, and soil temperature at three depths. Data from the Micronet are achieved at the Oklahoma Climatological Survey in Norman, OK and are available at: http://www2.mesonet.ou.edu/ars. Meteorological data are also collected at three additional stations, referred to as Apache, Ninnekah, and Acme, which are a part of the Oklahoma Mesonet (www.mesonet.ou.edu) (Fig. 1). The Mesonet stations represent a network of continuous recording gages that provide enhanced meteorological coverage across the State of Oklahoma. Barometric pressure, wind speed, and wind direction are monitored at the Mesonet stations, in addition to the meteorological variables collected at the Micronet sites.

A Soil Heat and Water Measurement Station (SHAWMS) is co-located at 12 of the Micronet stations (Fig. 1). Each SHAWMS provides an hourly profile measurement of soil water matric potential, and half-hourly measurements of soil temperature and soil heat flux. Matric potential is measured using a Campbell Scientific Model 229-L soil matric potential sensor. Eight of these sensors are located at each SHAWMS, with three at 5 cm, and one each at 10, 15, 20, 25, and 60 cm. Soil water release curves have been determined at each site, enabling the conversion of matric potential into an estimate of volumetric water content. Soil temperature is measured at 2.5, 5, 10, 15, 20, 25, 60 and 100 cm below the soil surface. Soil heat flux is measured at 5, 25, and 60 cm (Fig 2). Data collected from the SHAWMS sites are achieved at the USDA ARS GRL in El Reno.

Figure 2. Schematic Diagram of Micronet Station and SHAWMS Site.



Research Emphasis

The LWREW has been the site for a number of research projects since the 1960s. From 1961 to 1978 conditions were monitored on the watershed to determine the impacts of flood retarding structures (FRSs) on reducing damages caused by flooding and erosion from agricultural land. In this investigation streamflow, sediment, water quality, stage volume, and channel geometry data were monitored at selected FRSs within the LWREW (Schoof et al., 1987; USDA, 1983). From 1978 to 1985 the LWREW was one of seven watersheds chosen across the nation for the Model Implementation Project (MIP), which was jointly sponsored and administered by the USDA and U.S. Environmental Protection Agency (EPA) (Allen and Naney 1991). The primary objective of the MIP was to demonstrate the effects of intensive land conservation treatments on water quality in watersheds that are larger than about 65 km^2. During the MIP study, ground water levels, streamflow, sediment, nutrients, and selected chemical constituents were collected from eleven area source watersheds within the LWREW (Allen and Naney, 1991).

In recent years remotely sensed hydrologic data and meteorological data have been monitored on the Little Washita River Experimental Watershed, particularly in conjunction with intensive field campaigns such as WASHITA 92, WASHITA 94, SGP97, and SGP99, which were cooperative experiments conducted by USDA, National Aeronautical and Space Administration (NASA), and other agencies and universities. Results of these experiments may be viewed at the USDA ARS GRL website (http://grl.ars.usda.gov/lwashita.html). Low and medium altitude aircraft flights over the LWREW were coordinated with ground monitoring and, on two occasions, with Space Shuttle Endeavor experiments. The research emphasis of those experiments focused on the estimation of soil moisture and evaporative fluxes (Jackson and Schiebe 1993, Jackson et al. 1999, Mohr et al. 2000, Starks and Humes 1996) using technology that may be installed on future natural resources satellites. The LWREW is also a study site for the Global Energy and Water Cycle Experiment, an international cooperative effort to model global water and energy fluxes and improve predictions of regional impacts of climate change (www.gewex.org). In addition, research efforts on the LWREW serve to facilitate multi-stakeholder discussions for UNESCO's Hydrology for the Environment, Life and Policy (HELP) in the Red-Arkansas River Basin (www.unesco.org/water/ihp/help).

Current research on the LWREW by ARS scientists at the Grazinglands Research Laboratory is designed to address problems of national concern that relate to issues of water supply, management, and conservation in agricultural environments and semi-arid to mesic climatic conditions in the Southern Great Plains. The focus of the research is to better understand the role and impact of the integrated effects of land use, management, and climate variations on regional water supplies. Specific research objectives include the following:

1) to quantify the integrated effects of agricultural land use, land management and climatic variations on regional surface water supplies as a basis for development of strategies and methodologies to better meet the water quality and quantity needs of downstream users (Van Liew and Garbrecht 2003).

2) to determine the integrated effects of agricultural land use, land management and variable climate on infiltration, ground water recharge and return flows to provide information on regional aquifer sustainability and for ground water management.

3) to integrate remote sensing estimates of surface soil water content with other spatial data sets to monitor and predict root zone soil water content and availability at regional scales to more effectively manage water resources (Heathman et al. 2003, Starks and Jackson 2002, Starks et al. 2003a, Starks et al. 2003b).

The Little Washita River Experimental Watershed in Southwestern Oklahoma has been the site of intensive hydrologic research during the past four decades. As a world-class outdoor laboratory, it will continue to serve as an invaluable experimental watershed for investigating the exchange of water and energy to, within, and from managed agricultural ecosystems as defined by the interactions of climate, land surface hydrology, and agricultural activities.

Acknowledgments

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Hydrologic Instrumentation at the USDA-ARS Grassland, Soil and Water Research Laboratory, Riesel, TX

R. Daren Harmel, Clarence W. Richardson, Kevin W. King, Jeff G. Arnold

Abstract

The USDA-ARS Grassland, Soil and Water Research Laboratory watershed facility near Riesel, TX, is one of the most intensively monitored hydrological research sites in the country. The 340 ha research site is currently divided into sub-watersheds ranging from 0.1 to 125 ha under pasture and cropland management. Currently in operation are 18 runoff stations, 15 rain gauges, a weather station, a lateral flow station, and 7 shallow groundwater wells. Data from these stations are stored on dataloggers, downloaded with radio telemetry, and placed on the internet. Nutrient and sediment data are also collected for each runoff event with automated sampling equipment.

Keywords: hydrologic instrumentation, radio telemetry

Introduction

The objective of this paper is to describe the equipment and techniques used to collect and manage data at the USDA-ARS Grassland, Soil and Water Research Laboratory watershed facility near Riesel, TX. Rainfall, runoff, and erosion data for the site date from the late 1930s when the laboratory was established to evaluate the hydrologic response from watersheds influenced by various agricultural practices in the Texas Blackland Prairie.

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In the mid 1930s, the Soil Conservation Service (SCS), now the Natural Resources Conservation Service (NRCS), realized a need to understand and analyze hydrologic data from agricultural fields and watersheds. Based on this need, the Hydrologic Division of the SCS was established and a number of experimental watersheds were formed across the US. The primary functions of the facilities were to collect hydrologic data (precipitation, percolation, evaporation, runoff, etc.) and evaluate the hydrologic response from watersheds influenced by various agricultural land management practices (USDA-SCS 1942). One of those watersheds, originally called the Blackland Experimental Watershed, was established in 1937 in the heart of the Blackland Prairie near Riesel, TX, on the 2372 ha Brushy Creek watershed (Figure 1).

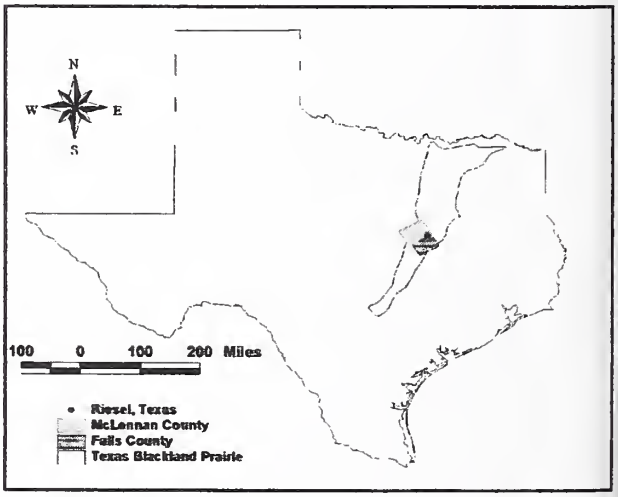

Figure 1. The USDA-ARS Grassland, Soil and Water Research Laboratory watersheds near Riesel, TX.

The experimental watershed facility later became part of the USDA-ARS Grassland, Soil and Water

Research Laboratory with headquarters in Temple, TX. Since 1937 hydrologic data have been collected throughout the Riesel watersheds, and during the height of activity, 35 runoff sites and 35 recording rain gauges were in operation. These data sets have been used for numerous purposes such as water quality studies, farming practice evaluations, and model development and evaluation. The long hydrologic records (in excess of 60 years on some watersheds) make the data particularly valuable for studies designed to identify trends or changes caused by climate change or other factors.

Site Description

The 4.45 million ha Blackland Prairie in Texas is a region of fertile agricultural land extending from San Antonio north to the Red River. Houston Black clay (fine, smectitic, thermic, udic Haplustert), which exhibits a strong shrink/swell potential, dominates the region. The slopes generally range from 1-3 % and are classified as gently rolling. Present day agricultural land use in the region consists of cattle production on pasture and rangeland, and corn, wheat, grain sorghum, and oat production under a wide range of tillage and management operations.

Hydrologic Instrumentation

Currently in operation at Riesel are 18 runoff stations, 15 rain gauges, a weather station, a lateral flow station, and 7 shallow groundwater wells. Data from these stations are stored on dataloggers, downloaded daily with radio telemetry, and placed on the internet with annual updates. Nutrient and sediment data are also collected for each runoff event with automated sampling equipment.

The USDA-ARS Grassland, Soil and Water Research Laboratory, Temple, TX website (http://arsserv0.tamu.edu/hydata.htm.) contains the data referred to in this paper.

Runoff stations

Collection of runoff data began at the Riesel site in 1938. Ten of the 18 runoff stations currently in operation are located at the outlet of small, single landuse watersheds (1.2 to 8.4 ha) to measure "edge of field" processes (Figure 2). Four of the stations are located at the outlet of 0.1 ha plots. Four of the stations are located at the outlet of larger downstream watersheds (17.1 to 125.1 ha) with mixed land uses to evaluate integrated processes.

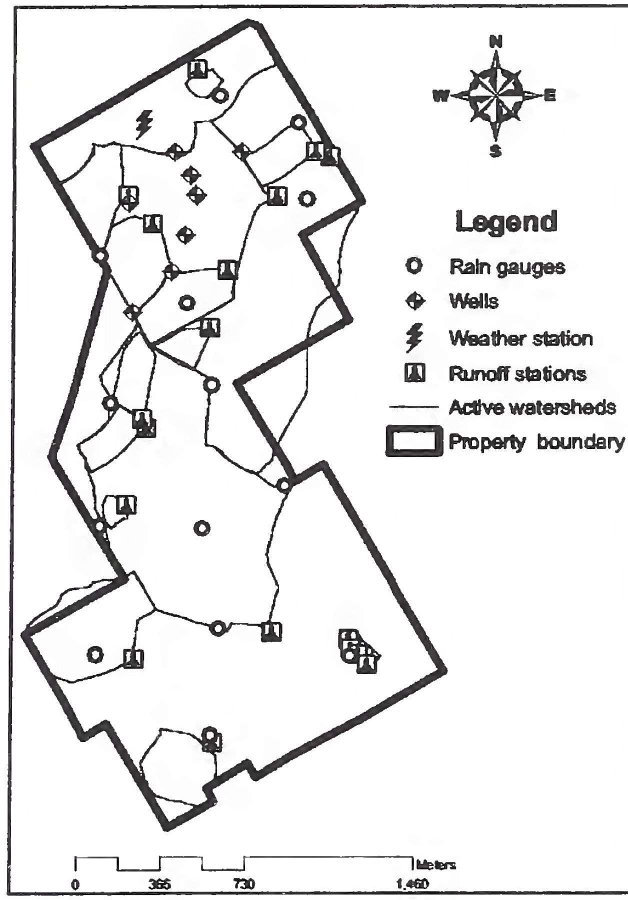

Figure 2. Hydrologic instrumentation network.

Currently, each of these runoff structures is instrumented with three flow level recording devices: 1) a KPSI pressure transducer (Keller Pressure Systems, Inc., Hampton, VA) connected to Campbell Scientific datalogger (Campbell Scientific, Inc., Logan, UT); 2) a float gauge with chart recorder; and 3) an ISCO automated sampler with bubbler level recorder (ISCO, Inc., Lincoln, NE). ISCO bubblers are used as the primary depth measurement devices, and the transducers and float gauges serve as back up devices. Discharge measurements are made by continuously recording flow levels in a stilling well located in each calibrated flume or weir structure (Figure 3). Flow depth data are converted to flow rate with established stage discharge relationships.

Historically, float gauges served as the primary flow measurement devices. The float gauges with chart recorders recorded flow depths, and each chart was digitized to create a record of flow depth (specifically, points of change in slope representing change in runoff rate were digitized). From the continuous stage records, flow rates were calculated with a known stage-discharge relationship.

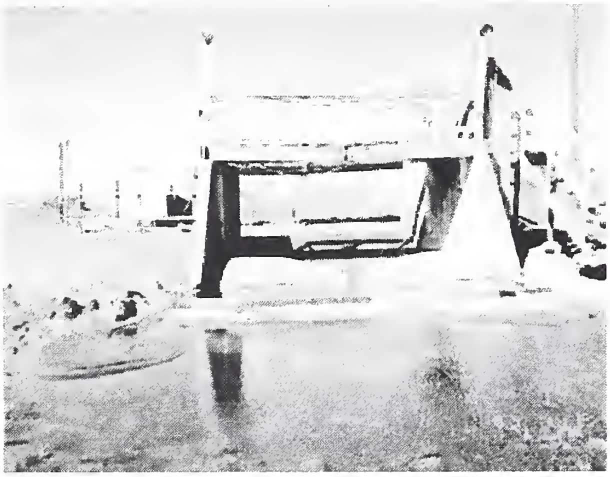

Figure 3. Flow control structure located on a 53.4 ha watershed.

Historical data from 1938 to 2002 for 40 runoff stations (approximately 1250 runoff station years) are accessible via the web. The internet site lists the stations, their watershed size, and the years for which runoff data are available for viewing and download. Specifically, flow rate (cfs and in/hr) with corresponding date and time (min) and daily runoff depth (in) data are available.

Water quality sampling

Water quality data, as well as flow data, are collected at the 18 active runoff structures. Currently, ISCO samplers are used to collect runoff samples, which are analyzed for dissolved and sediment-bound nitrogen and phosphorus concentrations and sediment amount. These automated samplers, installed in 2001, begin sampling when activated by a bubbler flow level recorder. Discrete samples are taken on variable time intervals with more frequent samples taken on the rising leg of the hydrograph.

From the 1970s to 2001, runoff water samples were taken with Chickasha samplers (Chichester and Richardson 1992). These automated, mechanical samplers were turned on with a float-activated water level switch. Discrete samples were taken on variable time intervals with more frequent samples taken on the rising leg of the hydrograph. Prior to the 1970s, runoff water samples were collected by hand during runoff events (Knisel and Baird 1970). On-call personnel collected discrete samples on variable time intervals similar to the collection frequency of the automated samplers.

Historically, sediment loss was the water quality issue of concern at Riesel, but limited nutrient and chemical data were also collected for specific studies. To quantify soil loss and sediment concentrations in runoff, collected sediment was dried, weighed, and recorded with the corresponding flow rate. Sediment data mostly from the late 1960s to 2002 for 22 runoff stations are accessible via the web. The internet site lists the stations and contains sub-daily concentration (ppm) and sediment amount (t/ac) data, and daily and monthly sediment amount data (t/ac).

Rain gauges

Collection of rainfall data at the Riesel watersheds began in 1938. Currently, 15 rain gauges are in operation (Figure 2). The Riesel rain gauge network is one of the denser rain gauge networks in the world with 15 gauges located within 340 ha. The current rain gauge network is comprised of Hydrologic Services tipping bucket rain gauges (Hydrologic Services PTY, Ltd., Sydney, Australia) connected to Campbell Scientific dataloggers (Figure 4). Sub-daily rain data are recorded in 10 min intervals (sensitivity 0.254 mm). A standard rain gauge is also used at each site as backup and calibration device.

Figure 4. Recording tipping bucket rain gauge and datalogger setup with adjacent standard rain gauge.

Historically, rainfall data were collected by various types of weighing rain gauges and recorded on chart recorders on 5 to 10 min intervals during precipitation events (sensitivity 0.254 mm). A standard rain gauge at each weighing gauge was used as a backup and calibration device. The electronic tipping buckets were installed in the 1990s.

Historical data from 57 rain gauges within the Brushy Creek watershed are available on the internet site, which list the stations and the years for which rainfall data are available (approximately 1350 rain gauge years). Specifically, accumulated sub-daily rainfall (in) with corresponding date and time (min) and daily rainfall (in) data are available for viewing and download. Rainfall estimates for individual watersheds can be calculated from Thiessen polygon weights, which are also available.

Weather station

Selected weather data have been collected at the Riesel weather yard since 1938 (Figure 2). Currently, weather data are measured and recorded with a Campbell Scientific weather station and datalogger, which were installed in 1996. Meteorological data collected includes: air temperature (average, maximum, minimum -°C), solar radiation (kJ/m^2), wind speed (m/s) and direction (°), precipitation (mm), maximum and minimum soil temperature (°C), and heating and cooling degree days. Daily data from the weather station are accessible on the web for 1996 to 2002.

From 1990 to 1995, daily maximum and minimum air temperature (°C), solar radiation (kJ/m^2), average wind speed (m/s), and rainfall (mm) data were measured with Campbell meteorological equipment connected directly to a computer at the Riesel headquarters building.

Daily temperature range data (°C) from 1940 to 1989 measured with a maximum temperature and a minimum temperature mercury thermometer are available. Daily evaporation data (in) measured from an evaporation pan from 1962 to 1972 are also available.

Lateral flow station

A lateral flow station was installed in 1970 to measure lateral subsurface flow from a portion of one watershed. Flow from a French drain is released into a boxed, sharp-crested v-notch weir. In 2000, a KPSI pressure transducer was installed with a Campbell Scientific datalogger to measure and record flow depth over the weir. Lateral flow is calculated with the weir's stage discharge relationship. Flow data (cfs) from late 2000 through 2002 are available on the website.

Well stations

In 1998, seven new wells were installed at Riesel to monitor shallow aquifer water levels and recharge properties. Five of the new wells are instrumented with KPSI pressure transducers connected to Campbell Scientific dataloggers to provide a continuous record of piezometric head. All of these seven active wells are monitored twice-weekly with a hand-held "e-line" water depth gauge to provide back-up and calibration data. Groundwater level data (relative elevation - ft) from these seven wells from late 2000 through 2002 are currently available on the website.

Telemetry network

Installation of a Campbell Scientific radiotelemetry network was completed in 2001. A total of 39 field telemetry stations were established at the 18 runoff stations, 15 rain gauges (including the weather station), lateral flow station, and the 5 shallow wells. A base station was established at the Riesel headquarters building to communicate with the field stations with a VHF radio signal. An automated data collection schedule runs continuously and collects data daily from each field station. From the base station, we can maintain and calibrate equipment and monitor realtime weather conditions, water levels, and rainfall amount. The base station is linked via phone modem to a dedicated computer at the laboratory headquarters in Temple, TX to automated data transfer. This phone link also allows manual operation and adjustment of the Riesel radio telemetry network from Temple. This radio telemetry network is unique in its location, the Texas Blackland Prairie, and is valuable in its ability to provide up-to-date hydrological data.

Current Research

The rare combination of a long, continuous hydrologic record and land dedicated for research make the Riesel facility a valuable research site, particularly for studies of hydrologic processes, water quality mechanisms, and climate change impacts. Previous research at Riesel includes water quality studies (Williams et al. 1971, Kissel et al. 1976, Richardson and King 1995), farming practice evaluations (Baird et al. 1970, Baird and Knisel 1971), and natural resource modeling (Arnold and Williams 1987, King et al. 1996, Harmel et al. 2000).

Several research projects are in progress at Riesel. These projects require data collection from land available for long time periods, reliable research funds, and dedicated staff. Without such a research commitment, studies such as these would be much more difficult and possibly infeasible.

A study of the water quality, crop production, and economic impacts of land-applied poultry litter was initiated in 2000. It is hoped that this cooperative study will continue for many years and address long-term impacts, which are often not addressed in short-term studies. The results of this study, funded by the Texas State Soil and Water Conservation Board and USDA-ARS, should impact agricultural and environmental policy and practice in terms of crop and pasture production, poultry litter utilization, and water quality.

Another current study at Riesel is attempting to quantify the hydrologic influence of the Blackland Prairie soils, which commonly exhibit strong shrink/swell potential. This behavior and the resulting slow permeability when soils are wet and rapid infiltration into cracks when soils are dry impacts rainfall/runoff, recharge, and lateral flow relationships. In addition to affecting hydrologic relationships, the shifting associated with shrink/swell processes can cause significant damage to building and road foundations. The soil physical properties also create challenges for tillage implement performance. Results from this study should provide valuable information on water resource management, construction design, and crop production.

Other research underway at Riesel includes studies of drought effects on native prairie pasture, nutrient cycling as influenced by soil microbial activity, performance of automated storm water sampling equipment, and nutrient transformations in land-applied poultry litter.

Conclusions

The hydrologic instrumentation network at the USDA-ARS Grassland, Soil and Water Research Laboratory watershed facility near Riesel, TX has been a valuable research tool since the late 1930s. The continuous data collection effort makes Riesel one of the most intensively monitored hydrological research sites in the country. Many years of data on runoff, water quality, precipitation, weather, and groundwater levels resulting from this coordinated monitoring effort are available on the USDA-ARS Grassland, Soil and Water Research Laboratory, Temple, TX website. The recent instrumentation updates should allow Riesel to remain a premiere research site for many years and continue to provide timely, pertinent information that benefits agriculture, the environment, and natural resource management.

Acknowledgments

We would like to recognize the efforts of many technicians that have contributed to the collection of data at Riesel. Operation of a hydrologic instrumentation network of this magnitude requires a talented and dedicated staff; and we have been blessed with a wonderful staff. We especially want to recognize current staff members Lynn Grote, Steve Grote, James Haug, and Gary Hoeft for their outstanding efforts in equipment maintenance, data collection, and record keeping. Without their service the intensive hydrologic monitoring program at Riesel would not be possible.

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Information Technology Applications in the ARS Watershed Network

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Abstract

Knowledge gaps relating to water supply, quality, and cycling processes have been identified as critical obstacles to improved water resource management in recent assessments. One critical gap identified was lack of adequate data to evaluate climate and hydrologic processes, particularly variability associated with climate and hydrology that affects management responses. The USDA, Agricultural Research Service operates long-term research watersheds in major physiographic regions of the US. These watersheds provide under-utilized opportunities to evaluate interactions of land use change, management practices, and climate variability in national assessments. A pilot project is underway to increase accessibility and utility of ARS Watershed Network data for enhanced research programs and to support a wide array of stakeholders. Approaches include: 1) evaluate data management models in other ecological and natural resources research networks, 2) develop formats and standards for metadata and data files, involving researchers, end-users, informatics and data base management specialists and 3) develop a common platform to access the data from multiple locations. The overall objective is to provide improved access to the watershed data for internal and external researchers, while retaining local control of and responsibility for the data. New data management systems for the experimental watersheds are expected to reduce delays and costs of developing new research thrusts and partnerships and increase data availability across the entire period of data collection and across different types of data. Users would obtain high quality and timely data, consistent across watersheds. All of this could extend the application of ARS watershed research to ecologic and socioeconomic, as well as agricultural and water resources problem-solving.

Keywords: informatics, hydrology, web-based data products

Introduction

Water resources face growing pressure globally, and with the prospect of possible future climate change, the water cycle (changes in precipitation frequency and intensity; evapotranspiration, runoff, snowmelt) will likely pose severe societal challenges (Gleik 1998). The critical role of experimental watershed data in the quest for hydrologic scientific understanding was clearly stated in the NRC (2001) report *Envisioning the Agenda for Water Resources Research in the Twenty-First Century*, "Intensifying water scarcity cannot be successfully addressed in the absence of reliable data about the quantity and quality of water over time and at different locations. The end-of-century trend of investing fewer and fewer dollars in data-gathering efforts will need to be reversed if availability is to be adequately characterized." In *A Plan for a New Science Initiative on the Global Water Cycle*, Hornberger et al. (2001) emphasized that "beyond the need to collect new data, existing long-term records must be archived and preserved carefully, and observations must be continued indefinitely at sites with long high-quality records, so that patterns of temporal variability, including long-term, low-frequency fluctuations, can be identified and studied."

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Kinzig and others (Kinzig et al. 2000, Kinzig 2001) assessed environmental research needs, and recommended increased interdisciplinary research in the area of communicating scientific information, particularly potential benefits of information technology (IT) on flows of scientific information to diverse citizenry and stakeholder groups. Their report explicitly addressed interdisciplinary environmental research across natural and social sciences. In no arena is such interaction of biophysical and social processes greater than in water resource management.

The ecological research community has supported a strong thrust to develop cutting-edge information technologies to transform the research process in support of multi-location, synthetic analyses. The Long Term Ecological Research Network (LTER) (http://www.lternet.edu/data) has provided leadership to develop these technologies (Baker et al. 2000). Similar to genomics, in which scientific advances require collaboration and data sharing among many researchers, new approaches to data sharing and data management have been required in the ecological sciences. The term bioinformatics is most often applied in the realm of genomics, so different terms were adopted for natural resources. Many in the ecological community have adopted the term "ecoinformatics" (http://ecoinformatics.org). In this paper we will use the term "natural resources informatics." A recent effort in hydrology, the Hydrological Data and Information System (HYDIS) (http://hydis.hwr.arizona.edu) also offers insights for multi-location and multi-attribute data synthesis.

Multi-site research efforts and analyses are not new. However, challenges are posed as researchers work across sites, each of which is highly complex, to address questions that span multiple scales of time and space. Not the least of these challenges is how to efficiently compile data sets from dispersed locations and ensure that researchers understand the nuances of the variables from sites where they have not worked. In addition they must ensure that like data from different sites that may have been gathered with different instrumentation or different measurement protocols are comparable in further analyses. A specific consideration for managing hydrologic data in the context of agricultural ecosystems analysis is the need to geographically link soil, crop, and livestock data with physical and chemical hydrologic data. These links are critical if watershed-scale water requirements and water quality responses to agricultural systems are to be understood. Watershed data management systems will be most useful to interdisciplinary studies if they incorporate flexible georeferencing technology.

The LTER Network Information System (Baker et al. 2000) has worked methodically to develop methods to describe and archive data for diverse types of future analyses. One aspect that has required considerable effort is development of metadata systems (Porter and Brunt 2001). The metadata are "data about data" and provide descriptive information to enable researchers who were not involved in collecting or processing the data to understand the details of how the data were collected and processed. Recent and ongoing efforts focus on systems to electronically search data libraries to aid research teams in compiling appropriate data sets to address a particular scientific question. To successfully tackle such problems requires expertise from the data information and computing disciplines as well as expertise in the ecological and natural resources sciences (Baker et al. 2000). The NRC (2003) report *Frontiers in Agricultural Research* identified research in environmental stewardship and integration of leading-edge science concepts and techniques, of which informatics is an example, as an opportunity for USDA research to better address societal needs.

Watershed Research in the ARS

History and overview

The U.S. Department of Agriculture established experimental watersheds as early as the 1930s, when the Soil Conservation Service was first established, and have continued under ARS management since the agency's establishment in the 1954. The ARS watershed facilities serve as stable, high-quality, outdoor laboratories that provide research capacity to conduct basic field research, evaluate management impacts, document effects of global change, and develop new instrument and simulation technologies. Over 100 ARS experimental watersheds ranging in size from a few hectares to over 600 km^2, are currently operated at 14 locations in major physiographic regions of the contiguous United States (Figure 1). The network, including descriptions of individual watersheds, is described at http://www.nwrc.ars.usda.gov/watershed/.

There is no comparable network of experimental watersheds in the world that combines intensive observational infrastructure with a longitudinal knowledge base. This network provides a platform to address complex research questions related to climate variability, atmosphere-earth interactions, and hydrologic processes. Several of these watersheds have served as field sites for large multi-organization remote sensing campaigns. Many of the watersheds are also linked with other national networks to broaden the type of observations made and leverage infrastructure, including: USDA-NRCS Soil Climate Analysis Network, AMERIFLUX, Surface Radiation Network (SURFRAD), ARS Rangeland Carbon Flux Network, and DOE/ARM/CART.

Figure 1. Locations that support experimental watershed research programs within the Agricultural Research Service.

Major contributions from these watershed programs have been made to hydrologic science include development of innovative instrumentation to measure primary components of the water cycle and water quality; development, testing, and application of remote sensing technologies; rainfall frequency analyses from dense gauge networks to modify NOAA National Atlases; improved understanding of spatial and temporal variability of infiltration across a range of hydro-climatic conditions; and development and validation of numerous hydrologic and natural resource models, such as USLE, Curve Number, USDAHL, HYMO, ACTMO, SWRRB, AGNPS, CONCEPTS, CREAMS, GLEAMS, EPIC, KINEROS, SPUR, SWAT, SRM, WEPP, and RUSLE. More recently some of the ARS watersheds have served as core sites and successful examples of integrating science with local policy and decision-making within the UNESCO Hydrology for the Environment, Life, and Policy (HELP) Program (www.unesco.org/water/ihp/help).

Data management at ARS Watersheds

Much of the original instrumentation, installation and data processing procedures for basic rainfall, discharge and meteorological data were guided by Handbook 224 (Brakensiek et al. 1979). However, data collection evolved differently at individual locations to address different research needs and dramatically different watershed response across hydro-climatic regions (e.g. snow, thunderstorm, groundwater dominated watersheds). Instruments, parameters observed, and data reduction procedures vary among watersheds. All locations have information and data about climate, discharge, soils, topography, and land management. Data about channel properties and processes is variable. Some sites collect groundwater and water quality data, and the parameters monitored vary among sites.

Availability of data from the watersheds also varies by location. Until 1990, basic rainfall-runoff data were compiled by the ARS Hydrology and Remote Sensing Laboratory and can be obtained at http://hydrolab.arsusda.gov/wdc/arswater.html. About 16,600 station years of data are stored there from watersheds ranging from 0.2 hectares to 12,400 km^2. After 1990, centralized data compilation and archiving ended. For many of the watershed sites, data are not uniformly accessible across the entire period of data collection or across different types of data. Climate and hydrologic data are generally most easily available, while land management and vegetative cover records are most difficult to obtain in an easily useable form. Many sites are addressing these issues, but have done so independently of one another (e.g., Slaughter et al. 2002). The network as a whole has not implemented many new information technologies, leading to delays and high costs when developing new research thrusts and partnerships. Such issues likely have contributed to under-utilization of ARS watershed data to evaluate interactions of land use change, management practices, and climate variability in national assessments.

Natural Resource Informatics in ARS

Overview

Plans are being formulated to provide computer access to ARS watershed data using modern information technologies. Several locations (Table 1) have agreed to enter into a pilot project to apply new information management principles to the ARS watershed network.

Table 1. Experimental watershed research sites in Pilot Project.

Experimental Watershed	**Location/** *Region*	**Description†**	*Research Focus*
Little Washita River Est. 1961	El Reno, OK (Chickasha)/ Great Plains	610 km^2 (236 mi^2)	climate variability, remote sensing, model testing
Walnut Gulch Est. 1953	Tucson, AZ (Tombstone)/ Southwest	150 km^2 (58 mi^2)	semiarid rangelands, downstream water yield, erosion, remote sensing, global change, modeling
Reynolds Creek Est. 1960	Boise, ID/ Pacific Northwest and Great Basin regions	239 km^2 (92 mi^2)	rangelands, snow deposition and melt, riparian processes, model development
Little River Est. 1967	Tifton, GA/ Coastal Plains of the Southeast	334 km^2 (129 mi^2)	low gradient flow, riparian processes, water quality, mixed use watershed, model development
Deep Loess Research Station Est. 1964	Ames, IA (Treynor)/ North Central Corn Belt	30 ha (74 ac) 61 ha (150 ac)	gully and channel erosion, cropping systems, water quality, riparian buffers, model testing
Goodwin Creek Est. 1981	Oxford, MS/ Bluff Hills of lower Mississippi Basin	21.5 km^2 (8.2 mi^2)	remote sensing, riparian corridors, erosion and sedimentation, fluvial geomorphology, model testing
WE-38 Est. 1965	University Park, PA/ Appalachian Valley and Ridge	7.4 km^2 (2.9 mi^2)	runoff generation, groundwater, surface-subsurface interactions, water quality, land use and management, modeling

† Most ARS watershed have a nested instrumentation network structure with gauged, internal sub-watersheds with intensive instrumentation.

Approaches include: 1) evaluate successful data management models in other ecological and natural resources research networks, 2) develop formats and standards for metadata and data files, using an interactive, consensus approach (researchers, end-users, informatics and data base management specialists), and 3) develop an operational structure for a common platform or linkages for the network. The overall goal is to develop new technologies to provide improved public access to the watershed data, while retaining local control and responsibility for the data.

Status

Data management will continue to be centered at the individual sites. An information system will be developed to extract, convert, and label data from multiple sites from a shared platform (Figure 2). The system will differ from that supported by the ARS Hydrology and Remote Sensing Laboratory through 1990 in several ways: 1) broader range of data types, e.g., weather, stream flow, topography, soils, land use, management practices, water quality or other parameters, depending on the site, 2) efficient operations that would allow automated updates of data, and 3) more diverse applications because of the more diverse data layers.

There are two primary contacts for each location, one focusing on hydrologic research issues and the other on data management issues. As this project develops, support staff and data managers will require more opportunity to network across sites than they have had historically and increased multisite researcher-data manager interactions. The Agency has committed support for a research associate to work across sites, to help ensure that data are accurately and adequately described. Additionally, this project will be supported by and provide products to the case-study watershed component of the NRCS-ARS Conservation Effects Assessment Program being conducted under the 2002 Farm Bill.

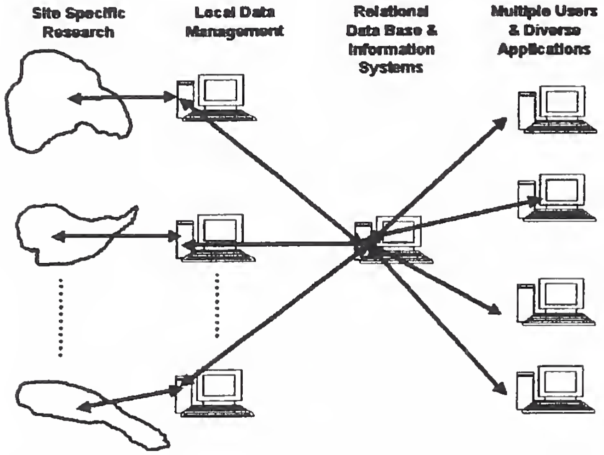

Figure 2. Data system with local control but with access to extract, convert, and label data from multiple sites for diverse problem solving.

Next Steps

As a team, we have identified data management and informatics models (e.g., U.S. Long Term Ecological Research Network http://www.lternet.edu/data/ Oklahoma Mesonet, http://okmesonet.ocs.ou.edu/, USDA UV-B Monitoring and Research Program, http://uvb.nrel.colostate.edu/UVB/uvb_climate_network.html, genomics data networks, and others) that can provide a framework to develop our system. The team will also coordinate with related activities in other USDA agencies, e.g., the Forest Service (FS) and Natural Resources Conservation Service, to ensure what we implement is compatible with other USDA activities. There may be opportunities to pursue a data network collaboratively with FS or other agencies. Three of the FS experimental watersheds are also LTER sites, and they have adopted the LTER system for data management (www.fsl.orst.edu/hydrodb/index.htm).

A critical step is to define formats and standards for metadata and data files. Because of the diverse research approaches and broad partnerships at various watershed sites, we will use an interactive, consensus approach that involves researchers, end-users, data managers, and IT specialists. Even though a pilot activity is planned, researchers and data managers from all the sites will be engaged in this step to facilitate their future incorporation into the data network. It will also be important to explore options for linking an ARS network into existing natural resources informatics networks.

In developing an operational structure, the research team is developing new partnerships within the agency to tap IT expertise. Historically, ARS has maintained a centralized IT staff primarily to support headquarters and administrative functions. The ARS Office of the Chief Information Officer (OCIO) recently developed a new function of support to Digital Research Products. Acceptance of this project within their program would provide IT staff time and some funding for development of the new information system. It could also enhance national/agency level visibility of this product (multi-site natural resource data base). Additionally, the National Agricultural Library has considerable expertise in information management and dissemination and may provide ideas and support to the effort.

This project must operate within Agency data, modeling, and GIS policies. The Agency policy may need to be updated to address critical issues such as standardization, quality control, accessibility, security, (e.g., Office of Science and Technology Policy data policy for Global Climate Research Program http://www.gcrio.org/USGSCRP/DataPolicy.html). In the future, this project could be linked to an ARS-NRCS initiative to develop an Object Modeling System in which simulation modules and appropriate databases could be assembled from a library in order to address specific scientific or natural resources questions.

Conclusions

Recent scientific assessments (NRC 2001, Hornberger et al. 2001) identified critical knowledge gaps relating to water supply, quality, and cycling processes. These reports highlighted the lack of adequate data to evaluate climate and hydrologic processes and variability. The ARS experimental watersheds provide exceptional "outdoor laboratories" where knowledge can be developed to address societal water resource issues in real world settings. These experimental watersheds provide stable and powerful research platforms to support collaborative research to investigate the hydrologic cycle and potential changes to it across a wide range of hydro-climatic conditions and agricultural ecosystems. However, lack of uniformity in data management and availability across sites and within sites impedes such new research collaborations. A pilot project to introduce natural resource informatics within the watershed network is underway to improve the data accessibility and utility. The project should increase productivity within research units, collaboration across units, and multi-partner collaboration. Internal and external researchers should obtain high quality and timely data, consistent across watersheds, extending the application of ARS watershed research to ecologic and socioeconomic, as well as agricultural and water resources problem-solving.

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The authors gratefully acknowledge the contributions of the many researchers and support staff at the experimental watersheds who have operated and maintained these research programs through the years. Their faithful and dedicated service provides the basis upon which this natural resource informatics project is built.

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The USDA-ARS Southeast Watershed Research Laboratory and Little River Experimental Watersheds

Joe Sheridan, David Bosch, Robert Hubbard, Richard Lowrance, Tom Potter, Tim Strickland, Clint Truman, Don Wauchope

Abstract

The USDA-ARS Southeast Watershed Research Laboratory (SEWRL) is one of six watershed hydrology research centers created by Congress as a result of Senate Document 59 to address critical hydrology and erosion research needs within major physiographic regions of the United States. The SEWRL, funded in 1965, has a primary focus within the Gulf-Atlantic Coastal Plain, an important agricultural region making up approximately 10 percent of the U.S. land area. A priority identified by Congress was establishing a hydrologic monitoring program to provide research quality watershed-scale hydrologic data within the region. Reliable, long-term hydrologic databases were not previously available for heavily vegetated, low-gradient stream systems. The Little River Watershed (LRW) in southern Georgia was selected as representative of mixed-use, agricultural watersheds in the region. In 1967, the SEWRL instrumented a 334 km^2 headwater portion of the LRW. The original network provided measurement of rainfall at 52 locations, stream stage at eight sites and groundwater stage at three locations. Currently, over 30 years of hydrologic data are available from up to eight watersheds ranging in area from 2.6 to 334 km^2. The SEWRL provides hydrologic data for characterizing Coastal Plain hydrologic processes and for development and testing of hydrologic modeling concepts. The hydrologic data, along with associated environmental quality data, have been used to support a broad range of natural resource and water quality research. This research has been aimed at developing agricultural management practices and systems that conserve natural resources while maintaining or enhancing the quality of our environment.

Keywords: watersheds, hydrology, streamflow, water quality

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Erosion I

Incorporating Bank-Toe Erosion by Hydraulic Shear into a Bank-Stability Model: Missouri River, Eastern Montana

Andrew Simon, Eddy J. Langendoen, Robert Thomas

Abstract

Bank-stability concerns along the Missouri River, eastern Montana are heightened by a simulated change in flow releases from Fort Peck Dam to improve habitat conditions for Pallid Sturgeon. The effects of the simulated flow releases on streambank pore-pressures and bank-toe erosion needs to be evaluated to properly model bank-stability. The Bank-Stability Model used incorporates pore-water pressure distributions, layering, confining pressures, reinforcement effects of riparian vegetation and complex bank geometries to solve for the factor of safety. To increase the applicability and accuracy of the model for use in predicting critical conditions, the hydraulic effects of bank-toe erosion have been added.

According to the simulated flow-release plan, flows of 216 m^3/s are increased by 38.3 $m^3/s/day$ for 12 days to 675 m^3/s, held for 60 days and decreased for 12 days back to 216 m^3/s. Results show the important contribution of bank-toe erodibility in controlling mass failure. Banks at River Miles 1624, 1676 and 1716 attain $F_s < 1.0$ indicating imminent failure. These sites contain less resistant sandy-silt material at the bank toe, and experienced simulated undercutting up to 3m. More resistant cohesive, clay bank toes at River Miles 1589 and 1762 were undercut only 0.2 m and remained stable.

Keywords: bank-stability, toe erosion, dams

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Introduction

Fort Peck Dam was constructed on the Upper Missouri River between 1933 and 1940. Closure in 1937 radically modified the downstream regime. Ecologic and geomorphic function of the Missouri River, with the natural cycle of intermittent, short duration, high flows during spring were replaced by a system of controlled, long duration, moderately high flows during the winter. This regime, combined with the sediment-depleted nature of the discharge, led to downstream incision. Work by Simon et al. (1999a) showed that bank instability along the Missouri River is promoted by a combination of channel incision, fluvial undercutting and increases in pore-water pressure in the bank due to the modified flow regime. During long-duration flow releases, water infiltrates into the riverbank eliminating matric suction that enhances soil strength and promoting positive pore-water pressures. Infiltration is enhanced compared with natural events since high stages are maintained for a longer duration. When flow is lowered in regulated rivers this has traditionally occurred at a rapid rate, resulting in a loss of confining pressure that is often more rapid than the dissipation of pore-water pressure from drainage. Such drawdown conditions often result in bank instability and mass failures, with associated problems including loss of farmland and damage to water-supply inlets.

Methods

The objective of this study was to determine the potential impact of a synthetic spring release from Fort Peck Dam, MT on bank erosion of five downstream sites in eastern Montana. The study reach extends from River Mile 1762, below Fort Peck Dam, downstream to River Mile 1589. Five sites were chosen: River Mile 1762 (Milk River), River Mile 1716 (Pipal), River Mile 1676

(Woods Peninsula), River Mile 1624 (Tveit-Johnson), and River Mile 1589 (Nohly). These sites represent a range of typical conditions along the Missouri River in eastern Montana.

For the purpose of this study we assumed a hypothetical spring release every three years on the Missouri River downstream of Fort Peck Dam, to trigger migration and spawning by Pallid Sturgeon (*Scaphirhynchus albus*). The simulated regime involved raising discharge from a base flow level, taken to be 216 m^3/sec (8000 ft^3/sec) to a maximum level of 675 m^3/sec (25000 ft^3/sec) in stages over 12 days, maintaining flow for between 6 and 36 days (depending on water temperature) and then returning flow to the original base level over 12 days. These two extremes were taken as the 'best-' (6 days) and 'worst-' (36 days) case scenarios from a bank stability perspective, and an interim scenario with a peak of 18 days was also tested.

A two dimensional hydrology model, GeoSlope SEEP/W™ (GeoSlope International Ltd 1998) was used to evaluate the effect of the simulated flow regime on streambank pore-water pressures. These pore-water pressures were combined with geotechnical field data to perform bank stability assessment using a bank-stability model (Simon et al. 2000). The Bank Stability and Toe-Erosion Model was used to investigate the effects of high flows on bank-toe scour, and resulting bank geometry. The eroded bank profiles were then re-analyzed in the Bank-Stability Model to differentiate changes in bank stability due to hydrologic effects and those due to erosion.

Pore-water pressure modeling

The SEEP/W™ software package was employed to model pore-water pressures created under the imposed hydrologic conditions. SEEP/W is a two-dimensional finite element hydrology model that simulates the movement of water and the resulting pore-water pressures for both saturated and unsaturated conditions using Richard's equation.

A finite-element mesh was created for each site based on profiles measured in the field (Simon et al. 1999a) to provide a framework to model pore-water pressures created under the simulated flow regime. Saturated hydraulic conductivity (Table 1) required for the SEEP/W modeling was measured in the field during the summer of 2001. Initial soil moisture conditions were simulated running a steady state analysis on each mesh with average spring groundwater level and slight surface evaporation, to create a realistic soil moisture distribution prior to imposition of the flow release. Local stage vs. time functions were developed from rating curves and used as boundary conditions for the transient analysis, simulating flow as a series of time-dependent heads on nodes along the bank toe and face.

Bank-stability analysis

The Bank-Stability Model calculates the ratio [Factor of Safety (F_s)] between the forces that drive and resist mass-bank failure. The model accounts for the geotechnical properties of the bank material including soil shear strength (cohesion, angle of internal friction, and unit weight), positive and negative pore-water

Table 1. Geotechnical and hydrologic parameters input to SEEP/W, Bank Stability Model and Bank and Toe Erosion Model.

Site Name	River Mile	USCS	Friction angle, ϕ' (degrees)	Cohesion, c' (kPa)	Saturated unit weight (kN/m^3)	ϕ^b (degrees)	Saturated conductivity (ms^{-1})	Critical shear stress (Pa)	Erodibility coefficient (k) (cm^3/N-s)
Nohly	1589	ML	30.1	13.2	21.4	17	$3.2e^{-7}$	3.94	0.5
		CH-CL	29.1	7.34	22.2	17	$9.9e^{-7}$	10	0.32
Tveit-Johnson	1624	CL	26.9	9.4	20.6	17	$3.5e^{-7}$	7.06	0.38
		CL	5.5	31.5	20.8	17	$3.5e^{-7}$	7.06	0.38
		SM	32.9	1.85	23.0	17	$5.0e^{-6}$	1.34	0.86
Woods Peninsula	1676	SM	26.9	0.36	21.4	17	$2.0e^{-6}$	1.34	0.86
		CL	0	78.9	21.6	17	$2.0e^{-6}$	7.06	0.38
		SP	35.0	0	21.6	17	$1.3e^{-6}$	0.31	1.8
Pipal	1716	SM	37.7	0	21.0	17	$8.5e^{-6}$	1.34	0.86
		CL-CH	13.4	22.3	21.4	17	$2.3e^{-8}$	10	0.32
		SM	37.9	0	20.9	17	$8.5e^{-6}$	1.34	0.86
Milk River	1762	CH	9.9	27.7	20.2	17	$4.3e^{-6}$	13.4	0.27

pressure and confining pressure exerted by flow (Simon and Curini 1998, Simon et al. 1999b). The model assumes a wedge-type failure mechanism.

In the part of the streambank above the "normal" level of the groundwater table, bank materials are unsaturated, pores are filled with water and with air, and pore-water pressure is negative. The difference ($\mu_a - \mu_w$) between the air pressure (μ_a) and the water pressure in the pores (μ_w) represents matric-suction (ψ). The increase in shear strength due to an increase in matric suction is described by the angle ϕ^b. Incorporating this effect into the standard Mohr-Coulomb equation produces (Fredlund et al. 1978):

$$S_r = c' + (\sigma - \mu_a) \tan \phi' + (\mu_a - \mu_w) \tan \phi^b \qquad (1)$$

where S_r = shear stress at failure, $(\sigma - \mu_a)$ = net normal stress on the failure plane at failure. The value of ϕ^b is generally between 10° and 20°, and increases with the degree of saturation.

It attains a maximum value of ϕ' under saturated conditions (Fredlund and Rahardjo 1993). The effects of matric suction on shear strength is reflected in the apparent or total cohesion (c_a) term although this does not signify that matric suction is a true form of cohesion (Fredlund and Rahardjo 1993):

$$c_a = c' + (\mu_a - \mu_w) \tan \phi^b = c' + \psi \tan \phi^b \qquad (2)$$

Negative pore-water pressures (positive matric suction; ψ) in the unsaturated zone provide an apparent cohesion over and above the effective cohesion, and thus, greater shearing resistance.

The factor of safety algorithm used by the bank stability model represents the continued refinement of bank-failure analyses by incorporating additional forces and soil variability to equations 1 and 2 (Osman and Thorne 1988, Simon et al. 1991, Simon and Curini 1998, Casagli et al. 1999, Rinaldi and Casagli 1999).

Geotechnical data (Table 1) and bank geometry used in the bank-stability analysis were taken from Simon et al. (1999a) and from field investigations. Pore-water pressures were taken from the seepage modeling, and river stage at a given time (used to calculate confining pressure) was determined from the synthesized discharge hydrographs. The effects of bank-toe erosion on stability were investigated by re-running the model using iterated bank profiles generated by the Bank and Toe-Erosion Model.

Bank and toe-erosion modeling

During the summer of 2001 critical shear stress and erodibility of cohesive materials were measured on a variety of bank and bank toe materials along the Missouri River using a non-vertical submerged jet-test device (Hanson 1990, Hanson 1991). The device applies an impinging, submerged jet on the bank materials and measures the applied shear stress and erosion rate. The relation between the two is used to calculate the critical shear stress (at zero applied stress) and erodibility coefficient (k; the slope of the erosion rate vs. applied stress curve).

The Bank Stability and Toe-Erosion Model predicts the change in channel geometry that will result from exposure of bank and toe materials to flows of a given stage and duration. It calculates erosion of cohesives using an excess shear-stress approach from the model of Partheniades (1965):

$$\varepsilon = k\,(\tau_o - \tau_c)^a \qquad (3)$$

where ε = the erosion rate, in ms^{-1}; k is an erodibility coefficient, in m^3/Ns^{-1}; $\tau_o - \tau_c$ is the excess shear stress, in Pa; τ_o is the average bed shear stress, in Pa; τ_c is the critical shear stress, in Pa; and a = an exponent (often assumed = 1.0). The measure of material resistance to hydraulic stresses is a function of both τ_c and k. Results of almost 200 tests at stream sites from Arizona, California, Iowa, Mississippi, Missouri, Montana, Nebraska, Nevada and Tennessee indicate that k can be estimated as a function of τ_c (Hanson and Simon 2001):

$$k = 0.1\, \tau_c^{-0.5} \qquad (4)$$

Resistance of non-cohesive materials is a function of surface roughness and particle size (weight), and is expressed in terms of the Shields criteria.

Average boundary shear stress

Average boundary shear stress (τ_o) was calculated from the hydrograph via the rating curve, and from channel slope, using the method outlined in Langendoen et al. (2001).

The channel geometry parameters input into the Bank Stability and Toe-Erosion Model (bank heights, average bank angle and bank-toe length) were calculated from bank profiles. Channel slopes were calculated from thalweg elevations obtained from Simon et al. (1999a). The model was run using the simulated flow conditions as a driving input. The predicted bank profile was calculated on a daily basis and imported into the Bank Stability Model so that the stability of both the initial and the predicted bank profile could be assessed.

Boundary and critical shear stress used

Critical shear stresses for the bank materials measured are shown in Table 1, with values ranging from 0.3 to 13.4 Pa. The most resistant materials were clay layers ($\tau_c = 7.1 - 10.0$ Pa) while the least resistant were sand layers ($\tau_c = 0.3$ Pa). For any given flow, boundary shear stress at the five sites varies due to local channel gradient and channel geometry with narrow channels confining flow, resulting in higher shear stresses. Peak, local boundary shear stress at the break of slope between the bank and the toe for the five sites is as follows (from downstream to upstream); Nohly: 3.9 Pa, Tveit-Johnson: 4.9 Pa, Woods Peninsula: 2.0 Pa, Pipal: 3.2 Pa, Milk River: 4.5 Pa.

Bank Stability and Erosion Results

Results from the stability analyses are expressed in terms of a Factor of Safety (F_s). A value of 1.0 indicates the critical case and imminent failure; values above one are theoretically viewed as stable. However, the uncertainty and variability of soil properties and failure geometries results are such that we consider values between 1.0 and 1.3 *conditionally stable*.

River mile 1624 (Tveit-Johnson)

The streambank is 9.6 m high and composed of a basal layer of sandy silt approximately 4.5 m thick with an upper layer of clay. Initial results show the streambank to be stable ($F_s = 1.69$) during baseflow conditions, and that negative pore-water pressure in the streambank decreased during the initial 12-day rise in stage. Stability increased very slightly with the rise in flow due to confining pressure ($F_s = 1.71$). During drawdown the streambank drained rapidly, and experienced a slight decline in stability as confining pressure was released (Regime 1: $F_s = 1.54$, Regime 2: $F_s = 1.46$, Regime 3: $F_s = 1.40$) after drawdown assuming no bank-toe erosion. The results for Regime 1 showed that F_s had not started to recover at the end of the flow release so this simulation was extended to ensure that F_s never reached critical levels. The value after 70 days was 1.49, and after 140 days 1.48 where after it recovered slowly.

Although the non-eroded bank was quite stable, results indicate that streambank failure is possible when bank-toe erosion is accounted for. Flow at this site is somewhat confined, generating relatively high boundary shear stresses. Critical shear stress for the bank base material is 1.3 Pa, compared with a local boundary shear stress of 4.9 Pa during peak flow. End-of-simulation stability values accounting for erosion were as follows; Regime 1: $F_s = 1.31$, Regime 2: $F_s = 1.13$ and Regime 3: $F_s = 0.98$. As with the non-eroded simulation, under Regime 1 the F_s value had not recovered at the end of the initial period and an extended simulation was performed, resulting in a minimum value of 1.28 after 70 days. Bank-toe erosion produced increasingly large failures with each successive flow regime.

Results highlight the vulnerability of this site to bank-toe erosion. Even the shortest regime results in approximately 1.5 m of bank-toe erosion, with approximately 2 m of erosion under the worst-case flow release.

River mile 1589 (Nohly)

The streambank is 6.5 m high and is composed of a basal layer of clay approximately 2.5 m thick with an upper layer of silt. Initial results show the streambank to be stable ($F_s = 1.45$) during baseflow conditions. Negative pore-water pressure in the streambank decreased during the initial 12-day rise in flow and continued to decline during the period of maintained high flow. Stability increased ($F_s = 1.58$) during the initial 12-day rise in flow as confining pressure increased F_s more rapidly than rising pore-water pressures could decrease it. F_s decreased during the maintained high flow as pore-water pressure continued to increase due to water infiltration from the channel into the bank. During drawdown stability declined but the streambank remained stable under the two shorter regimes (Regime 1: $F_s = 1.34$, Regime: 2 $F_s = 1.31$) and conditionally stable under the longest regime (Regime 3: $F_s = 1.25$) as confining pressure was removed faster than drainage allowed the pore-water pressures to equilibrate.

Critical shear stresses for the materials at this site (10 Pa at the bank toe) exceeded the boundary shear stresses (3.9 Pa at the base), resulting in no erosion during the flow release. The results suggest that the simulated flow regime incorporates sufficiently slow changes in stage to maintain bank stability by allowing pore-water pressure to equilibrate.

River mile 1676 (Woods Peninsula)

The streambank is 6.4 m high and is composed of a 2.4 m thick basal layer of sand, a middle 0.6m thick layer of clay and an upper 3.4m thick layer of sandy silt. Initial results show the streambank to be barely stable (F_s = 1.06) during baseflow conditions.

Negative pore-water pressure was reduced during the initial 12-day rise in flow, with a lag effect due to the low permeability of the bank materials. Stability remained constant (F_s = 1.06 after 12 days) as the resisting force of confining pressure matched the driving force caused by the loss of negative pore-water pressure. Negative pore-water pressure continued to fall during the period of maintained high flow, and F_s fell accordingly. The bank became unstable under all three regimes, with minimum values of F_s = 0.61, 0.59 and 0.57 respectively for the three flow scenarios.

Although the bank toe material at this site has a low critical shear stress (0.3 Pa) the applied boundary shear stresses are also quite low (peak stress at bank toe of 2.0 Pa) due to the wide channel and low gradient. The relatively small amount of bank toe erosion that occurred was not sufficient to displace the optimum location for the failure surface significantly, and the minimum stability values are almost unchanged for the eroded banks. The results suggest that this site is already very vulnerable to instability, and that the simulated flow regime is likely to trigger streambank failure due to detrimental hydrologic effects.

River mile 1716 (Pipal)

The streambank is 6.7 m high and is composed of a 2.8 m thick basal layer of sandy silt, a middle 0.9m thick layer of brown clay and an upper 3 m-thick layer of sandy silt. Initial results show the streambank to be conditionally stable (F_s = 1.28) during baseflow conditions. Negative pore-water pressure decreased rapidly during the initial 12-day rise in stage. Stability declined sharply as the resisting confining pressure was less than the driving force caused by the loss of negative pore-water pressure. Negative pore-water pressure continued to decline during the maintained high flow as water continued to infiltrate the streambank. Under all three regimes the bank failed before drawdown began, with minimum values of F_s = 0.94, 0.91 and 0.8 reassuming no bank erosion. A small amount of toe erosion was predicted by the model, reducing bank stability to F_s = 0.88, 0.81 and 0.75 under the three regimes. Failure was due to a combination of loss of matric suction and bank-toe erosion. Both bank saturation and bank undercutting are critical issues at this site.

River mile 1762 (Milk River)

The streambank is 6.7 m high and is composed of homogenous dark brown silty clay. Initial results show the streambank to be extremely stable (F_s = 3.71) during baseflow conditions. A high level of stability is maintained throughout the simulated flow regime due to the relatively low bank angle compared to other sites, and the cohesive nature of the bank material. Negative pore-water pressure declined rapidly during the initial 12-day rise in flow level. Stability over this period increased slightly (F_s = 3.80) suggesting that the confining pressure of the flow was able to offset the rapid loss of suction caused by the infiltration of water into the streambank. Pore-water pressure remained fairly constant during the maintained high flow indicating rapid equilibration between channel and banks during this period and resulting in a fairly constant factor of safety. During the 12-day drawdown period the streambank remained very stable although F_s declined (F_s = 3.32, 3.31 and 3.31 under the three regimes) as confining pressure was removed faster than drainage allowed the pore-water pressure to equilibrate. Due to the high critical shear stress of the bank material (13.4 Pa compared with a peak local boundary shear stress of 4.5 Pa) no bank or toe erosion occurred. The combination of non-vertical banks, high cohesion and high critical shear stress resulted in a very stable bank.

Discussion and Conclusions

A combination of hydrology, erosion and bank stability modeling has been used to predict the impact of flood release on five riverbanks typical of conditions along the Missouri River in eastern Montana. The simulations and field data collection undertaken show a range of responses illustrative of different processes controlling bank-stability. Two sites (Milk River and Nohly) appear to be relatively stable and are unaffected by the simulated flood release. One site (Tveit-Johnson) resists the hydrologic (infiltration) effects of the flood, but is sensitive to basal undercutting of the banks. This site would require bank-toe protection to maintain stability. Woods Peninsula is close to the failure threshold under ambient spring conditions, and infiltration-induced failures are likely to be triggered by any increase in stage. The Pipal site is conditionally stable under ambient spring conditions, and is vulnerable both to infiltration and bank erosion.

Results indicate that the slow drawdown incorporated in the simulated flow regime permits pore-water pressure to dissipate sufficiently, and is not a factor in potential instability. In a wider context the work shows how site vulnerability, and potential remedies, can be identified relatively quickly using a combination of a comparatively sophisticated, seepage model coupled to two simple and widely accessible bank erosion and stability models. It also highlights the need to account for the three processes simulated here; comparison of the Factor of Safety data reveals the extent to which it is combinations of infiltration, hydraulic erosion and geotechnical failure that lead to bank failure. Simpler modeling approaches run the risk of overestimating bank stability.

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Effects of Forest Management on Streamflow, Sediment Yield, and Erosion, Caspar Creek Experimental Watersheds

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Abstract

Caspar Creek Experimental Watersheds were established in 1962 to research the effects of forest management on streamflow, sedimentation, and erosion in the rainfall-dominated, forested watersheds of north coastal California. Currently, 21 stream sites are gaged in the North Fork (473 ha) and South Fork (424 ha) of Caspar Creek. From 1971 to 1973, 65% of the timber volume in the South Fork was selectively cut and tractor yarded, and from 1985 to 1991, 50% of the North Fork basin was harvested, mostly as cable-yarded clearcuts. Three unlogged tributaries serve as controls.

Annual suspended sediment loads changed 331% after logging the South Fork compared to 89% for the North Fork and -40% to 269% for North Fork subwatersheds. In clearcut units, storm peaks increased as much as 300%, but as basin wetness increased, percentage peak flow increases declined. Flow increases are explained by reduced transpiration and interception. Ongoing measurements show a return to pre-treatment flow conditions approximately 12 years post-harvest, but sediment yields have yet to recover.

Landslides are predominantly associated with roads, landings, and tractor skid trails in the South Fork watershed and windthrow in the North Fork watershed.

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Keywords: peak flow, sediment, erosion, landslides, timber harvest

Introduction

For more than four decades, researchers have investigated the effects of forest management on streamflow, sedimentation, and erosion in the Caspar Creek Experimental Watersheds of north coastal California. The California Department of Forestry and Fire Protection and the USDA Forest Service, Pacific Southwest Research Station, began a simple paired watershed study in 1962 with the construction of weirs on the two major Caspar Creek tributaries, the North Fork and the South Fork. Initially, this partnership was born out of necessity. The research station was charged with evaluating harvest impacts in major timber production regions, but the National Forest system lacked significant ownership within the coast redwood Douglas-fir forest type. The Jackson Demonstration State Forest, comprised of nearly 20,000 ha of second-growth forest, met this need, and a successful, long-standing partnership was begun. As management practices have evolved, so, too, have the research questions and technologies. Today, researchers operate 21 gaging stations within the experimental watersheds and utilize state-of-the-art data loggers programmed with sophisticated sampling algorithms, instream turbidimeters, and automated pumping samplers to measure discharge and sediment transport. Additional investigations of the processes important to hydrologic and ecosystem function are emphasized. The Caspar Creek Experimental Watersheds have produced a wealth of data and an extensive library of scientific publications used to guide natural resource management policy.

Methods

Site

The Caspar Creek Experimental Watersheds are located about 7 km from the Pacific Ocean and about 10 km south of Fort Bragg in northwestern California at 39°21'N 123°44'W (Figure 1). Uplifted marine terraces incised by antecedent drainages define the youthful and highly erodible topography. Hillslopes are steepest near the stream channel and become gentler near the broad, rounded ridgetops. About 35% of the basins' slopes are less than 17 degrees, and 7% are steeper than 35 degrees. Elevation ranges from 37 to 320 m.

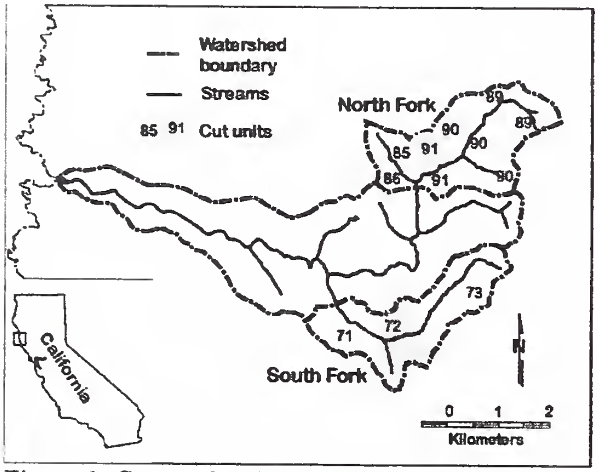

Figure 1. Caspar Creek Experimental Watersheds.

Soils are well-drained clay-loams, 1 to 2 meters in depth, derived from Franciscan greywacke sandstone and weathered, coarse-grained shale of Cretaceous age. Hydraulic conductivities are high and subsurface stormflow is rapid, producing saturated areas of only limited extent and duration (Wosika 1981).

The climate is typical of low-elevation coastal watersheds of the Pacific Northwest. Winters are mild and wet, characterized by periods of low-intensity rainfall delivered by the westerly flow of the Pacific jet stream. Snow is rare. Average annual precipitation is 1170 mm. Typically, 95% falls during the months of October through April. Summers are moderately warm and dry with maximum temperatures moderated by frequent coastal fog. Mean annual runoff is 650 mm.

Like most of California's north coast, the watersheds were clearcut and broadcast burned largely prior to 1900. By 1960, the watersheds supported an 80-year old second-growth forest composed of coast redwood (*Sequoia sempervirens* (D.Don) Endl.), Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), western hemlock (*Tsuga heterophylla* (Raf.).Sarg.), and grand fir (*Abies grandis* (Dougl. ex D.Don) Lindl.). Forest basal area was about 700 m^3 ha^{-1}.
Anadromous fish, including both coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*) inhabit the North Fork and the South Fork of Caspar Creek and are protected by state and federal endangered species regulations.

Study design

The Caspar Creek study is a classic paired watershed design where one or more gaged catchments are designated as controls and others are treated with road building, logging, and other timber management practices. After a calibration period wherein a statistical relationship between the catchments is defined, any subsequent change is inferred to be a treatment effect.

The 473-ha North Fork of Caspar Creek and the 424-ha South Fork of Caspar Creek have been gaged continuously since 1962 using 120° V-notch weirs widening to concrete rectangular sections for high discharges. During the early 1980s, three rated sections were constructed upstream of the North Fork weir and 10 Parshall flumes were installed on North Fork subwatersheds with drainage areas of 10 to 77 ha.

Stream discharge was initially recorded using mechanical chart recorders. These were replaced in the mid-1980s with electronic data loggers equipped with pressure transducers. Subsequent upgrades have been implemented as technology has progressed. Early suspended sediment estimates were derived from sediment rating curves, manual depth-integrated sampling, and fixed stage samplers (Rice, et al. 1979). Statistically based sampling algorithms that trigger automated samplers were utilized beginning in the 1980s (Lewis, et al. 2001). In addition, an annual survey of sediment accumulation in the settling basin upstream of each weir has been made since 1963.

Erosion measurements include periodic field surveys to document the location, size, and disposition of landslides. Erosion features greater than 7.6 m^3 (10 y^3) have been recorded annually since 1986. Erosion has on occasion been sampled at a finer scale using erosion plots (Rice et al. 1979, Rice 1996).

Treatment phase I: selection harvest with tractor yarding

After establishing a calibration relationship between the North Fork and the South Fork (1963 to 1967), a main-haul logging road and main spurs were built in the South Fork. The road right-of-way occupied 19 ha, from which 993 m^3 ha^{-1} of timber was harvested. The entire south Fork watershed was logged and tractor yarded between 1971 and 1973 using single-tree and small group selection to harvest 65% of the stand volume. Roads, landings, and skid trails covered approximately 15% of the South Fork watershed area (Ziemer 1981).

Treatment phase II: clearcutting with skyline-cable yarding

A study of cumulative effects began in 1985 in the North Fork watershed. Three gaged tributary watersheds within the North Fork were designated as controls while seven were designated for harvest in compliance with the California Forest Practice Rules in effect in the late 1980s. Two units (13% of the North Fork watershed) were clearcut in 1985-86 and excluded from the cumulative effects study. However, this harvest affects all subsequent analyses of North Fork weir data. After a calibration period between 1985 and 1989, clearcut logging began elsewhere in the North Fork in May 1989 and was completed in January 1992. Clearcuts occupied 30-99% of treated watersheds and totaled 162 ha. Between 1985 and 1992, 46% of the North Fork watershed was clearcut, 1.5% was thinned, and 2% was cleared for road right-of-way (Henry 1998).

In contrast to the harvest treatment of the South Fork in the 1970s, stream-buffer rules mandated equipment exclusion and 50% canopy retention within 15 to 46 m of watercourses providing aquatic habitat or having fish present. Most of the yarding (81% of the clearcut area) was accomplished using skyline-cable systems. Yarders were situated on upslope landings constructed well away from the stream network. New road construction and tractor skidding was restricted to ridgetop locations with slopes generally less than 20%. Four harvest blocks, 92 ha total, were broadcast burned and later treated with herbicide to control competition (Lewis, et al. 2001). Pre-commercial thinning in 1995, 1998, and 2001 eliminated much of the dense revegetation and reduced basal area in treated units by about 75%.

Results

Storm peaks

Ziemer (1981) analyzed peak discharges from 174 storm peaks occurring between 1963 and 1975 and later (1998) expanded upon this analysis with data collected through 1985. This analysis detected no significant increases in storm peaks following selection harvest of 65% of the South Fork watershed stand volume *except* within the smallest flow classes (recurrence interval less than 0.125 year). Early fall peaks increased by about 300%, but these were small storm events.

Lewis et al. (2001) analyzed the peak flow response to clearcutting in the North Fork using 526 observations representing 59 storms on 10 treated watersheds. After logging, eight of the 10 tributary watersheds experienced increased storm peaks ($p < .005$). In clearcut units, storm peaks increased as much as 300%, but most increases were less than 100%. The largest increases occurred during early season storms. As basin wetness increased, percentage peak flow increases declined. In the larger, partially clearcut North Fork watersheds, smaller peak flow increases were observed. Under the wettest antecedent moisture conditions of the study, increases averaged 23% in clearcut watersheds and 3% in partially clearcut watersheds. The average storm peak with a 2-year return period increased 27% in the clearcut watersheds (Ziemer 1998) and 15% in the partially clearcut watersheds. Ongoing measurements show a return to pre-treatment flow conditions approximately 12 years post-harvest and minimal response to the pre-commercial thinning (Figure 2).

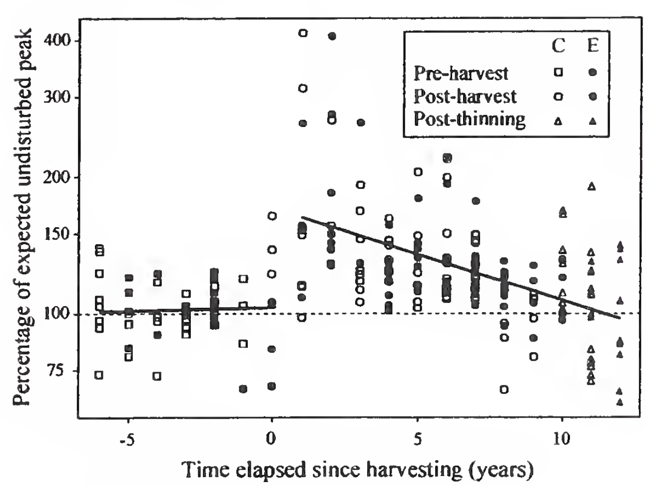

Figure 2. Peak flows observed in North Fork clearcut units C and E from 1986 through April 2003.

Reduced transpiration resulting in wetter soils in logged units explains some of the observed increases in streamflow. In addition, recent research at Caspar Creek has documented significant increases in net precipitation within clearcut areas due to reduced canopy interception. Under forested conditions, canopy interception is significant even during the wettest mid-season storms. Preliminary results show that, annually, about 20% more precipitation is delivered to the forest floor after logging.

Sediment loads

Sediment load estimates for the North Fork and South Fork are the sum of the sediment deposited in the weir pond and the suspended load measured at the weir. Comparison of sediment loads produced following the 1971-73 harvest of South Fork and the 1989-92 harvest of North Fork must be made cautiously. Improved and more intensive sampling methods greatly enhance the accuracy of load estimates for the latter study. And large landslides in the North Fork in 1974 and 1995 strongly influence the comparison.

On the South Fork, the suspended sediment loads increased 335% after road building and averaged 331% greater during the 6-year period after tractor yarding. Annual sediment load (including suspended and pond accumulations) increased 184% for the 6-year post-harvest period 1972-1978, returning to pretreatment levels in 1979 (Lewis 1998).

Using the South Fork as the control basin for logging the North Fork, no significant change in annual sediment load was detected after clearcutting 48% of the watershed area. However, analyses using tributary controls were more illuminating. Suspended sediment loads changed 89% at the North Fork weir, primarily due to one landslide in 1995, and –40% to 269% at other gaged locations. The mean annual sediment load increased 212% (262 kg $ha^{-1}yr^{-1}$) in clearcuts and 73% (263 kg $ha^{-1}yr^{-1}$) in partially clearcut watersheds. Recent data analysis suggests that sediment loads in North Fork tributaries remain elevated through water year 2002, more than a decade after harvest (Figure 3).

Erosion

Increased sediment loads in the South Fork following road building and tractor harvest are explained by increased sediment delivery to stream channels (Rice 1979). Road building and bridge construction within the riparian zone directly impacted much of the perennial stream. The following winter, 36 discrete landslides were documented along the newly constructed road—17 delivered an estimated 822 m^3 to the stream and 19 deposited 382 m^3 along the road

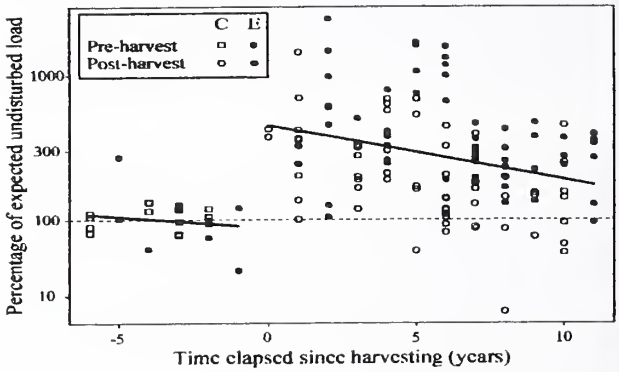

Figure 3. Sediment loads observed in North Fork clearcut units C and E from hydrologic year 1986 through 2002.

surface (Krammes and Burns 1973). Aerial photos of South Fork Caspar taken in 1975 portray 66 recently active landslides. Of these, all but three are associated with roads, landings, or skid trails (Cafferata and Spittler 1998). A field survey of landslides conducted in 1976, three years after tractor harvest was completed on the South Fork, recorded 99 discrete erosion features as small as 4.2 m^3 (150 ft^2). Landslides displaced approximately 189 m^3 ha^{-1} of material (Tilley and Rice 1977). Of these, 85% were associated with roads, landings, or skid trails. In 1994, this survey was repeated documenting 10 additional or re-activated landslides displacing 1515 m^3 of material. Only two of these were not road-related. Another episode of road-related landsliding was observed in the mid-1990s as stream crossing failures became more common. Of the 38 South Fork landslides documented between 1994 and 2003, 89% are road, landing, or skid trail related. These more recent landslides displaced 5804 m^3 and delivered 3503 m^3 to the stream channel. An aging system of logging roads and skid trails continues to deliver sediment to the stream channel.

North Fork sediment load increases were correlated to flow increases and, to a lesser degree, the length of intermittent channels logged or burned (Lewis et al. 2001). Increased erosion is attributed to increased gullying of headwater channels. Field investigations documented gullying and bank erosion in unbuffered channels subjected to intense broadcast burns and logging disturbance.

The annual inventory of failures exceeding 7.6 m^3 suggests that post-harvest erosion and sediment delivery mechanisms are quite different in the North Fork than were documented in the South Fork (Table 1). North Fork windthrow plays a far greater role in soil displacement, but delivers less displaced sediment to stream channels. Of 145 erosion features documented post-harvest (1990-2003), 84 were windthrow-related and only 10 were road-related. Uncut areas of the North Fork are included in this tally because these areas were impacted by edge-effect windthrow and new road construction. Windthrow displaced 2240 m^3 but delivered only 27% of this sediment. Clearcutting left adjacent timber stands and riparian buffers vulnerable to windthrow, but relatively little of the sediment displaced by uprooted trees was delivered to the stream. In contrast, road-related landslides on the North Fork delivered about half of the 3264 m^3 volume displaced. Most of these, including the largest (2012 m^3), are associated with the pre-existing mid-slope road that spans the north side of the watershed. This road was constructed circa 1950 to the standards of the time.

Table 1. Comparison of post-harvest Erosion features inventoried on the North Fork and South Fork.

Erosion Features	South Fork	North Fork
6-year post-harvest[1]		
Total number	99	81
Volume (m^3)	80046[2]	7285
Delivered Volume (%)	na	39%
Road-related number	85	6
Volume (m^3)	na	533
Delivered Volume (%)	na	8%
Windthrow-related number	na	45
Volume (m^3)	na	1204
Delivered Volume (%)	na	25%
1990-2003		
Total number	38	145
Volume (m^3)	5804	11878
Delivered Volume (%)	61%	45%
Road-related number	34	10
Volume (m^3)	5556	3264
Delivered Volume (%)	63%	52%
Windthrow-related number	5	84
Volume (m^3)	316	2240
Delivered Volume (%)	20%	27%

[1]1971-1976 on South Fork, 1990-1995 on North Fork.
[2]Reported as 100 yd^3 $acre^{-1}$ (Tilley and Rice 1977).

Most of the erosion features discussed above are smaller than 76 m^3. Of greater concern to land managers is how timber harvest alters the frequency of large landslides. Debris slides account for a major amount of mass wasting within the Franciscan geology of the Caspar Creek region. Such landslides occur infrequently in response to critical rainfall intensities. Clearly, mass wasting increased following tractor harvest of the South Fork, but attempts to discern a post-harvest change in landslide frequency in the North Fork have been inconclusive (Cafferata and Spittler 1998). Twelve large landslides have occurred post-harvest in the North Fork watershed. The two largest occurred in clearcut units more than 10 years after harvest and account for 60% (5617 m^3) of the volume of all post-harvest erosion features. Of the remaining 10, five occurred in harvest units and five in control watersheds. While serving as a control watershed, the North Fork experienced two other large landslides (in 1974 and 1985) that displaced 4568 m^3.

Bawcom (2003) evaluated 50 clearcut units on Jackson Demonstration State Forest including the 10 North Fork Caspar clearcuts. Of 32 recent debris slides larger than 76 m^3, 28 (two of six in North Fork Caspar) were road-related. Most were associated with decades-old roads low on the slope near watercourses. No increase in the rate of landsliding within JDSF clearcuts was detected.

Conclusions

Timber harvest and road building affect runoff processes, sediment yields, and erosion. Caspar Creek studies document increases in peak flows, suspended sediment loads, and erosion after two very different harvest treatments. Response was highly variable between treatments and among individual treated tributaries. California's modern forest practices rules appear to mitigate, but do not eliminate these impacts.

Changes in basin wetness and canopy interception explain post-harvest flow increases. Sediment loads following partial clearcutting were correlated to flow increases. With forest regrowth, flow increases diminish returning to pre-harvest flow conditions after about 12 years. Sediment yields do not appear to recover as quickly and persist at double the pretreatment levels 12 years after harvest.

Erosion and sedimentation from ground extensively disturbed by road building and tractor yarding remain elevated decades after harvest. The present condition of the South Fork watershed is typical of much of the tractor-yarded lands in the redwood region that are entering yet another harvest cycle. It is becoming crucial for landowners, regulatory agencies, and the public to understand the interactions between proposed future activities and prior disturbances. A third phase of Caspar Creek research is being initiated in the South Fork to examine the effects of re-entry on runoff and sediment production from previously tractor-logged redwood forests. Much remains to be learned regarding restoring impacted ecosystems and mitigating impacts from future harvests. The Caspar Creek Experimental Watersheds provide a long-term research resource for furthering this scientific endeavor.

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Variable Rainfall Intensity Rainfall Simulator Experiments on Semi-arid Rangelands

Jeffry Stone, Ginger Paige

Abstract

Most rainfall simulator experiments have used a constant rainfall intensity in their experimental design. However, when multiple intensities are used, the steady state infiltration rate tends to increase with increasing rainfall rate, indicating that runoff contributing area is a function of rainfall intensity. Hydrologic data from soil vegetation complexes (Ecological Sites) in Arizona and Mexico suggest that at typical rainfall simulator rainfall intensities, not all of the area is contributing to runoff with the effect being greater for coarse textured soils. Erosion data from similar Ecological Sites indicate that deposition can be a significant component of the total detachment on uniform slopes when microterraces are present. Variable intensity rainfall simulator experiments are necessary to understand and predict small scale hydrologic and erosion processes that may be important in evaluating the sustainability of rangeland hillslopes.

Keywords: runoff, erosion, plots, rainfall simulation

Introduction

Rainfall simulator experiments on rangelands have been conducted since the 1930s to investigate fundamentals of the rainfall/runoff/erosion process and the impacts of grazing management and land characteristics on these processes. Rainfall simulation provides a relatively easy and economical way of obtaining a large amount of data under controlled conditions in a short period of time. In addition, controlled application rates allow for the comparison of steady state infiltration response to alternative management systems, to differences in vegetation and soil characteristics, and facilitates model parameter identification.

The majority of rainfall simulator experiments on rangelands have used a constant application rate in the experimental design. At the Walnut Gulch Experimental Watershed (WGEW), the first rainfall simulator experiments used constant intensities of 100 mm/hr (Kincaid et al. 1964) and 45 mm/hr (Tromble et al. 1974). In the 1980's a series of experiments were conducted on 3 x 10.7 m plots using the Rotating Boom rainfall Simulator (RBS) (Swanson 1965) to parameterize the Universal Soil Loss Equation (USLE) (Simanton and Renard 1985) and Water Erosion Prediction Project (WEPP) model (Simanton et al. 1991). The USLE experimental design consisted of three simulation runs, a dry run of one hour, and 24 hours later, a wet and very wet run both for 30 minutes on three treatments, natural, clipped, and bared. The water application rate for all runs was 60 mm/hr. The WEPP experimental design added 0.75 m^2 bared plots for infiltration parameterization, two water application rates to the very wet run (60 and 120 mm/hr), and multiple overland flow rates introduced at the top of the bare plots. Although the variable flow rates were used for WEPP rill erosion parameter identification on the bared plots, the constant intensity wet run was used for parameterizing the effective hydraulic conductivity term, K_e (mm/hr), of the WEPP infiltration model, the Green-Ampt Mein-Larsen (GAML) equation (Mein and Larsen 1973).

With the introduction of multiple application rates, it was observed that the steady state infiltration rate tended to be higher at higher application rates. In

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Figure 1, rainfall, observed and predicted infiltration curves and hydrographs from the RBS experiment are plotted for a multiple intensity simulation run on a sandy loam soil. The observed steady state infiltration rate, f obs, was computed as the difference between the application rate and the runoff rate at steady state. The predicted infiltration curve, f pred, was computed by adjusting the GAML K_e until the computed runoff volume matched the observed. The predicted hydrograph, q pred, was computed using the IRS model (Stone et al. 1992) which routes rainfall excess using a method of characteristics solution of the kinematic wave equations. Note that the observed infiltration rate, f obs, varies with rainfall intensity and is higher at the higher rainfall rate while the fitted infiltration curve does not duplicate the observed infiltration response. The result of under predicting the infiltration rate at the higher application rate is over prediction of the peak discharge rate.

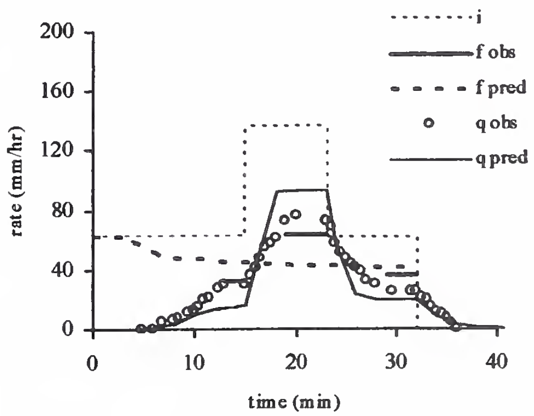

Figure 1. Example of observed and predicted hydrograph and infiltration by fitting the GAML model to runoff volume.

A proposed explanation for the increase in infiltration rate with increasing water application rate shown in Figure 1 is that there is a non-uniform distribution of infiltration capacity over an area such that portions of the area have higher infiltration capacities than other portions. The distribution of infiltration capacity is hypothesized to be caused by the spatial variation of soil and vegetation characteristics over the area. At the lower application rates, only those areas that have an infiltration capacity lower than the application rate will contribute to runoff. As the application rate increases, more of the area that has higher infiltration capacity contributes to runoff. Because of this, the infiltration rate computed as the difference between the application rate and steady state runoff increases. Hawkins (1982) suggested a relationship between the infiltration rate, $f_s(i)$ (mm/hr), and application rate, i (mm/hr), assuming an exponential distribution of infiltration capacity over an area as

$$f_s(i) = u_f\left(1 - e^{-\frac{i}{u_f}}\right) \tag{1}$$

where u_f (mm/hr) is the average aerial infiltration rate when the entire area is ponded. For Equation 1, the fraction of the area, $A_c(i)$, contributing to runoff for a given rainfall intensity is the cumulative density function of the exponential distribution or

$$A_c(i) = 1 - e^{-\frac{i}{u_f}} \tag{2}$$

The purpose of this paper is to examine the relationship between infiltration rates and rainfall intensity using data collected on soil vegetation complexes or Ecological Sites (ES) and to discuss the implications for runoff and erosion studies.

Methods

The data used for this paper are from two separate experiments, one using the RBS and one using the Walnut Gulch Rainfall Simulator (WGRS) developed by Paige et al. (in review). The WGRS is an oscillating boom simulator that uses the same nozzle, the VeeJet 80100, as the RBS and can apply water at variable intensities in user defined increments ranging from 12 to 177 mm/hr. Both experiments were conducted on similar ESs. The ES is the basis of a land classification scheme used by the Natural Resources Conservation Service (NRCS) in rangeland assessment and planning. The ES is defined by the National Range and Pasture Handbook (USDA NRCS 1997) as "...a distinctive kind of land with specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation."

Study areas

Five ESs located at the WGEW and in Chihuahua, Mexico were used for the RBS experiments; Sandy Loam Upland (SLU), Loamy Upland (LoU), Limey Slopes (LS), Limey Upland (LiU), and Clay Loam Upland (CLU). All of the soils were sandy to gravely sandy loams with the exception of the CLU ES that was a clay loam. The LS ES was brush dominated with no grazing and the remainder of the ESs had primarily grass vegetation with various levels of grazing intensity. The SLU ES had three separate locations with the ecological status ranging from fair to excellent and the CLU ES had two locations ranging from poor to excellent. All of the plots were 3 x 10 m and consisted of a natural treatment.

Two ESs at WGEW and at The Research Ranch (TRR) near Elgin, AZ were used for the WGRS experiment. All of the plots were 2 x 6 m and consisted of two treatments, natural (LoU-n and LS-n) on the WGEW and burned (LoU-b and LS-b) on The Research Ranch. The dominant vegetation at all of the sites was grass (pre-burn for LoU-b and LS-b) in good to excellent ecological condition. See Table 1 for additional characteristics of the ESs for both experiments.

Table 1. The range of canopy cover, CC, ground cover, GC, and average slope for the Ecological Sites.

ES	n[1]	CC (%)	GG (%)	Slope (%)
SLU	11	29-78	57-89	8
LiU	2	52	83	9
LS	2	34	84	11
LoU	4	17-46	26-51	11
CLU	8	23-31	31-39	3
LoU-n	2	88	82	8
LoU-b	2	0	73-29[2]	8
LS-n	3	64	60	11
LS-b	2	0	76-58	12

[1] number of plots, [2]pre and post simulation.

Experimental design

The RBS experimental design was similar to the WEPP design with the exception that three intensities were applied for the very wet run on some of the plots. For the WGRS experiment, the simulation run sequences were a 45 minute constant intensity run at 60 mm/hr followed one hour later by a variable intensity run. For the variable intensity run, the rates were changed after runoff had reached steady state for at least five minutes.

For both experiments, a flume was used to measure runoff depth from the plot that was converted to discharge using a pre-calibrated stage-discharge relationship. Rain intensity for the RBS was measured by a weighing bucket recording raingage and adjusted for wind effects by six non-recording rain gages distributed on the plot. Rain intensity for the WGRS was obtained through calibration and wind effects were minimized through the use of wind screens on the simulator. Canopy and ground cover were measured using a point frame at 490 points for the RBS experiment and 390 points for the WGRS experiment. Canopy cover was recorded as grass, shrub, or forb and ground cover was recorded as rock (>2 mm), litter, vegetative base, and bare soil. Ground cover was measured both outside and inside canopy cover. For the LoU-b and LS-b, cover was measured before and after simulation.

The RBS very wet run data were used to parameterize u_f in Equations 1 and 2 because the data represented a wide range of soils, vegetation composition, and ecological status. The WGRS multiple intensity data were used to examine the erosion response because of the large differences in canopy and soil surface characteristics caused by the burn treatment.

Results

RBS experiment

Plotted in Figure 2a are the f_s-i curves generated by manually optimizing for u_f in Equation 1 and the average and range of u_f are listed in Table 2. For the SLU ES, $f_s(i)$ of some of the simulator plots did not reach a final value at the highest rainfall intensity while for the CLU, a final value was reached at the lowest intensity for all the plots. The variability of u_f within an ES was greater than the variability among ESs. Using the criteria of no overlap of the ranges of u_f, the SLU, LoU, and CLU ESs are different while the LiU ES was similar to the SLU ES and the LS ES was similar the LoU ES.

Table 2. u_f values for the f_s-i relationship.

ES	u_f (mm/hr) average	u_f (mm/hr) range
SLU	82	50 - 130
LiU	57	55 - 75
LS	30	30
LoU	28	18 - 45
CLU	10	10

The range of runoff contributing area or partial area response, $A_c(i)$, with intensity was calculated using Equation 2 with the range of u_f in Table 2 and is plotted in Figure 2b. Referring to Table 3, at the lower intensity of 60 mm/hr, the partial area response is significant for the SLU ($A_c(i) = 0.38$-0.70), LiU ($A_c(i) = 0.55$-0.65), and portions of the LoU. At the higher intensity of 150 mm/hr, although most of the ESs have 90% or greater of the area contributing to runoff, some of the SLU and LiU plots still have less than 90% area contributing to runoff.

Table 3. Lower and upper limits of runoff contributing area, $A_c(i)$, for 60 and 150 mm/hr rainfall intensity using the range of u_f from Table 2 with Equation 2.

ES	A_c(60 mm/hr)		A_c(150 mm/hr)	
	Lower	Upper	Lower	Upper
SLU	0.38	0.70	0.70	0.95
LiU	0.55	0.65	0.86	0.93
LS	0.86		0.99	
LoU	0.74	0.96	0.96	1.00
CLU	1.00		1.00	

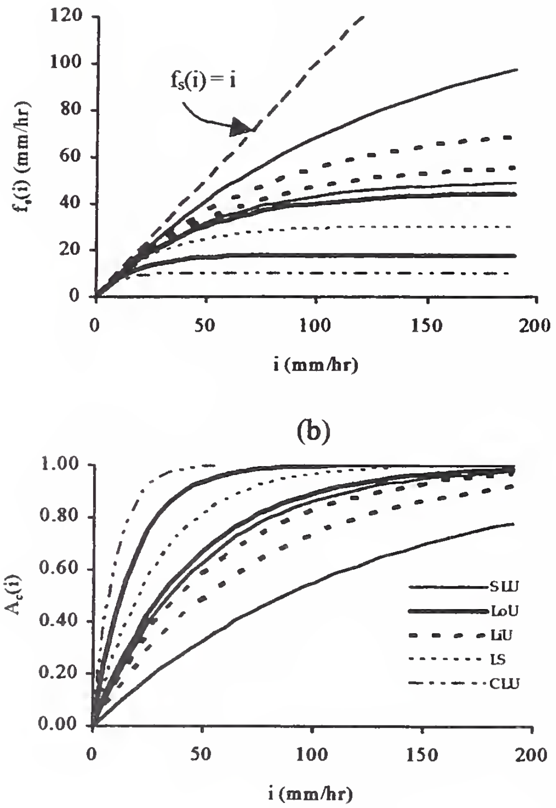

Figure 2. a) f_s-i relationship and b) contributing area for selected Ecological Sites in MLRA 41.

Using total ground cover, gc (%), in a regression with u_f, an exponential model proved to be the best fit and the following equation was obtained

$$u_f = 13.2\, e^{0.024\, gc} \quad R^2 = 0.66 \quad SE = 24.3 \qquad (3)$$

where R^2 = coefficient of determination and SE = standard error (mm/hr). For comparison purposes, the GAML K_e parameter was computed for the same data set and the following regression equation was obtained

$$K_e = 13.5\, e^{0.013\, gc} \quad R^2 = 0.44 \quad SE = 10.0 \qquad (4)$$

The data points and Equations 3 and 4 are plotted in Figure 3. Although there is a fair amount of scatter for both parameters, the similarity between the intercepts in Equations 3 and 4 implies that u_f is a conductivity term. The intercept is the bare soil value of the parameter and can be interpreted as a textural based conductivity that is modified for cover. The intercept value of 13 mm/hr is very close to the IRS model's (Stone et al. 1992) default bare soil saturated hydraulic conductivity of 10 mm/hr for a sandy loam based on Rawls et al. (1982). The positive correlation with ground cover suggests an interpretation of Figure 2a. As ground cover increases, u_f increases and higher rainfall intensities are required for the entire area to contribute to runoff. For an individual ES, the shape of the curves in Figure 2a may be an indicator of hydrologic condition with the flatter the curve, the poorer the hydrologic condition.

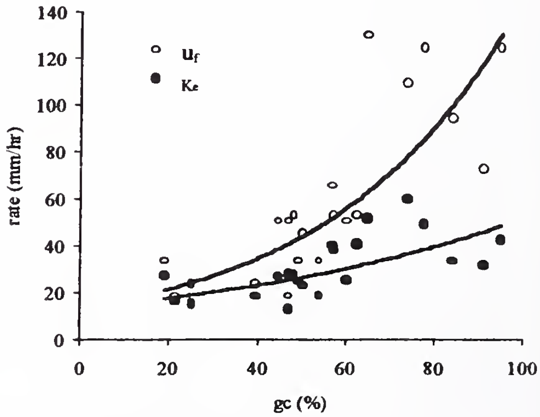

Figure 3. Relationship between ground cover, gc, and K_e and u_f.

WGRS experiment

The burned treatment was the result of a low to moderate severity wildfire that burned all the vegetation at TRR. Litter and ash from the fire made up a major portion of the ground cover. From observations during the burned plot simulations, litter was transported off the plot by overland flow or formed litter dams behind flow obstructions caused by rocks or vegetative bases. This process was dynamic, with the dams forming during the lower runoff rates of the dry and wet runs and being breached at the higher rates. After a dam was formed, sediment was deposited upstream from the dam creating a microterrace. When the dam was breached during the higher runoff rates, the runoff began to erode the microterrace much like a headcut. The geometry and number of microterraces on several of the plots was used to compute an estimate of deposited sediment. According to these calculations, about 40% of the detached soil was deposited on the LS-b plots and about 80% on the LoU-b plots. Although these estimates are very rough due to uncertainty of the initial microtopography, they do suggest that deposition of sediment on these sites is a significant component of the erosion process and that the microterraces had an ameliorating effect on the total sediment yield.

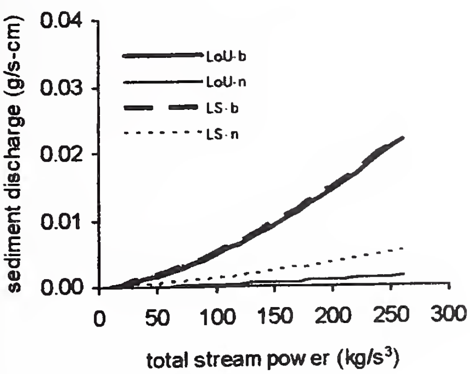

Figure 4. Steady state sediment discharge rate versus total stream power.

The dynamic nature of the erosion process was more dramatic at the burned sites but was also present at the unburned sites. As an illustration, steady state sediment discharge, q_s (g/s-cm) is plotted versus total stream power, ω (kg/s^3) in Figure 4 for the two treatments on the two ESs. Stream power is computed as ρ gqS where ρ = density of water (kg/m^3), g = gravitational constant (m/s^2), q = unit discharge (m^2/s), and S_0 = average plot slope (m/m). There was a very strong log-log relationship between q_s and ω for both the burned and unburned plots with coefficients of determination all greater than 0.85. The two burned treatments had very similar relationships and notably higher sediment discharge rates at the same stream power when compared to the unburned treatments.

Discussion and Conclusions

The vast majority of rainfall simulator experiments have used a single application rate as part of the experimental design. For example, Alberts et al., (1995) used the wet run with a 60 mm/hr application rate from the WEPP field experiment to parameterize K_e. However, as shown in Figure 2b, for ESs with coarse texture surface soils, one rate does not ensure that the entire plot is contributing to runoff. In the case of parameterizing K_e, if A_c(i) is not known a priori, then the selection of a single application rate is arbitrary and the resulting parameter value may not fully describe the hydrologic response of the site. A single application rate may also lead to mis-interpretation of results or incorrect relationships between hydrologic variables and plot characteristics. Under partial area response, local rates and amounts of runoff and erosion and hydraulic parameters such as flow shear are underestimated if the entire plots is assumed to be contributing.

In general, erosion models do not account directly for the influence of microtopography on detachment and deposition nor will they compute deposition on a uniform slope. Although the formation of microterraces on the burned plots occurred within the duration of the simulation experiment, their influence on sediment yield appeared to be significant. Preliminary studies on the characteristics of microterraces on the LoU-n and LS-n ESs suggest that these features make up from 30-50% of the hillslope microtopography. The rate at which microterraces form and how they affect both the runoff and erosion processes on unburned areas is largely unknown. Variable intensity rainfall simulator experiments should give insight into these processes. For example, from observations during the lower application rates at the burned sites, water ponded on the microterraces behind the litter dams while water did not pond on the sloping non-microterrace areas. Future experiments are planned using tracers to quantify how different hillslope microtopographic areas respond at different rainfall intensities.

The hydrologic indicators of the multi-agency rangeland health evaluation (Pyke et al., 2002), such as water flow paths, the presence of erosional pedestals and microterraces, can be interpreted as qualitative descriptors of partial area processes. In order to quantify and model hydrologic and erosion processes and their effects on the sustainability of an ecosystem, it will be necessary to define hydrologic relationships such as the f_s-i relationship described in this paper.

Acknowledgments

The authors appreciate the reviews of Mariano Hernandez and Evan Canfield. The fieldwork done in Mexico was funded by the USDA Foreign Agricultural Service, Office of International Cooperation and Development Grant MX65. Dr. Paige was partially funded through a grant from the Semi-Arid Hydrology and Riparian Areas Science and Technology Center, University of Arizona, Tucson, AZ. Dan Robinett and Don Breckenfeld of the Arizona State NRCS assisted with Ecological Site classification in Arizona and Mexico. Bill Brennan and Linda Kennedy of the Audubon Society facilitated the research at The Research Ranch.

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Methodology for Determining Effects of Extent and Geometry of Impervious Surface on Hydrologic Balance

Elizabeth Warnemuende, William Shuster, Doug Smith, James Bonta

Abstract

Urbanization of watersheds previously managed for agricultural uses results in hydrologic changes associated with increased flooding, erosion, and surface water degradation. Few studies have been conducted to quantify these effects under controlled conditions and standard rainfall simulation methodologies have not been established. In this project, the feasibility of rainfall simulation methods to evaluate hydrologic and erosional responses to various impervious treatments is examined. In addition, a modular segmented soil box design is developed in order to quantify the hydrologic, erosional, and water quality impacts of specific urban land use configurations, including the impacts of land uses of areas hydrologically connected to impervious areas. Hydrologic, nutrient, and pesticide data from runoff under rainfall simulation will be collected and analyzed. Treatments will include the following distributions of imperviousness at the 20%, 30%, and 40% total impervious area level: effective impervious elements each 1.25%, 5%, and 20% of total hydrologic area, non effective impervious elements each 1.25%, 5%, and 20% of total hydrologic area. In addition, potential urban and turf best management practices of will be evaluated. In conjunction with this study, a field study will be conducted at the North Appalachian Experimental Watershed near Coshocton, OH by members of the USDA-ARS and U.S. EPA. The field study will investigate impervious surface effects at the small watershed scale.

Keywords: urbanization, impervious surfaces, urban runoff, rainfall simulation

Introduction

The current socioeconomic climate favors the conversion of land previously in agricultural management for urban and suburban uses. As agricultural watersheds are urbanized, the resultant increase in impervious rooftops and transportation surfaces becomes a major controlling factor of the new urban watershed hydrology. Precipitation that falls on rooftops and pavement quickly runs off, instead of infiltrating into the soil as it would generally do in a natural or farmed landscape. This often results in increased runoff volume, peak flow rates, soil erosion, and contaminant transport, and decreased time of concentration. The economic and environmental impacts of the resulting damage to property and ecosystems are significant.

Hydrology of urban watersheds is often characterized by the extent and type of impervious areas, but impacts of the spatial and size distributions of impervious elements is not well understood (Shuster et al. 2003). The Soil Conservation Service curve number procedure is commonly used to estimate runoff. Different methods that have been previously used to estimate curve numbers for impervious areas and their host

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watersheds yield highly inconsistent estimates (Pandit and Regan 1998). These methods do not explicitly account for spatial and size and distributions of impervious surfaces. Rainfall simulations have been used to estimate curve numbers (Hawkins 1979, Pierson et al. 1995), but methods are needed to adapt these methods for urbanized settings.

The primary objectives of this study are to develop laboratory rainfall simulation methods sensitive to the spatial and size distributions of impervious surfaces, and to use these to evaluate the hydrologic, erosional, and water quality impacts of various impervious surface configurations and to guide field work in this area. This project is intended as a supporting project to a U.S. EPA/USDA-ARS joint pilot project investigating urbanization by utilizing existing experimental watersheds at the North Appalachian Experimental Watershed near Coshocton, Ohio, having the overall objective of determining the impacts of increasing urbanization on hydrology and water quality. This paper describes initial efforts to detect runoff and sediment loss differences between impervious surface configurations under laboratory rainfall simulation, and proposes a modular soil box design for this purpose.

Methods

Rainfall simulation

A programmable variable – intensity oscillating nozzle rainfall simulator (Foster et al. 1979) was used for all trials. Vertical distance between the nozzles and the soil surface was approximately 2.5 m, and nozzle pressure was 41.4 kPa. The water source was deionized.

Prior to each rainfall event, the soil bed was initially prepared by draining and drying under a large fan. Soil clods were then broken to 2-3 cm, and the surface was graded by hand or with a rake to achieve the desired slope. Impervious elements were installed as prescribed. Three layers of 17 mesh 0.011 aluminum screen were suspended above and parallel to the soil surface to help prevent soil crusting and sealing and maintain infiltration on the pervious portions of the soil box. A prewetting rain of 60 minutes at 10 mm/hr was applied and the soil box was then left to drain and dry for 24 hours.

Runoff samples were collected every 2 minutes during rainfall simulation and immediately weighed to the nearest 0.01g. These weights were adjusted to account for the tare weight of each container. Then 10 – 20 mL of saturated alum solution ($AlK(SO_4)_2$) was added to each sample to flocculate suspended sediment and allowed to settle at room temperature for 12-18 hours.

After settling, the runoff water was poured out. Sediment was transferred to 1-L bottles and placed in ovens at 105°C for at least 24 hours, or until dry. Dry weights were recorded to the 0.01g. Runoff volumes were determined for each sample according to (1), and runoff and sedimentation rates were determined for each sample interval according to (2) and (3), respectively.

(1) Runoff Sample Volume = (Sample Wt – Container Tare Wt – Sediment Wt) × 1mL/g

(2) Runoff Rate = Runoff Volume / Sample Duration

(3) Sediment Rate = Sediment Mass / Sample Duration

Two-dimensional slope rainfall simulation trials

In the first trial set of rainfall simulations, impervious elements were installed on a 4 × 4-m soil box having a two-dimensional slope and a soil depth of 5-8 cm (Figure 1).

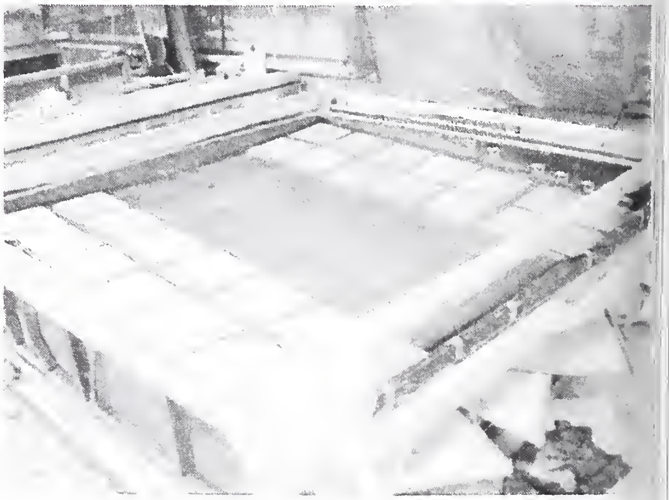

Figure 3. Two-dimensional slope soil box with impervious elements.

The soil bed had 4% side slopes and a 3% channel grade. Side slopes were 1.5 meters in length and 4 meters in width, and the main channel was 4 meters in length and 1 meter in width. The lower edge was connected to a flume for sample collection. Subsurface drainage was allowed to flow freely from the box through drainage tubes installed in a sand layer underlying the soil bed. This drainage was not collected. Rainfall simulations were performed on impervious configurations representing 0% impervious cover, and 35% impervious cover at the periphery and adjacent to the main channel. Twenty-eight impervious elements each representing 1.25% of total area were used. The 35% impervious configurations are shown in Figures 2 and 3.

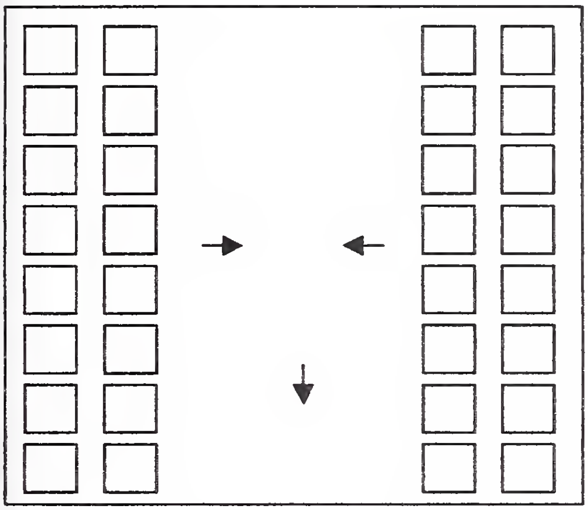

Figure 2. Configuration of impervious elements within 4×4-m soil box, representing peripheral development in two-dimensional slope trials. Arrows indicate slope direction.

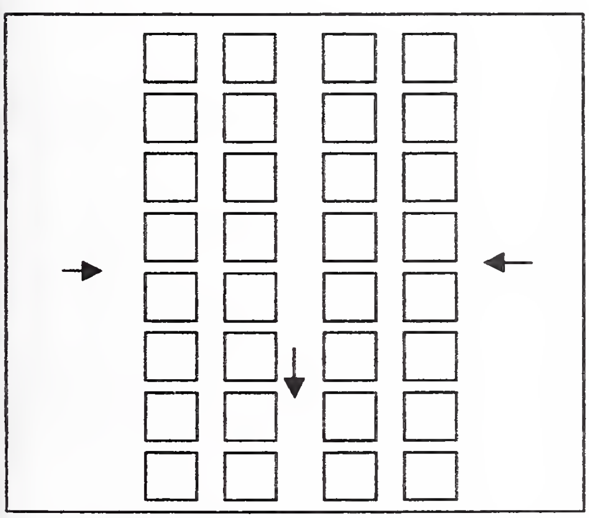

Figure 3. Configuration of impervious elements within 4×4-m soil box, representing channel development in two-dimensional slope trials. Arrows indicate slope direction.

Impervious elements were five-sided boxes constructed of sheet metal and installed by pressing the open edges of the box into the soil so that 4 cm of each edge was below the soil surface, and 4 cm remained above the soil surface. This simulated a rooftop-type impervious surface.

A rainfall sequence of 60 minutes at 20 mm/hr, followed by 10 minutes at 30 mm/hr, and then 10 minutes at 40 mm/hr was used for each rainfall simulation. For the 0% impervious treatment, the 40 mm/hr intensity segment was carried out for 40 minutes, to account for the longer time to reach steady state discharge. Rainfall interval times were based upon observed time to achieve steady state.

One-dimensional slope rainfall simulation trials

In the second trial set of rainfall simulations, impervious elements were installed on a divided 5×1.6-m soil box having a uniform lengthwise 5% slope, such that each impervious treatment was applied to a 5×0.6-m area (Figure 4).

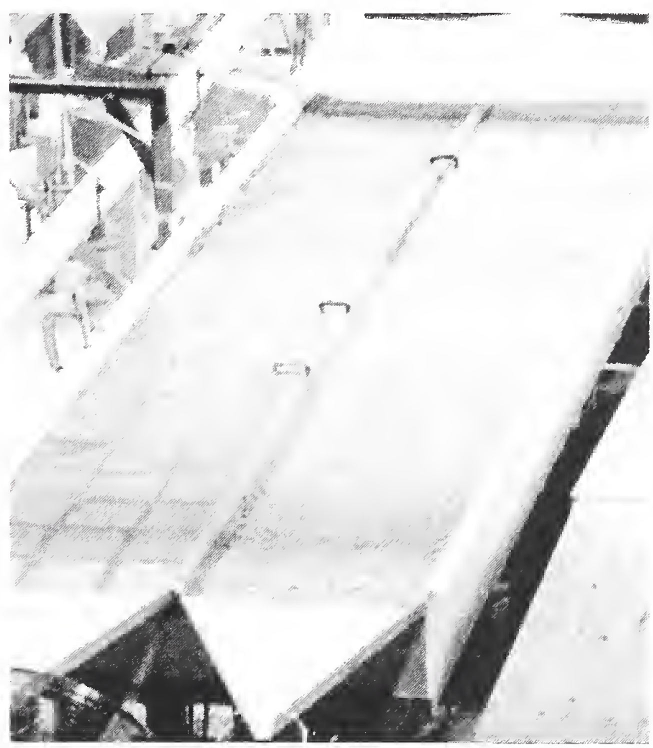

Figure 4. One-dimensional slope soil box with impervious elements.

Runoff samples were again collected through flumes at the plot ends, and water drained freely into an underlying sand layer and out of the box through drainage tubes.

Rainfall simulations were performed on 40% impervious configurations simulating urban development at the periphery and adjacent to the main channel, along a one dimensional watershed section, as shown in Figure 5.

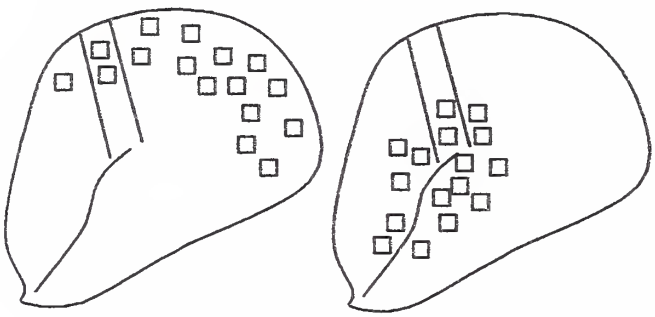

Figure 5. One-dimensional watershed flow path represented in rainfall simulation trials.

Impervious elements consisted of unglazed residential clay tile, with dimensions 20×20×0.6 cm. Because a raised impervious element would block flow in one-dimension, these were installed flush with the soil surface, representative of a transportation surface. Silicone caulk was used to seal between tiles, so that the upper 40% of the soil box was entirely impervious in one case, and the lower 40% of the box was entirely impervious in the other case.

A rainfall sequence of 50 minutes at 25 mm/hr followed by 16 minutes at 75 mm/hr was used. Rainfall interval times were based upon observed time to achieve steady state.

Results and Discussion

Two-dimensional slope rainfall simulation trials

Hydrographs from two-dimensional slope rainfall simulations are shown in Figure 6. Onset of runoff was significantly earlier where impervious elements were present. Onset of runoff was slightly earlier where impervious elements existed at the periphery, verses adjacent to the channel. However, this difference is more likely attributed to subtle differences in the soil surface shape than to treatment differences. Sediment loss rates (Figure 7) were similar between impervious treatments. As for runoff, sediment loss was lower and delayed in the 0% impervious treatment, as compared to the impervious treatments.

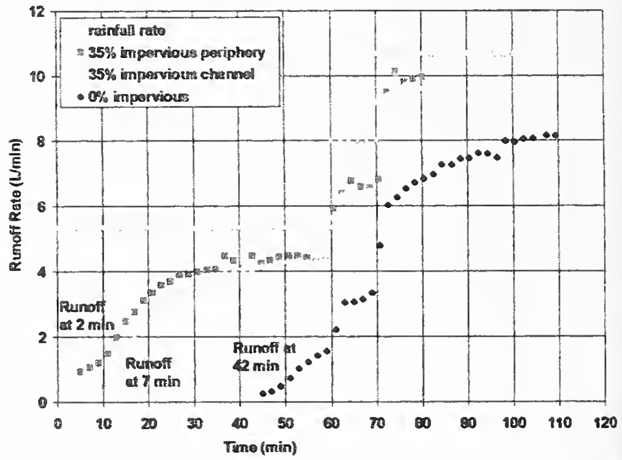

Figure 6. Rainfall and hydrographs from two-dimensional slope rainfall simulations with impervious surfaces.

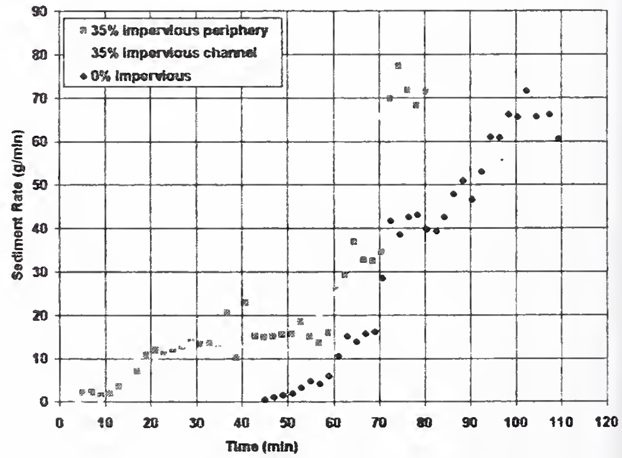

Figure 7. Sediment loss rates from two-dimensional slope rainfall simulations with impervious surfaces.

The soil surface shape was difficult to reproduce precisely for all runs, and it is believed that these inconsistencies were sufficient to impact runoff and sediment losses. In addition, treatments were geometrically very similar, since differences were implemented on the relatively short side slopes of the soil box. These difficulties led to the development of the one-dimensional flow system.

One-dimensional slope rainfall simulation trials

Hydrographs and sediment losses from the one-dimensional slope simulations are shown in Figures 8 and 9, respectively.

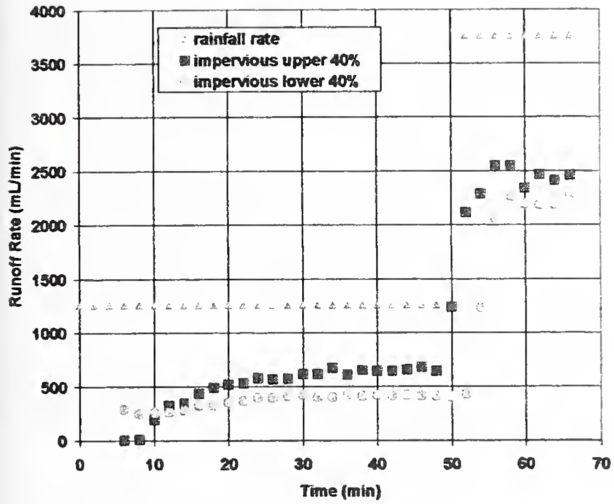

Figure 8. Rainfall and hydrographs from one-dimensional slope rainfall simulations with impervious surfaces.

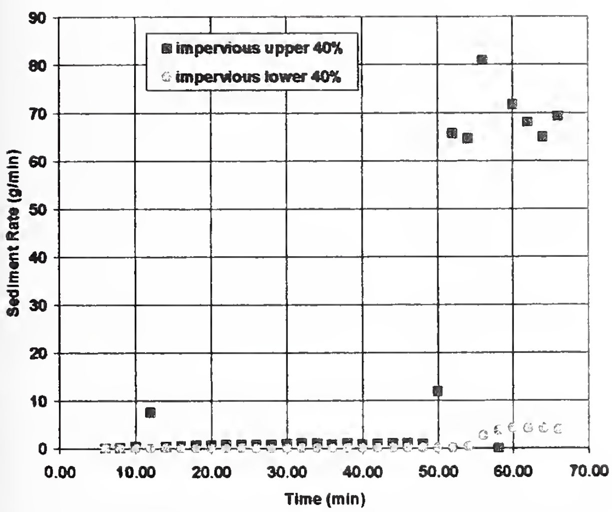

Figure 9. Sediment loss rates from one-dimensional slope rainfall simulations with impervious surfaces.

Impervious treatments yielded differences in both hydrograph and sediment losses, with the most clear differences in sediment losses. The upper impervious treatment yielded initially less runoff, as runoff generated by the impervious surface at the top of the slope infiltrated into the pervious soil surface below. However, this pervious zone became more quickly saturated than its upslope counterpart, due to this run-on. As a result, the upper impervious treatment yielded generally higher runoff and sediment after an initial wetting period.

This one-dimensional slope rainfall simulation approach appears to be sensitive to impervious treatments. However, impervious / pervious transition points were prone to scouring and undercutting at higher rainfall intensities. In order to reduce variability due to inconsistencies in these transitions, a modular segmented soil box system is proposed.

Modular soil box design

A modular soil box design is proposed for use in laboratory urbanization rainfall simulations (Figure 10).

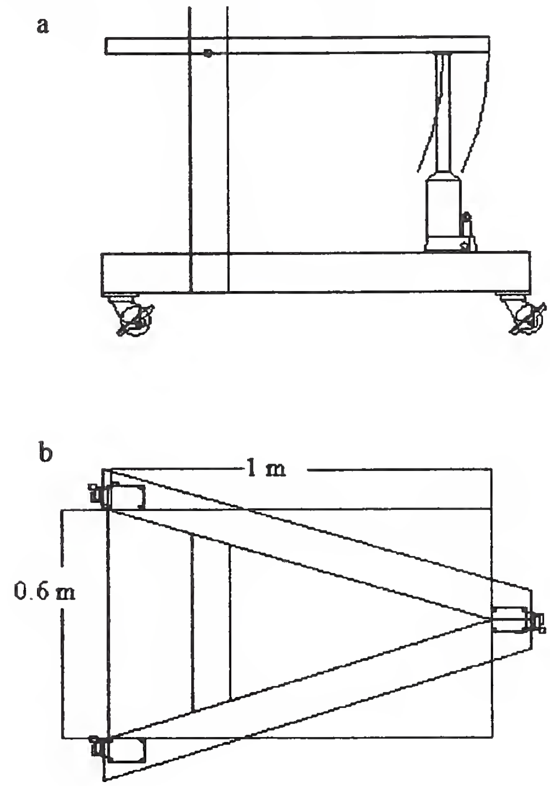

Figure 10. Single modular soil box segment side view (a) and top view (b).

Each soil box has a length of 1 m and is 0.6 m wide. Soil depth is 20 cm. Runoff and sediment will flow from upslope soil boxes into downslope soil boxes through short baffled flumes. Current designs allow up to 8 soil boxes to operate in series, for a maximum slope length of 8 meters. Boxes can be easily transported to and from growth facilities in order to develop different vegetative covers and / or turfs, and can be implemented with a variety of pavers or infiltration bed materials, without the transition concerns associated with the single box design. This modular design will allow researchers to construct specific sequences of land treatments along a one dimensional flow path. This system will be used to evaluate the effectiveness of specific impervious configurations as urban best-management practices, turf best management practices, and to guide the field watershed experiments described by Bonta et al. 2003.

Conclusions

Laboratory rainfall simulation methods can be used to detect hydrologic and erosional differences between impervious surface treatments. A modular soil box system design was developed in order to minimize inconsistencies in soil box preparation and impervious / pervious transitions, while maximizing treatment capabilities and slope length. This system will be used alongside a field watershed study to evaluate the hydrologic, erosional, and water quality impacts of impervious configurations and potential urban best management practices.

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U.S. Geological Survey Research on Surrogate Measurements for Suspended Sediment

John R. Gray, Theodore S. Melis, Eduardo Patiño, Matthew C. Larsen, David J. Topping, Patrick P. Rasmussen, Carlos Figueroa-Alamo

Abstract

The U.S. Geological Survey is evaluating potentially useful surrogate instruments and methods for inferring the physical characteristics of suspended sediments. Instruments operating on bulk acoustic, bulk and digital optic, laser, and pressure-differential technologies are being tested in riverine and laboratory settings for their usefulness to Federal agencies toward providing quantifiably reliable information on bed-material and bed-topography characteristics, and on concentrations, size distributions and transport rates of sediments in suspension and as bedload. The efficacy of four suspended-sediment surrogate technologies has been demonstrated to varying degrees of success in Kansas, Florida, Arizona, and Puerto Rico.

Keywords: fluvial sediment, turbidity, suspended sediment, monitoring, sediment surrogate

Introduction

A two-thirds decline in the amount of daily sediment data collected by the U.S. Geological Survey (USGS) since 1980 has occurred concomitant with a substantial increase in sediment-data needs and availability of potentially useful but largely untested sediment-surrogate monitoring technologies. Additionally, the Nation lacks nationally accepted standards for the collection or use of data derived from data-collection technologies other than those described by Edwards and Glysson (1999). These factors were instrumental in development of a recommendation by the Federal Interagency Workshop on Turbidity and Other Sediment Surrogates, April 30-May 2, 2002 (Gray and Glysson 2003) to form a Sediment Monitoring Instrument and Analysis Research Program.

The USGS continues to evaluate instruments and methods that show promise for providing reliable data on selected fluvial-sedimentary characteristics in riverine and laboratory settings, on bed-material and bed-topography characteristics, and on concentrations, size distributions and transport rates of sediments in suspension and as bedload (Gray 2002). This paper provides some examples of USGS research using bulk optical (turbidity), acoustic, laser, and pressure-differential technologies to infer selected characteristics of suspended sediments (Gray et al. 2002, Gray et al. 2003).

Turbidity Data as Suspended-Sediment Surrogates in Kansas

Sensors that measure the bulk optical properties of water, including turbidity and optical backscatter, have been used to provide automated, continuous time series of suspended-sediment concentrations (SSC) in marine and estuarine studies, and show promise for providing automated continuous time series of SSC and fluxes in rivers (Schoellhamer 2001). Continuous, in-situ measurements of turbidity to estimate SSC have been made at a stream monitoring site at the Kansas River at DeSoto, Kansas, since 1999.

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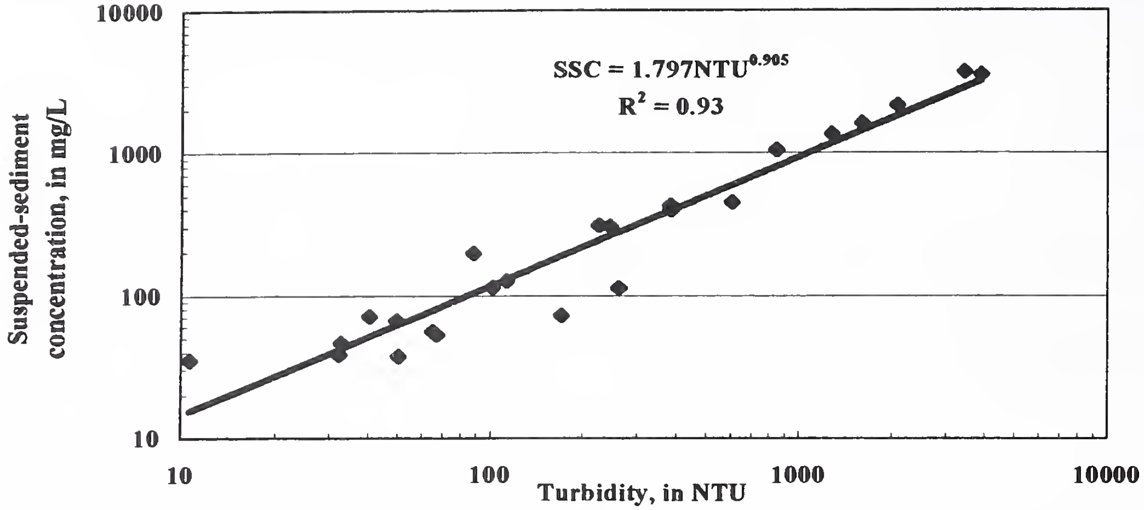

Figure 1. Comparison of field turbidity and suspended-sediment concentrations for the Kansas River at DeSoto, Kansas, 1999 through 2002.

Continuous turbidity measurements have been shown to provide reliable estimates of SSC with a quantifiable uncertainty. Simple linear regression analysis explained in Christensen and others (2000) was used to develop a site-specific model using turbidity to estimate SSC (Figure 1). The model explains about 93 percent of the variance in SSC. Continuous suspended-sediment discharge estimates from the model are available on-line (U.S. Geological Survey 2002). The advantages of continuous regression estimates using continuous turbidity measurements over discrete sample collection are that continuous estimates represent all flow conditions regardless of magnitude or duration, and sediment-discharge estimates are obtained essentially continuously at the interval in which water discharges are recorded.

Acoustic Data as Suspended-Sediment Surrogates in South Florida

Use of acoustic instruments worldwide for the measurement of stream velocities has increased substantially over the last two decades. These instruments are capable of providing information on acoustic return signal strength, which in turn has been shown in some settings to be useful as a surrogate parameter for estimating SSC and fluxes (Gartner and Cheng 2001). Two main types of acoustic instruments have been used extensively in the United States: the acoustic velocity meter (AVM), and the newer acoustic Doppler velocity meter (ADVM). The AVM system provides information on automatic gain control (AGC), an index of the acoustic signal strength recorded by the instrument as the acoustic pulse travels across a stream. The ADVM system provides information on acoustic backscatter strength (ABS), an index of the strength of return acoustic signals recorded by the instrument. Both AGC and ABS values increase with corresponding increases in the concentration of suspended material. SSC is then computed based on site-specific relations established between measured SSC values and information provided by the acoustic instrument.

Data from AVM and ADVM systems were collected in the L-4 Canal in Broward County, Florida, and the North Fork of the St. Lucie River at Stuart, Florida (Byrne and Patiño 2001). In addition to the acoustic instruments, water-quality sensors were installed at both sites to record specific conductance (or salinity) and temperature data. These data were used to monitor the potential effects that density changes could have on the AGC/ABS to SSC relations.

Results shown in Figure 2 suggest that this technique is feasible for estimating SSC in south Florida streams and other streams with similar flow and sediment-transport characteristics. Additional research is progressing on the effects of changes in the physical composition of suspended sediments, including the percent organic material, and the effect that a varying particle-size distribution may have on the established acoustic-SSC relations.

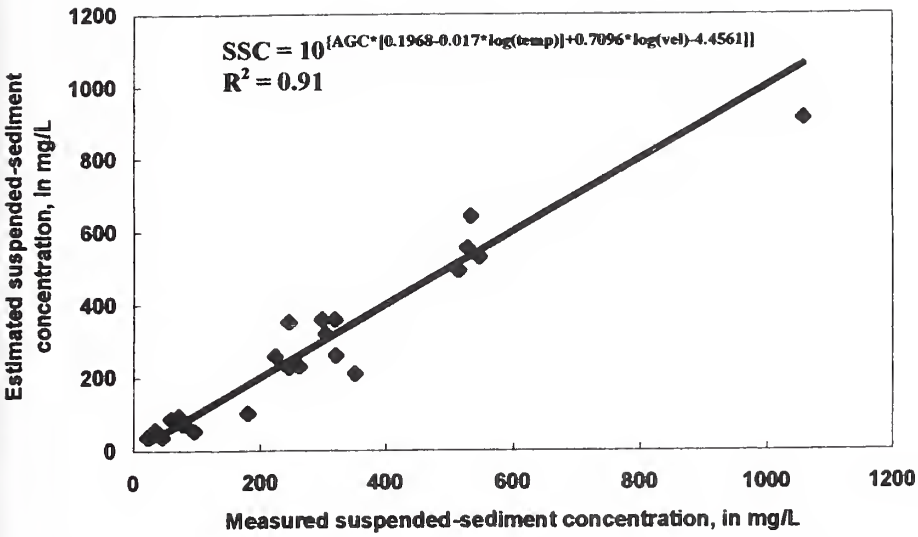

Figure 2. Comparison of estimated and measured suspended-sediment concentrations for the L-4 Canal site, Broward County, Florida.

Laser Data as Suspended-Sediment Concentration and Particle-Size Distribution Surrogates in Arizona

Laser diffraction grain-size analysis, a technique pioneered in the 1970's, is predicated on the concept that light impinging on a particle is either absorbed by the particle or is diffracted around the particle. The diffracted rays appear in a small-angle region. The Laser In-Situ Scattering and Transmissometry (LISST) technology measures the small-angle diffraction of a laser and inverts the signal to infer the in- situ particle-size distribution of the material being measured. Summing the volume of sediment in each particle-size class enables calculation of volumetric SSC (Agrawal and Pottsmith 2001).

Laser sensors are currently being investigated as an alternative monitoring protocol for tracking reach-scale suspended-sediment supply in the Colorado River at Grand Canyon, Arizona, located 164 km downstream from Glen Canyon Dam. This approach provides continuous suspended-sediment transport data that may reduce uncertainty in estimates of the transport of sand and finer material. The LISST data reported here were collected using LISST-100-B manufactured by Sequoia Scientific, Inc. (Agrawal and Pottsmith 2001, Gartner and Cheng 2001, Gray et al. 2002). The LISST-100-B is designed to measure suspended particles over a size range of 1.3-250 μm. The standard sample path of this device is a cylindrical volume with a diameter of 6 mm and a length of 50 mm.

Initial point data collected at a fixed-depth, near-bank site were obtained averaging 16 measurements at 2-minute intervals during a 24-hour deployment on July 19, 2001. The 720 LISST-100-B point measurements shown in Figure 3 compare favorably with cross-sectional data obtained concurrent with some of the laser measurements by techniques described by Edwards and Glysson (1999). In addition to accurately tracking sand concentrations, the LISST-100-B also recorded the expected increase of variance in the concentration of sand-size particles with increasing flows, with peak values ranging up to 150 mg/L (Figure 3).

These initial results, coupled with subsequent testing, suggest that the LISST-100-B is suitable for providing SSC and particle-size data for the Colorado River at Grand Canyon, Arizona. A manually deployable version of the LISST technology is under development (Gray and others 2002).

Pressure-Differential Data as a Suspended-Sediment Concentration Surrogate in Puerto Rico

Estimation of suspended-sediment concentrations from fluid density computed from pressure measurements shows promise for monitoring highly sediment-laden

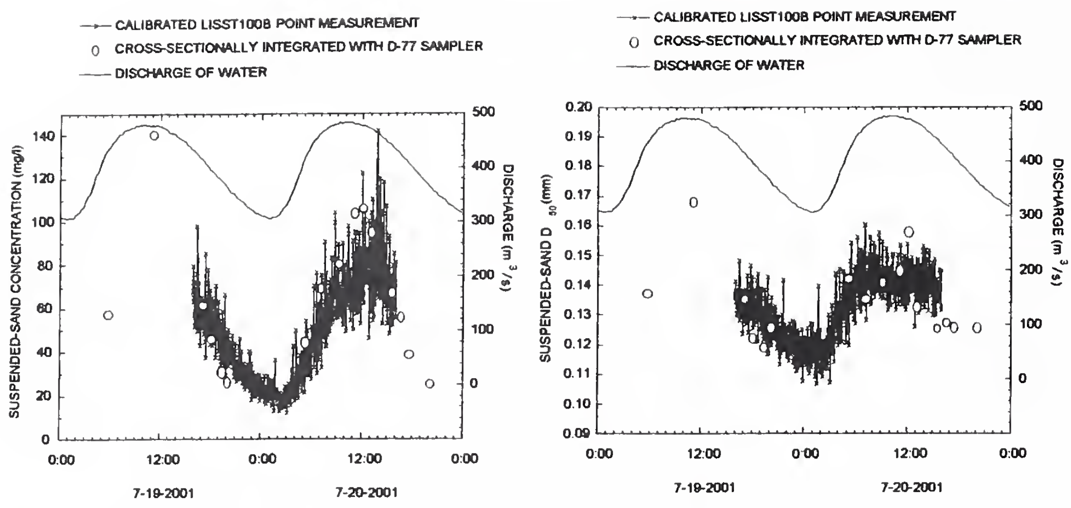

Figure 3. Comparison of sand concentrations and median grain sizes measured in the Colorado River at Grand Canyon, Arizona, using a LISST-100-B and a US D-77 bag sampler.

streamflows. Precision pressure-transducer measurements from vertically imposed orifices at different elevations are converted to density data by use of simultaneous equations. When corrected for water temperature, the density data are used to estimate sediment concentrations from a density-concentration relation (U.S. Geological Survey 1993). Thus, the device provides continuous (typically on 15-minute interval) sediment data that can be transmitted by satellite as stage and other data are currently transmitted. The cost savings and improved data quality can be substantial over those for traditional techniques.

An instrument for continuously and automatically measuring the density of a water-sediment mixture as a surrogate for SSC, referred to as a double bubbler precision differential-pressure measurement system by the manufacturer, was tested in Puerto Rico (Larsen et al. 2001). Continuous double bubbler instrument data were collected during October-December 1999 at a stream gaging station on the Río Caguitas, Puerto Rico. As of 2000, the maximum SSC measured at the site using techniques described by Edwards and Glysson (1999) was 17,700 mg/L, corresponding to a specific gravity of about 1.02, which also represents the signal-to-noise ratio.

The data collected during October-December 1999 at this site showed relatively poor agreement between discharge, SSC, and water density (Figure 4). The 1999 tests indicate that the double bubbler instrument values generally track substantial variations in SSC, but a large amount of signal noise remains.

This test of the double bubbler instrument showed the need for temperature compensation, and possibly the need to deploy the instrument at a site where the signal-to-noise ratio is substantially greater than 1.02. The double bubbler is being tested in Paria River, Arizona, where SSC in excess of 1×10^6 mg/L have been measured, yielding a signal-to-noise ratio of as much as 2. If adequate results can be achieved, increases in data accuracy and substantial reductions in costs of sediment monitoring programs for rivers carrying moderate-to-large SSC can be realized.

Summary

The USGS is evaluating surrogate technologies for estimating SSC and fluxes. Those based on bulk optic, acoustic, and laser technologies have been shown to be successful at selected test sites, although the robustness of these techniques must be more fully evaluated. The approach using the pressure-differential principle shows promise for use in highly concentrated streamflows.

Note

Use of trade or firm names in this report is for identification purposes only and does not constitute endorsement by the U.S. Government.

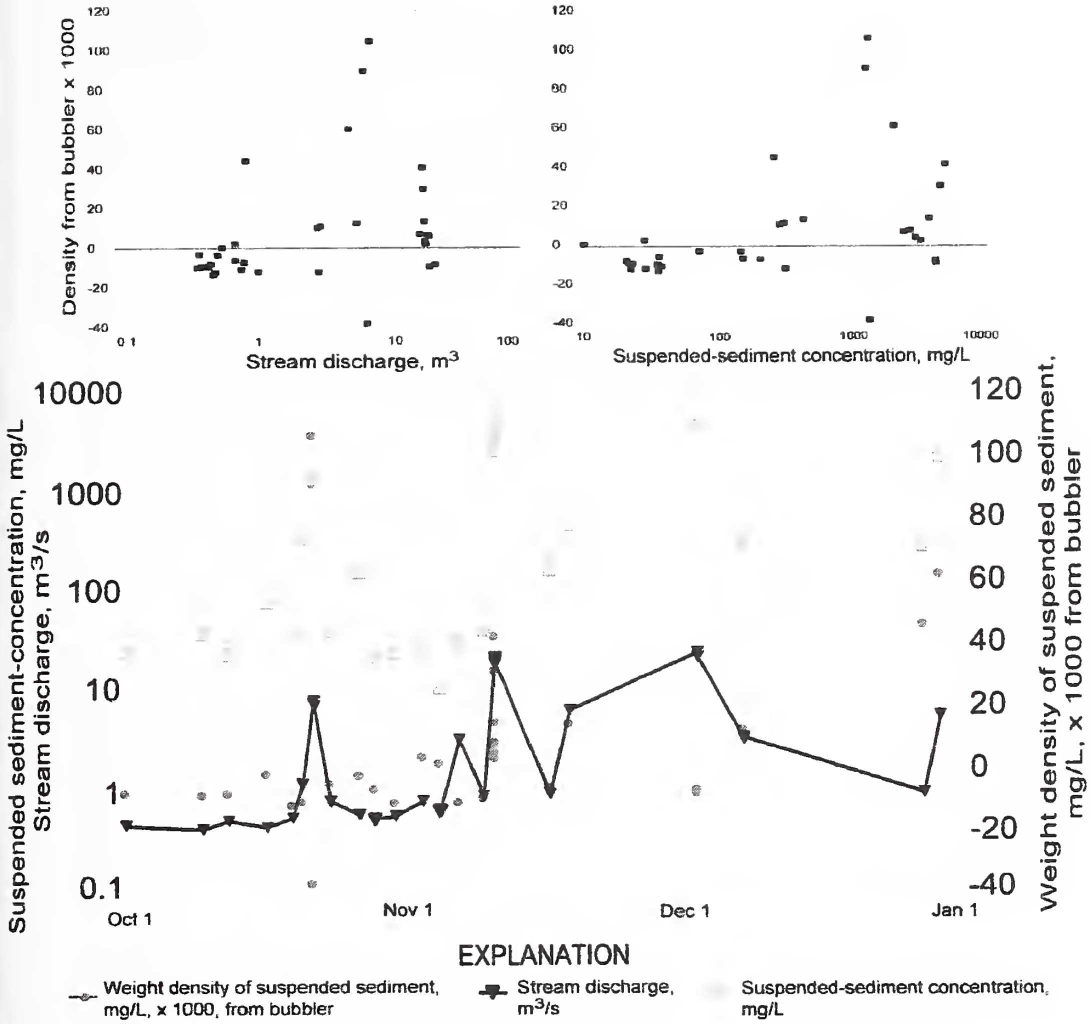

Figure 4. Scatter plots and time series of stream discharges, suspended-sediment concentrations, and weight density of suspended sediments and dissolved solids measured with a double bubbler, October 1, 1999 to January 1, 2000. Discharge and sediment data are instantaneous samples, and the double bubbler weight density value is a 30-minute mean of measurements made at 5-minute intervals.

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Title of Monitoring and Modelling Runoff in Semi-arid Areas from the Hillslope to the Watershed Scale

Richard Brazier, John Wainwright, Tony Parsons, D. Mark Powell, Roger Simanton, Howard Larsen

Abstract

Response of the landscape to intense rainfall events is a complex and poorly understood problem. An understanding of the spatial variability of runoff generated by such storms at the hillslope scale is a necessary goal if patterns of runoff and soil erosion are to be understood at the field and catchment scale also. In recent years, it has been recognised that linking these scales of runoff may provide an approach by which accurate predictions may be made at all scales from the small hillslope to the large catchment (Wainwright et al. 2001). Furthermore, by studying the way in which patterns of runoff vary with spatial scale a better understanding of sediment delivery problems and the dynamic connectivity of systems at a variety of scales can also be made.

To address the issue of scaling within runoff, a series of nested experiments was carried out to monitor the flux of runoff after intense, natural rainfall events at a range of scales at the Walnut Gulch Experimental Watershed in the semi-arid south western US. Data from these experiments were used to evaluate a distributed, dynamic, process-based model, previously shown to perform well at the plot scale on semi-arid shrubland (Parsons et al. 1996). To extend previous work, the model was applied to sites ranging in size from 2 m^2 up to 0.5 km^2 to investigate model response to changes in scale and to provide a means of linking predictions made at the hillslope scale with those made at the catchment scale. Results indicate that given high quality input data accurate predictions can be made at a range of hillslope lengths. Limitations focus upon high data requirements, though remote sensing techniques are being developed to reduce time spent on data capture of surface condition parameters. Scaling of erosion and sediment transport is being investigated also using a unified approach that uses characteristics of transport distances to provide an inherent scaling factor. Initial results of the runoff modelling are presented as a basis for future development of the erosion model.

Keywords: semi-arid, runoff, erosion, scaling

Introduction

Understanding the response of hillslopes to extreme rainfall events is a complex problem. To date, numerous monitoring and modeling strategies have been employed in an effort to not only characterize hillslope runoff and erosion as a response to rainfall, but also to extend lessons learnt at the hillslope scale to the wider environment at the sub-watershed or watershed scale. Examples in the United States date back to the work of Cook (1936) who identified the chief controlling variables of soil erosion by water, through to Wischmeier and Smith (1965) who developed the Universal Soil Loss Equation (USLE)

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and more recently Lane and Nearing (1989) who present a more process-based understanding of rainfall-runoff and soil erosion in the framework of the Water Erosion Prediction Project (WEPP). Such work has been instrumental in furthering the understanding of runoff and erosion processes and crucially has led to the development of tools which can guide policymakers and farm managers alike as to the effects of cultivation or grazing upon the natural response of the environment to rainfall events.

To further understanding in this field, it is suggested herein that the problem of up-scaling assessments of runoff and erosion from the hillslope to the watershed scale is addressed. Many of the existing predictive models rely upon empirical observations made at the hillslope or plot scale (from the USLE plots for instance). These data sets, though undoubtedly a unique resource and clearly very useful in their time, tend to rely upon uniform plot dimensions - typically 22 x 4 m in the case of the USLE plots (Wischmeier 1976), which do not describe hillslope responses over a range of scales. Therefore, the following paper presents results from a nested monitoring and modeling scheme which seeks to overcome this spatial limitation of existing data sets (and models) by explicitly considering runoff (and in due course soil erosion) as a function of hillslope length on a range of sites from 2 m^2 to 1200 m^2 in size.

A spatially designed monitoring experiment, to complement the existing monitoring infrastructure at the USDA-ARS Walnut Gulch Experimental Watershed, Arizona, was developed and maintained for three monsoon seasons. The approach taken coupled field observations directly with model development in order to ensure that full evaluation of the model was possible, as called for by Brazier, (in press). The following paper describes preliminary results of the hydrological model performance against observed data from a range of scales.

Nested Monitoring Scheme

In order to observe water and sediment fluxes at the hillslope scale, four pairs of erosion plots were constructed within watershed 223, downstream of the Lucky Hills watersheds. Each of four large plots (Wise, Abbott, Laurel and Dud) were constructed alongside four small plots (Morecambe, Costello, Hardy and Pete) of equal size (2 m in length) on interrill areas. The large plots ranged in length from 4 m to 28 m and were installed to sample:

- Rainfall
- Event hydrograph
- Total flow
- Suspended sediment flux (1 minute intervals)
- Total soil loss
- Nutrient fluxes

In this manner, it was anticipated that detailed description of hillslope response to natural events could be made and comparisons drawn between plots and on both an inter- and intra-event basis. An example of observed results for a single event is included in Figure 1.

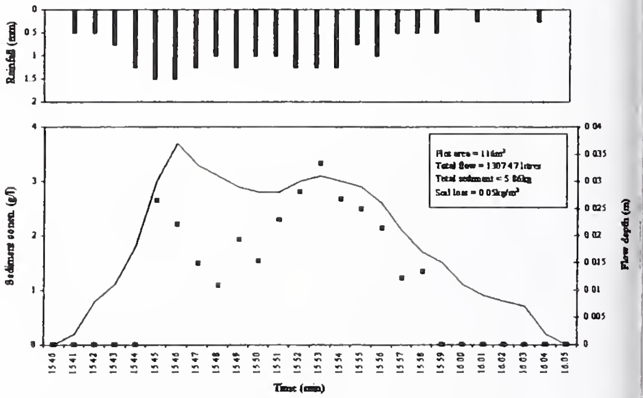

Figure 1. Rainfall, flow depth and suspended sediment concentration from the Abbott hillslope plot - 30/07/00.

Varying rates of sediment flux from all of the hillslope plots is shown in Figure 2. A clear relationship between soil erosion and plot length can be seen, with shortest plots (ca. 2 m) yielding highest fluxes per unit area and longest plots yielding lowest concentrations of sediment.

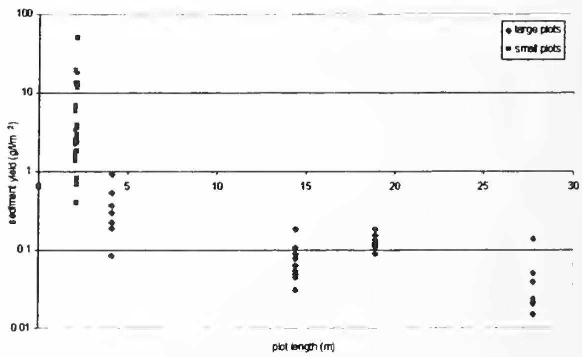

Figure 2. Observed interrill sediment flux as a function of hillslope length.

To supplement the hillslope monitoring and bridge the gap to the larger watershed scale monitoring conducted by the USDA-ARS at Walnut Gulch, a number of small watersheds, again nested within watershed 223 were also instrumented. These were the Cleese and Alan Bennett watersheds, both watersheds covering areas of approximately 1220 m^2 and five watersheds draining through the main channel of watershed 223 known as; 103, John, Paul, George and Ringo with areas of 35,065 m^2, 57,102 m^2, 285,692 m^2, 377,787 m^2 and 468,691 m^2 respectively. For the purpose of this paper, results from the Cleese watershed are presented alongside the hillslope observations, results from the larger watersheds are detailed in Brazier et al. (2003). Within this watershed, similar parameters to the hillslope monitoring schemes were observed with the notable addition of a bedload monitoring trap to provide information on the real time fluxes of coarser, bed material. Surface cover maps of the four large hillslope plots and the Cleese watershed are shown below (Figures 3 and 4.) to illustrate variation in pavement cover as surveyed during the pre-monsoon period.

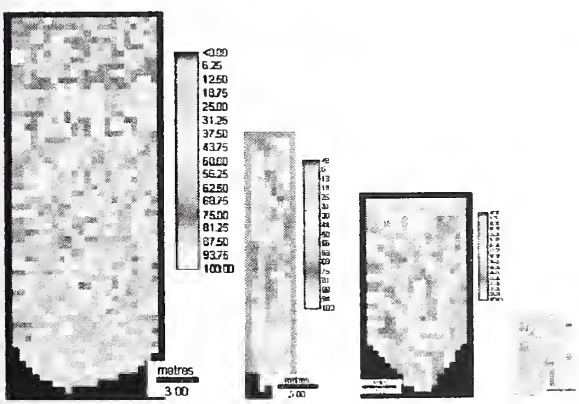

Figure 3. Desert pavement cover for hillslope plots; ranges from 100% (dark red) to 0% (blue).

Figure 4. Desert pavement cover for Cleese watershed; ranges from 100% (dark red) to 0% (blue).

Modeling rainfall and runoff response

As detailed above, the nested monitoring scheme was specifically designed to educate model development; this made it possible to conduct a direct evaluation of model performance for sites with adequate observed data to provide confidence in model results. The model developed the work of Scoging (1992) who used a distributed approach to predict the spatial pattern of overland flow hydraulics (Parsons et al., 1997). Overland flow is first generated using a modified Green-Ampt equation:

$$f_t = a + bt^{-1} \qquad (1)$$

where f_t is the infiltration rate (mm min^{-1}), a is the final infiltration rate, b is the rate of decline of infiltration rate to its final value and t is time (min). The following continuity equation (2) is then used in combination with rating equation (3) to distribute the flow as a 1-D kinematic wave:

$$\frac{\partial q}{\partial x} + \frac{\partial d}{\partial t} = e_x \qquad (2)$$

$$q = \alpha d^m \qquad (3)$$

where q is overland flow discharge per unit width (cm^2 s^{-1}) , x is distance (cm), d is depth of flow (cm), e_x is rainfall excess (cm s^{-1}), with α and m being the empirical terms of rating equation (3). The Darcy-Weisbach friction factor f is used to compute flow velocity (v) which, combined with flow depth gives q:

$$v = \sqrt{\frac{8gds}{f}} \qquad (4)$$

where g is acceleration due to gravity (cm s^{-2}) and s is the surface slope (m m^{-1}). Water will then move from cell to cell along one of the four cardinal directions within a finite difference grid controlled by the greatest difference in height between cells.

In order to build spatial representation of infiltration rates into the model the driving parameters of equation (1), (a and b) were related to pavement cover (%P) in each cell by the following empirical equations (after Abrahams and Parsons, 1991a):

$$a = 1.628 - 0.014\%P \quad (5)$$

$$b = 0.785 + 0.021\%P \quad (6)$$

Also, the friction factor (f) was related to the depth parameter (d) (Abrahams and Parsons, 1991b) by the following equation:

$$f = 14.46 - 17.35d \quad (7)$$

The model was then applied without calibration, to the four large hillslopes hillslope plots. The initial results of which are described below.

Results

Hillslope plot scale

As an initial test of model performance, flow routing at the hillslope scale was output to verify that flow direction corresponded well with expected flow patterns from the high resolution (0.5 m) DEM. Figure 5. illustrates flowpaths for the event dated 30/07/2000 on the Abbott plot which coincides with the observed data illustrated in Figure 1 and the hydrograph predictions made in Figure 6.

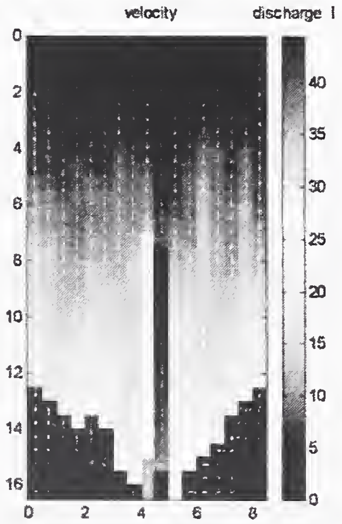

Figure 4. Predicted flowpaths from the Abbott hillslope plot for the 30/07/2000 event.

Results from 2 single events for the Abbott hillslope plots are shown below in Figures 5 and 6. Observed hydrographs at the outlet and total flow data are represented incorporating associated RMS error (see Table 1).

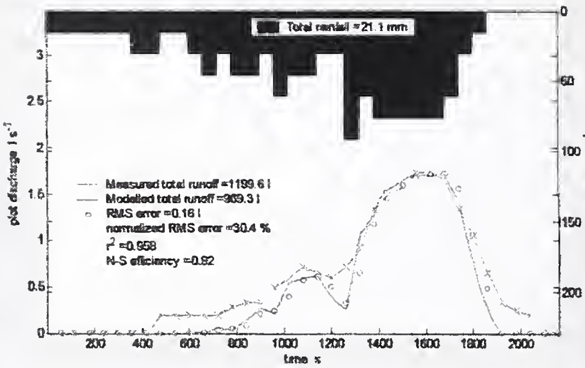

Figure 6. Observed and predicted hydrographs from the Abbott hillslope plot 30/07/2000.

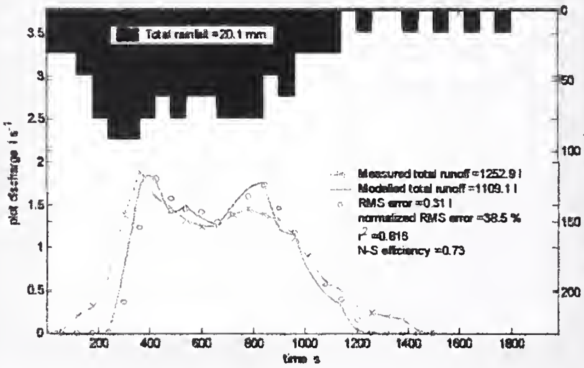

Figure 7. Observed and predicted hydrographs from the Abbott hillslope plot 10/08/2000.

In general results from these two events are encouraging with high r^2 values indicating a good level of agreement between observed and predicted hydrograph form and timing. However, these events are very similar in nature, both being in the region of 20 mm rainfall total with rainfall intensities only reaching 100 mm hr^{-1}, thus it might be expected that the model would perform equally well for each event. Clearly, further model simulations need to be performed on the range of hillslopes for a full range of storm characteristics before definite conclusions about model performance at different scales can be drawn.

Table 1. Model performance statistics for simulations on the Abbott Hillslope plot: 30/07/2000 and 10/08/2000.

	Abbot plot Event 30/07/00	Abbott plot Event 10/08/00
Observed total runoff (l)	1199.6	1252.9
Predicted total runoff (l)	969.3	1109.1
RMS error	0.16	0.31
Normalized RMS error	30.4%	38.5%
r^2 value	0.958	0.816
N-S Efficiency	0.92	0.73

Nonetheless, it is encouraging to note that the bi-modal characteristics of both the observed hydrographs are simulated reasonably well and the timing of runoff peaks is also simulated well, despite the disparity between the magnitudes which are particularly noticeable for the 10/08/2000 event. Also noteworthy are the levels of error associated with predictions for the two events. In both cases the normalized RMS values are in excess of 30% indicating that significant error is associated with model predictions. Furthermore, here consideration of error in observations has not been made nor has it been incorporated in goodness of fit tests. Thus, these results must be interpreted as preliminary and will undoubtedly become more meaningful with further effort to quantify both error associated with the observations and uncertainty surrounding model predictions.

Conclusions

Variation in observed results from the range of hillslope lengths indicates that hillslope length plays an important role in controlling flow and sediment flux from the hillslope as a whole. Recourse to data sets based on single length plots can therefore not be made if the scientific goal is to learn about the influence that hillslope length (to the channel for instance) plays in semi-arid environments. For future studies, it is suggested that plot length is incorporated into the list of variables that are varied between monitoring sites in order to more fully describe the change in both water and sediment fluxes as upscaling from the (small) hillslope to the watershed scale is made. Furthermore, it is shown that nesting monitoring sites within pre-existing frameworks (as at Walnut Gulch) provides a straightforward means of bridging the gap between scales of observation which can educate model development.

Model results indicate that the model performs reasonably well in predicting the event hydrographs of 30/07/2000 and 10/08/2000 on the Abbott hillslope plot. However, errors associated with observed data are not inconsiderable and must be taken into account when considering the validity of model results. It is noted that no observed data will be error free, (though hydrographs in particular are often treated as such), thus it is advisable to fit predictions to data sets which explicitly demonstrate this error, to provide a more meaningful assessment of model performance.

Future work will build upon both the data collection and modeling work presented here to construct a soil erosion component to the model that also considers the effect of slope length upon transport distance of individual particles.

Acknowledgments

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The Soil and Water Quality Link – Using Composted Products for Effective Stormwater Management

Rodney W. Tyler

Abstract

The link between water quality and soil quality is finally becoming clearer. When native soils are plentiful and left undisturbed, water quality increases. However, the urbanization of American landscapes has caused significant disruption to this native layer of soil and as a result, Phase II NPDES has been enacted. University research, private research, field demonstrations, and now commercial use of compost for erosion and sediment control show it works as well or much better than most BMPs available today, yet it continues to suffer an identity crises. Here are the facts after reviewing some of the commonly referenced papers that have been available over the last several years. Since compost mimics the layers of native soils, it should be considered as an option in the new Toolbox of the contractor to stay in compliance with Phase II.

Keywords: compost, water quality, erosion control, stormwater management

Introduction

According to the U.S. Department of Agriculture, the United States loses more than 2 billion tons of topsoil each year through erosion (U.S. EPA 1997). The link between water quality, sediment control, erosion, and eventually water quality has widespread impacts on our sustainable future. Stormwater runoff pollution is 80% of water quality violations in many states and is the first line of defense when it comes to creating proactive, sustainable cultures (ESTS 2000).

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Worldwide, estimates indicate erosion may cost us as much as $400 billion annually. There are literally hundreds of products to control sediment and erosion. Very few commercial products involve the use of compost and the composting industry is suffering from an awareness problem relating to the benefits of compost in environmental applications. One of the main identity problems is credible sources that claim compost works. This article focuses on the review of these papers indicating the effectiveness of the use of compost.

Phase II NPDES became effective in March of 2003 and promises to deliver some very strong regulations which should favor the use of compost because of its proven effectiveness and local availability. Compost is available in every major city in the U.S. Phase II has several key points, which should be noted - most importantly is that the requirement for a stormwater management plan drops from five acres to one acre. This five-fold increase will immediately have an impact on sites that are inspected. Many permit issuers are saying at this point, they will not be giving out permits without a stormwater management plan up front. This may be easier to manage and could definitely hold up the permitting process.

Composters should be happy about Phase II because of the opportunity it holds for developing a new market in erosion and sediment control. Erosion prevention (keeping soil from moving off of slopes) is about 90-98% effective. Trying to control the mud and sediment once moving (sediment control) is normally less than 50% effective when using other commercially BMPs like silt fence. Therefore, compost blankets should become a leading tool, especially for challenging projects. The following are reviews of research from using compost over the last several years.

Reasons to Use Compost for Erosion Control as Blankets and Filter Berms

- Construction can run it over and it still works – and it is easy for them to fix with a shovel
- Re-use of material afterwards makes it twice as good – in landscaping or seeding activities
- It works better than standard BMPs like silt fence and straw bales
- Berms offer more actual filtration than coir rolls, silt fence or straw bales
- Compost is annually renewable
- Compost is 100% recycled & locally made
- Compost is all organic & all natural
- It helps create an annual market in all municipalities that generate compost of some kind
- Compost is critter friendly – aquatic wildlife can negotiate berms
- Compost is a biobased product while other common products are a petroleum based product
- Compost provides chemical, biological and physical filtration while others provide only physical filtration
- Compost is less expensive when construction, maintenance, removal and disposal costs are considered

W&H Pacific Demonstration Project Using Yard Debris Compost for Erosion Control

This report is considered by many to be the landmark paper on using compost for erosion control because it points out several items that are crucial to this developing marketplace. There are over 70 references and many projects (including many of my own field demos) have been tailored after this simple project design and report. Bill Stewart, as one of this project's investigators, has gone on to show many benefits of treating stormwater with compost (pelletized compost filters) as well as using compost for treating aerosols via biofilters.

Two main themes brought out in Stewarts' work include issues relating to vegetation establishment. One problem with vegetation establishment currently is that most construction site soils are heavily compacted. As such, they offer little means for water penetration and have normally high runoff rates (Figure 1). Compost applications, in the form of compost blankets, slow down water, allowing greater infiltrations. When seeding with the use of compost blankets, huge performance differences exist over any current leading method. The industry standard, hydroseeding, is sure to lose market share to compost seeding technologies because of these results. Compost offers moisture holding capacity, slow release nutrients, complete coverage of seed, and a depth of ¼" to 2" which hydroseeding cannot offer. Finally, the layers of seed that germinate in a 2" deep compost blanket that is seeded are tremendously beneficial because they offer a matting effect and can withstand harder and longer intensity rainfalls than the hydroseeding counterparts.

GRAPH 2
Surface Water Runoff Rate - Austrian Vineyard Data
Municipal Solid Waste Compost Application
30% Slope

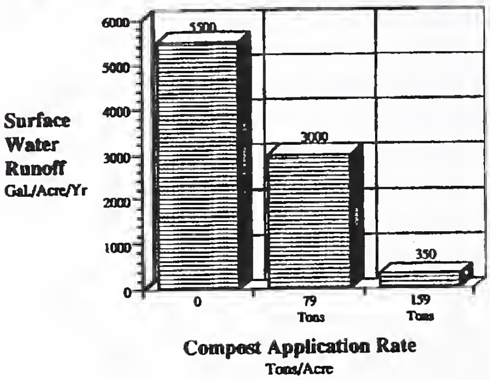

Figure 1. (Source: Stewart 1993).

Based on the work by W&H Pacific, they determined: "based on the results, all three yard debris composts are at least as effective as the conventional erosion control measures currently specified (i.e., silt fence). The erosion effectiveness of the composts, measured in terms of soil loss (suspended solids), was better than that measured from sediment fences…(Stewart 1993)."

Many people confuse TSS with turbidity. Turbidity is "a measurement of the clearness or transparency of water. In addition to soil particles, colloidal organic matter in particular will scatter or absorb light and thus prevent its transmission, resulting in increased turbidity. Turbidity is measured in NTU units. The turbidity of a clear lake will have a turbidity of 20 to 25 NTU (Stewart 1993)."

How do compost blankets stay on steep slopes? Just look at any material yard the next time it rains and you are sure to see the conical piles consistently perform in preventing and resisting erosion. The soil piles next to them become rilled and begin eroding after the very first rain event. This is due to the compost acting like a shingle roof on the slope. Think of compost as a wet bunch of paper towels overlapped on a slope, two or three layers deep. Because compost does not roll, it resists erosion. Soil is round and when it begins to roll downhill with the force of water behind it, the combination acts like a sandblaster to other soil in the way. The fibers of compost also have the ability to interlock with one another and this interlocking mechanism allows materials to hold slopes and some amount of directional flow of water. Even fine ground materials have this interlocking system, on a smaller scale.

The W&H Pacific study included three types of yard debris compost, coarse, medium and fine and also included leaf humus. These are the most common types of compost available today in most metropolitan areas, where urban wastes have been turned into valuable products via the compost process. Our company has repeated not only the types of products used in the initial study, but also basically the same type of demonstration set up in the field. Although we did not collect data, the visual results are obvious when comparing to other BMPs in the field that are properly installed.

As the Stewart work continued to study various key elements about the benefits of compost use for erosion, it indicates a tremendous opportunity in the reducing of both suspended and settleable solids from entering waterways (Figure 2).

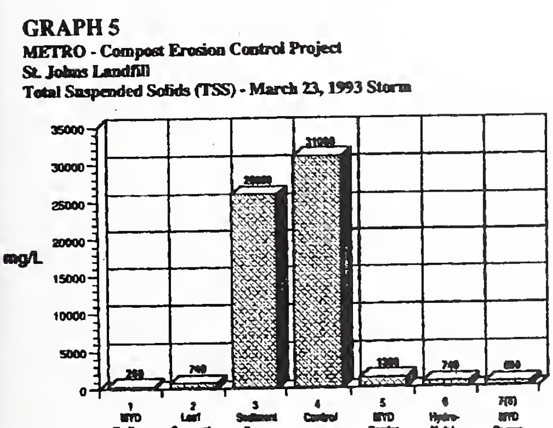

Figure 2. (Source: Stewart 1993).

Figure 2 shows the impact of control plots as well as sediment fence (silt fence). Note the comparison of any compost application, including blankets or berms (MYD barrier) is over 10 times as effective as using silt fence. This is what the new Phase II regulations will be targeting, so the use of compost as a tool is bound to become more popular.

Other chemical binding properties were noted by the Stewart (1993) study: "At a construction site, in addition to its erosion control benefit, a good quality compost is capable of binding and removing pollutants from storm water runoff including oil and grease, fuel from accidental spills, heavy metals, herbicides, pesticides, and other potentially hazardous substances associated with construction or pre-construction activities."

Concerns with movement of nutrients and heavy metals has not been widely documented with the use of compost for erosion control applications. Testing for this requires precise science, controlled conditions, and normally significant funding. Stewart placed a high importance on using mature compost. The majority of the scientific community agrees that mature compost helps prevent movement of some nutrients (nitrogen) because it is largely contained in the organic form. However, due to the amount of water passing through compost filter berms, there is more research needed in the area of nutrient transfer, leachability of nutrients and heavy metals. When compost is used in situ, or in soil, it has more complex binding relationships with the parent materials. When used as a compost blanket or filter berm, the bonding relationship can only occur at the compost soil interface because the material is not mixed into the parent soil.

"These data indicate that the composts tested do not release heavy metals significantly greater than that released by soils and, in fact, can result in a reduction in heavy metal runoff from soils which contain higher quantities of these elements. However, it is important to note that compost quality and background heavy metal quantities in the compost is a factor to be carefully evaluated (Stewart 1993)."

Quilceda-Allen Watershed Erosion Control Program: Water Quality Monitoring Report

Although the regs for the burying depth of silt fence varies, most states require some depth to be achieved. Because this practice is not inspected heavily (or often from the vehicle), installers are able to 'get away with' not trenching in the silt fence. Without the trench and weight on the bottom of the fence, the silt fence may simply allow sediment and water to run underneath. More recent studies indicate that the fine particles in some of the soils are finer than the openings on the sediment fence and are not affected at all. "When the bottom of the silt fence is properly buried, then the silt fencing acts as a water barrier, but the turbidity is not reduced. A mulch berm provides filtration as runoff passes through (Caine 2001)."

The project included compost berms along with coir rolls and other possible BMPs for controlling erosion. "When used in conjunction with appropriate ground cover, silt fencing is assumed to provide stormwater runoff with adequate turbidity treatment. Monitoring of water quality from actual construction sites, however, indicated that silt fencing did not provide adequate water quality treatment (Caine 2001)."

According the results from Snohomish County, berms also absorb water more than we originally thought, which may give them higher density when wet. "By blowing the compost to form the berm, the compost had a lot of pore space. Consequently, the berm absorbed a volume of water equal to approximately 30% of the volume of the berm. It took approximately 17-26 minutes after water flowed onto the berm for water to percolate through the berm and the water was released at a very slow rate and at multiple locations along the length of the berm (Caine 2001)."

In the data, comparisons show that a mulch berm reduced turbidity compared to silt fence and coir fiber rolls used as BMPs. In fact, the mulch berm reduced the turbidity to 33% of the entering level while the silt fence and coir roll remained at 100% (Caine 2001).

Settling of water is a leading mechanism for getting sediment out of water and sediment ponds or detention/retention basins are leading recommendations among engineers when all other items fail. A sediment pond allows water to settle out over a long term prior to discharge back into the waterway being protected. However, there are problems with this design as well because in severe events (remember, designs for capacity do not include severe events) the overflow of these ponds occurs sooner and the water goes directly into the protected watershed. In some cases, the detention area or pond allows the water to heat and the extra temperatures play havoc with aquatic life downstream due to temperature increases. "Water released from detention ponds, however, exceeds existing allowable thermal limits between May and October (Caine 2001)."

The Snohomish county project had a very clear purpose for the endangered species act and saving the Salmon that were endangered from sediment… "The purpose of this project is to reduce the sediment input into streams and wetlands in the Quilceda-Allen watershed, thereby improving the water quality in the streams and decreasing sediment clogging of fish spawning gravels (Caine 2001)."

Compost: New Applications for an Age-Old Technology - U.S. EPA

Another landmark publication was produced in 1997 by the U.S. EPA. The familiar green publication is perhaps best known for putting compost on the map for all of the remedial properties compost provides in various application technologies. The publication is available on the EPA web site and has been widely distributed in the U.S.

According to U.S. EPA (1997), "Depending on the length and height of the slope, a 2-3 inch layer of mature compost, screened to ½ to ¾ of an inch and placed directly on top of the soil, has shown to control erosion. On steep slopes, berms of compost at the top and bottom can be used to slow down the velocity of water and provide additional protection to the receiving waters."

Our company has verified these results in the field in at least ten different states where we have worked on projects. Most notably, even when the slopes are not covered with compost blankets, compost filter berms still reduce overall erosion because they allow run-on water to be converted from rills back into sheet flow down the slopes and the overall velocity is reduced as well.

When used as a filter berm, compost is 'nature's coffee filter,' leaving the residue from stormwater behind in a tell tale thin film that is easy to see on the berm surface after water drains through or subsides. Many people confuse filtration, bioremediation and biofiltration. Chemicals trapped by the coffee filter mechanism of the berms are often remediated. "Biofiltration implies physically separating particles based on their size. Bioremediation, by contrast, implies biological change as contaminants or pollutants are metabolized by microorganisms and broken down into harmless, less stable constituents, such as carbon dioxide, water and salt (U.S. EPA 1997)." Depending on the concentration, there is very good chance that compost can bioremediate some of these compounds within the berm *while it is filtering out more sediment.*

There is little research on this particular topic, however, EPA has recognized that the need for 'prescription' composts that are specially made to remediated particular spills or situations are definitely a possible common product in the future. "The metal binding capacity of compost can be improved by the addition of inorganic materials. For example, the addition of soluble iron and phosphate salts to compost increases lead immobilization as a result of forming complex lead-iron-phosphate minerals. Similarly, research by several investigators indicated that some clay minerals interact with lead to form lead-containing minerals in which the bioavailability is remarkably low. Addition of such clay may enhance the ability of compost to decrease lead availability (U.S. EPA 1997)."

The prescription process for special problems in the environmental contamination game are just beginning to unravel. Brownfields contaminated with heavy metals and unable to establish vegetation also pose huge opportunities due to the ability of compost to establish vegetation. Phytoremediation, (the use of plants to help immobilize or degrade compounds), can also be a tool with the use of compost. "Difficulties in establishing plants in toxic, contaminated matrices, and in compacted and barren materials that are not conducive to plant growth…can be overcome with the addition of compost (U.S. EPA 1997)."

As Phase II is implemented, perhaps many of the prescription products will evolve to target specific cleanup concerns. Compost has proven effective in degrading or altering many types of contaminants, including chlorinated and non-chlorinated hydrocarbons, wood preserving chemicals, solvents, heavy metals, pesticides, petroleum products and explosives. The contaminants are digested, metabolized, and transformed into humus, inert byproducts like CO_2, water and salts (U.S. EPA 1997).

At worst case, compounds absorbed or adsorbed by compost as stormwater passes through could be remediated by composting the materials at a compost site if the product can be collected and transported to the compost site after its effective life at the construction site.

Costs for the application of compost are issues that vary around the country and with each type of application technology. We hope to cover costs more thoroughly in another report issued later this year.

Performance Specifications for Wood Waste Materials as an Erosion Control Mulch and Filter Berm, and Use of Wood Waste Materials for Erosion Control - NETCR

Dr. Ken Demars for the New England Transportation Consortium recently completed another study in March of 2001. This study was unique in that the design called for the use of glass beads of a known size and an erodible soil from a field test site, which was mixed with water and passed through the testing apparatus (a tilt table with controlled irrigation). The suspended solids of the effluent, including a portion of the glass beads, which were analyzed, were used as a measure of filter effectiveness.

The study points out some excellent mechanics of how berms work. "There are two aspects of filtration: retain the soil particles and allow the water to drain away. The retention of particles is a function of the opening sizes in the berm and the sizes of the soil particles. The opening sizes in the berm are in turn related to the sizes of the wood particles (mulch). The ultimate filtration achieved is actually a function of both the opening sizes and the particle sizes. A berm will retain certain sized particles, the retained particles will in turn retain smaller sized particles (Demars and Long 2001)." The Demars study concluded that the mulches used were more effective than geosynthetic silt fence or hay bales.

The particle sizes finer than the #20 mesh sieve were found to be important because they affected the size of port openings in the mulch through which suspended solids may be transported (Demars and Long 2001). This means that in filter berms the ideal percentage of fines vs. coarse materials, regardless of weather or not it is mulch or compost, need to be considered for trapping suspended solids in the #20 mesh sieve size area.

Demars' work included trying dry products and wet products, thinking the moisture would assist in removing a higher percentage of fines. Adding moisture helped when the tests included Pine Bark

Mulch, but did not improve when Ground Stump Mulch was used (Demars and Long 2001).

Compost filter berms are somewhat three-dimensional. As the face of these berms clogs with sediment, the 'coffee filter' mechanism is apparent. Demars found where the filter cake was developed on the face of the berm, some of the flow would pass over the top of the cake and into the berm where no cake had yet formed (Demars and Long 2001). This is obviously similar to water rising up the height of silt fence except that compost has depth that can also filter water inside the berms. As these berms clog, the face of the berm becomes more saturated with soil particles, and water flow rises over this layer to the next available filtration area. We believe this three dimensional situation gives berms their effectiveness compared to other one-dimensional, gravity oriented BMPs.

There is a limitation to the system design, however. The limitation of the filtration process is that the smaller particles reduce the permeability of the system so that the reduced permeability will eventually cause the system to be overtopped during severe rain events, allowing some sediment to escape (Demars and Long 2001). We have seen this in the field and the regulatory field simply wants to make sure berms are maintained as silt fence or other BMPs are maintained throughout the life of the project.

A study commissioned by the New England Transportation Authority (Demars et al. 2000) indicated that wood waste materials are effective in minimizing erosion when applied to the soil surface as a blanket with a thickness of at least 3/4 inches or greater. The untreated control in these experiments produced over 50 times the sediment than the treated surfaces (Demars et al. 2000). The study went on to further indicate other benefits: Wood waste materials were particularly effective at reducing runoff during storms under ½ inch by absorbing rainwater (Demars et al. 2000). This is critical and data from the Bill Stewart work in 1993 suggests the same reduction in runoff water. A reduction in runoff water absolutely increases water infiltration, and cannot help but benefit efforts towards re-vegetation and initial seed germination. These would be especially crucial items for those projects that just need a little more rain to allow germinated seed to fully establish.

Demars also studied filter berms made from wood waste and found they were more effective than either hay bales or geosynthetic silt fence at controlling erosion. Both hay bales and silt fence released one order of magnitude more sediment than the wood waste filter berm (Demars et al. 2000). Of course, wood waste is not compost. The purpose of this study was to determine if the physical properties of wood waste would assist in erosion and sediment control, similar to the project conducted by the same team with compost in 1998. The wood waste materials still underwent a litany of tests, including a solvita test for stability. The previous work in 1998 resulted in a CONEG specification recommending that erosion control materials should be very stable to stable which was not the case for the fresh ground wood waste materials (Demars et al. 2000). A particular test in this research shows similar data in the 10-fold effectiveness claim for the performance of wood waste filter berms. In this case, the wood waste filter berms were 8 times as effective as silt fence and 10 times as effective as hay bales, when the data is compared directly (Demars et al. 2000).

USCC Soil-Water Connection

A leading handout from the U.S. Composting Council entitled, the Soil-Water Connection has been widely referenced and has a number of solid references relating to erosion control using compost. "Research in Kennebec, Maine has shown that surface-applied compost performs as well or better than traditional erosion control techniques. A yard trimmings compost – spread two to four inches over the surface (a compost blanket) – outperformed a jute mat and ground wood waste for erosion control at five sites (USCC 1997)."

Compost was as effective as the standard erosion materials used for protection, but surpassed them in cost effectiveness, vegetation establishment, and slope protection. Costs for compost applications were about 1/3 of the cost of traditional synthetic blankets (USCC 1997).

Compost applied as erosion control tools are often incorporated into the soil after use, offering further benefit and environmental impacts that we are not measuring currently. "Soils rich in organics store, degrade, and immobilize nitrates, phosphorous, pesticides, and other substances that can become pollutants in air or water. Compost, because it adds organic matter to soil, has the ability to bind pollutants to soil systems, reducing both their leachability and absorption by plants (USCC 1997).

Note the majority of this article deals with research regarding composted products. The U.S. Composting Council has a program entitled the Seal of Testing Assurance, under which many of the products used for erosion control are enrolled. Composting of the materials prior to application offers numerous benefits. There is a growing interest in using mulch for many of these applications and although mulch may physically perform some of the same functions as compost, it cannot offer the diverse microbial remediation properties nor the chemical bonding or scrubbing action compost provides. Other concerns about mulch or woody materials being composted are real and are related more to health and safety concerns.

Why compost first?

Weed seed problems

If the material is not composted, you could end up weed seeds like Kudzoo, purple loosestrife, dock, velvetleaf, wild cucumber, or other recognized noxious weeds being spread onto your slopes. Weed seeds are normally killed during the composting process. Kudzoo is a real problem in much of the Southeast and grows rampant along expressways where it takes over like a jungle. The last thing we need is a mechanism to assist its natural spread and composting helps to make sure we will not spread noxious weed seeds. Reasons given by growers for not wanting to use un-composted green materials in California include fear of disease and weed seed problems (CIWMB 2000). The Southeast recently reported estimated losses of $35.5 Billion from the infestation of alien weeds (ESTS 2000).

Insect larvae or egg problems

The health and safety factors compost provided during proper heating is important. The grinding process alone does not necessarily destroy insect larvae, so composting makes sure the cycle is broken. Consider the spread of Gypsy Moths, Borers or other pests that are now causing quarantine restrictions on shipping of nursery stock from state to state. Many of the mulch materials, especially those from yard wastes or land clearing debris include the infested feedstocks that can rapidly spread once used as a mulch. Composting this feedstock first is a key quality control ingredient. Examples of losses indicate this is a severe problem as the Southeast estimates they lose $20 Billion per year from foreign insects (ESTS 2000).

Disease or fungi problems

Fungi and diseases can also be spread but this is even a more serious nature. Cankers, blights, and other diseases, when introduced to a new area via a carrying mechanism like non-composted organics, could easily find a home and become a huge problem. Again, quarantines already exist for many of these problems. Woody wastes from diseased trees, tree trimmings from line clearing companies and other horticultural wastes all may contain some form of infested feedstock that needs to be composted to be safe.

To limit the spread of pitch canker, an endemic disease of Monterey Pine in the coastal area around Santa Cruz, it is recommended that uncomposted materials not be transported to other forested areas in the state (CIWMB 2000). Estimates in the Southeast are significant, including $6.5 Billion in annual losses due to diseases (ESTS 2000).

Vegetation establishment is normally the goal, EVENTUALLY

Regardless of the initial reason for using any kind of a commercially available BMP on slopes, the eventual goal is normally to allow native vegetation to grow and permanently stabilize the slope. Using mature compost allows application of known materials, which enhances plant growth. In tests conducted at the Texas Transportation Institute, Hydraulics and Erosion Control Field Laboratory, vegetation establishment was around 50% when tackified wood chips were used (CIWMB 2000). As a result, this product was disallowed under the Texas DOT standards. Another project at Caltrans used green material mulch and the distinction was made clear. "The materials utilized were variously called "mulch" and "composted mulch" but were, in fact, not compost. Composted materials are those that have undergone thermophillic decomposition and organic matter stabilization (CIWMB 2000)."

The latest in R&D with compost – public and private

Environmental impacts of compost applications on construction sites – Iowa State University

Recent public research completed in 2003 at Iowa State University concluded that compost blankets helped reduce runoff, decrease soil erosion, was successful at replacing soil normally specified for DOT slopes, reduced weed competition, and improved chances for establishing vegetation during extreme climatic conditions. Dr. Tom Glanville's research clearly show that concerns about leaching nutrients from compost is not well founded and in fact, compost helped to reduce overall chemical discharge from the system when compared to control slopes. The total mass of nutrients and metals were significantly lower in the compost plots compared to soil control plots.

This research is important because few systems are measuring success of procedures and practices already accepted and in the Product Acceptability Listings (PAL) of most state DOTs. For instance, normally questions are raised about the addition of nutrients to a watershed via the addition of compost, with a significant nutrient content. However, according the work recently completed by Glanville, even nutrient dense composts like Biosolids did not leach more nutrients than the control. Glanville's work does a good job of separating soluble nutrients contributed from the liquid phase (or those in suspension) compared to those that are adsorbed to the soil surface. When adsorbed to the soil surface, the nutrients are often transported to a water environment (i.e., settling ponds) where they can become more reactive. The 'system' of using compost compared to the 'system' of normal soil seeding generates far less overall nutrients in both categories. In Glanville's report details, many chemicals are from 10-100 times more in the control soil plots vs. the compost applications. This leads us to accept the fact that not all practices are performing as well as some of the new products being introduced, yet little work has been done to measure, study or quantify their impact.

Perhaps the best info in the Glanville work was two fold. First, the time it took water to run off from compost slopes was 8-10 times slower than soil plots. This means that compost has water retention capabilities that are important for conserving moisture for processes like germination. The slower runoff times also translated into lower total runoff volume. Compost generated nearly 20 times less volume of runoff during the same 25-year storm event depicted. This data showed that compost, because it protects soil from movement on the slope, can add significantly to the design parameters currently used in common engineering practices. It also should be taken into account that perhaps with the use of compost, some of the retention and detention ponds designed today could be downsized in order to accommodate less predicted overall runoff. These facts may translate into a significant value for the use of compost in trade for additional building lots that used to be in the space of a retention pond. The value of local real estate in some areas will help to drive this equation, but awareness about the benefits of compost must first be generally known.

Private Research

Two other private companies have performed significant private research. Rexius Inc. in Eugene, Oregon and Filtrexx International, LLC, in Grafton, Ohio, have both used compost commercially in their international erosion control programs. Work concluded in 2002 from Rexius indicated that their Micro blend additive helped reduce total solids 96% when used in a formed filter berm application system. This system, called Eco-Berm, successfully trapped a larger portion of suspended solids than many other tools that have been studied. Compost berms are generally known to have especially good effects on suspended solids, which have proven difficult to trap with geosynthetic systems. The Rexius work shows many similar conclusions to the earliest work conducted by Bill Stewart, in 1993. However, the Rexius work also indicates a large increase in the population of microbes responsible for the absorption and degradation of hydrocarbons. Since this data was published, the company has continued to conduct research on efficacy of using compost for removal of hydrocarbons.

In 2003, Filtrexx International tested a variety of compost products, wood products and mulches to identify flow through rates of each material. Flow through rates are required so that engineers can correctly calculate and predict storm flows when using various tools. Since compost is available in many forms, from many feedstocks, the samples used were indicative of products commonly available in most major metro areas of the U.S. Results from these tests indicated that composted products filtered slightly better than non-composted products, but that when particle size dropped below a certain level, performance of flow through was diminished. The message from this data means that the proper sizing of products used in berms, FilterSocks or other such devices, should be considered.

As an example, wood chips, a common organic material available in most areas, resulted in a 55% reduction in solids > 63um and had a void space of 62%. The flow rate through wood chips was extremely high compared to normal composts. Since wood chips have not been screened, the fines associated with both flow through and filtration is not present. Nor is there any punky mantle around the edges of the carbon chips, which is thought to add significantly to the filtering capabilities of mature composts. A specially screened yard waste compost was also tested and yielded much more favorable results. The yard waste product was approved as a Filtrexx Certified product and removed 100% of the solids in water > 63um, with a flow through rate of 11 gals/min/lin ft. and void space 57%. Other testing with mini bark nuggets indicated that although they had similar pore space to the Filtrexx material (56%), the removal rate was only 12% of solids >63 um. Again, this suggests that composted products somehow filter slightly better than non-composted materials. The Filtrexx data was not repeated in replicate and testing is currently underway to repeat many of these experiments at universities this fall.

Conclusions

Compost is the only 'silver bullet' (if there was only one) to combat many of our environmental challenges and is actually the least expensive opportunity for us to revert to a sustainable culture. Immediately after understanding the benefits of compost when used for erosion control, we wonder about the natural extension into wetlands, and other environmental applications. Of course we realize compost must be used in a BMP approach, integrated with other effective tools, which are also effective at achieving our erosion reduction goals.

Future immediate research needs include understanding how various types of compost perform when screened to a number of particle sizes. Many regulators have expressed concern about berms ponding water when fine composts are used. We need answers to questions about nutrient leaching, binding capacities for all of the chemicals that could be targeted as clean up products and what type of fertilizer, if any, if needed when compost blankets are used. What is the permeability of various types of compost used for filter berms? How long of a slope can effectively drain to a compost berm before needing to add more berms to handle the flow? Is the system currently used (i.e., seed, fertilizer and straw) generating more 'leachate' or nutrients adhered to soil particles than the proposed system using compost products because of the nutrients immediately available in commercial fertilizers? If so, can we promulgate regulations shifting to the new proposed practices? How long will that take? Is compost approved in every state as a BMP?

Due to the number of various composts available, and to the number of soils and rainfall capacities that are in every city, this research will take a long time. However, for the main types of compost produced currently (yard trimmings, biosolids, etc), it is an option that performs at least as well as the other tools currently available for erosion and sediment control.

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Hydrology I

Multiple Approaches to Estimate Ephemeral Channel Recharge

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Abstract

Ephemeral channel transmission losses play an important role in ground water/surface water dynamics in arid and semi-arid basins in the Southwest. However, identification of the processes driving these dynamics is difficult. Specifically, data on the proportion of runoff transmission losses that escape from near-channel evapotranspiration (ET) and wetted channel evaporation to become deep ground water recharge are difficult to obtain. Quantifying recharge with greater certainty is a critical need required to manage basins whose primary source of water supply is derived from groundwater. This issue was addressed via coordinated field research within the USDA-ARS Walnut Gulch Experimental Watershed (WGEW) located in southeastern Arizona. Groundwater, surface water, chemical, isotopic, tree sap flux, micrometeorological techniques, and changes in microgravity were used to independently estimate ephemeral channel recharge. Wet 1999 and 2000 monsoon seasons caused substantial changes in near-channel groundwater levels. Crudely scaled to the basin level, this recharge would constitute between 20 and 50% of basin recharge as estimated from a calibrated groundwater model.

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Keywords: recharge, ephemeral channels, isotopes, evapotranspiration, microgravity, chloride

Introduction

Groundwater recharge is arguably the component of a basin's water balance that is known with the least certainty. In arid and semi-arid regions there is mounting evidence that recharge is likely to occur in only small portions of a basin, where flow is concentrated, such as depressions and ephemeral stream channels (Walvoord et al. 2003). Recharge along ephemeral channels can be large and play an important role in groundwater/surface water dynamics in arid and semi-arid basins (Goodrich et al. 1997). However, it is very difficult to quantify the proportion of transmission losses that escape from near-channel evapotranspiration (ET) and wetted channel evaporation to become groundwater recharge. This project has two principal objectives:

1. Assess the magnitude of ephemeral channel recharge to the regional aquifer.
2. Estimate channel evaporation and near-channel ET.

Study Site

The highly instrumented USDA-ARS Walnut Gulch Experimental Watershed (WGEW) is located in southeastern Arizona. (Figure 1). The watershed has the following attributes (Renard et al. 1993):

- Area: 149 km^2
- Elevation: 1250 to 1585 m MSL.
- Climate: mean annual temperature: 17.6°C
- Precipitation: 324 mm annually (~65% summer convective, ~35% winter frontal).

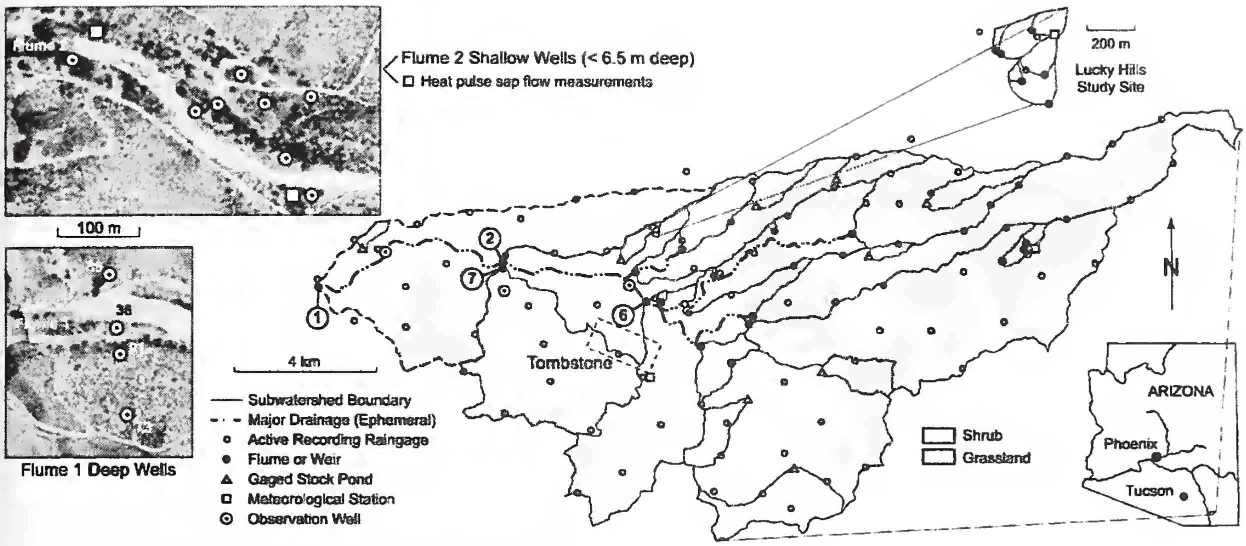

Figure 1. Walnut Gulch Experimental Watershed.

- Soils: well drained, calcareous, gravelly loams with large percentages of rock and gravel at the surface.
- Runoff: almost exclusively from summer monsoon storms via infiltration excess.
- Groundwater: depths to the regional aquifer ranges from 50 to 145 meters.
- Vegetation: dominated by desert shrub steppe species and desert grasslands.

Methods

The primary methods employed to estimate ephemeral channel recharge (R) include:

1. Channel reach water balance:

$R = P + Q_i + Q_l - Q_o - T - E$ where,

P = precipitation from multiple rain gages

Q_i = measured inflow into study reach (flumes 2 and 7 in Figure 1),

Q_l = runoff modeling using KINEROS2 (Smith et al. 1995) to estimate lateral inflow (from the area delineated by a dashed line in Figure 1),

Q_o = measured outflow (flume 1 in Figure 1),

T = scaled sapflow (Barrett et al. 1995) or energy flux (Scott et al. 2003) estimates of near-channel transpiration,

E = estimates of channel evaporation (Sorey and Matlock 1969),

The water balance approach assumes that recharge equals channel transmission losses ($P + Q_i + Q_l - Q_o$) less additional near-channel E and T.

2. Observations and modeling of groundwater mounding (Hantush 1967).
3. Chloride concentration change (Allison et al. 1994).
4. Natural tracers (Allison et al. 1994).
5. Microgravity measurements of water mass change (Pool and Eychaner, 1995).

Instrumentation and Measurements

Figure 1 illustrates the location of rain gages, runoff flumes, and gaged subwatersheds within the WGEW. The Walnut Gulch supercritical runoff flume was specifically designed to provide accurate runoff measurements in mobile bed alluvial channels. In addition to the runoff flumes, the following instrumentation was installed and the following observations were made over 1999 and 2000 (Figure 1):

1. Water levels in deep wells above flume 1 and along the main channel.
2. Water levels in a shallow occluded aquifer above flume 2.
3. Runoff samplers at flumes 6 and 2 for chloride and isotopes.
4. Water samples from precipitation gages and wells for chloride and isotopes.
5. Sapflux measurements above flume 2.
6. Meteorological measurements.

In addition, USGS studies at flume 1 included unsaturated zone sampling, monitoring, simulation of streamflow infiltration, and gravity monitoring.

Observations

Groundwater responses recorded by the shallow wells above flume 2 for 1999, and for the deep wells upstream of flume 1 for 1999-2001, are illustrated in Figure 2.

An example of discharges from flume 2 and chloride concentrations from runoff samples is illustrated in Figure 3.

Oxygen and hydrogen isotope composition of rainfall and runoff for various events in 1999 is illustrated in Figure 4.

Isotope compositions from selected deep wells for 1999-2000 and runoff for 1999 are plotted in Figure 5. The relationship between hydrogen and oxygen isotope composition of precipitation, runoff and deep groundwater for 1999 is plotted in Figure 6.

Isotope compositions from selected deep wells for 1999-2000 and runoff for 1999 are plotted in Figure 5. The relationship between hydrogen and oxygen isotope composition of precipitation, runoff and deep groundwater for 1999 is plotted in Figure 6.

Water from runoff and deep wells at Walnut Gulch apparently is not significantly evaporated from meteoric waters, whereas waters from the slow moving San Pedro River are.

Mean sap flow rates for four well-watered mesquite trees located above flume 2 in the perched aquifer were measured during August of 1999. These measurements were combined with a field survey to establish a relationship between mesquite sapwood area and canopy area. A field survey was also conducted to estimate the area adjacent to the main channel where it was assumed that deep-rooted trees could access water from channel

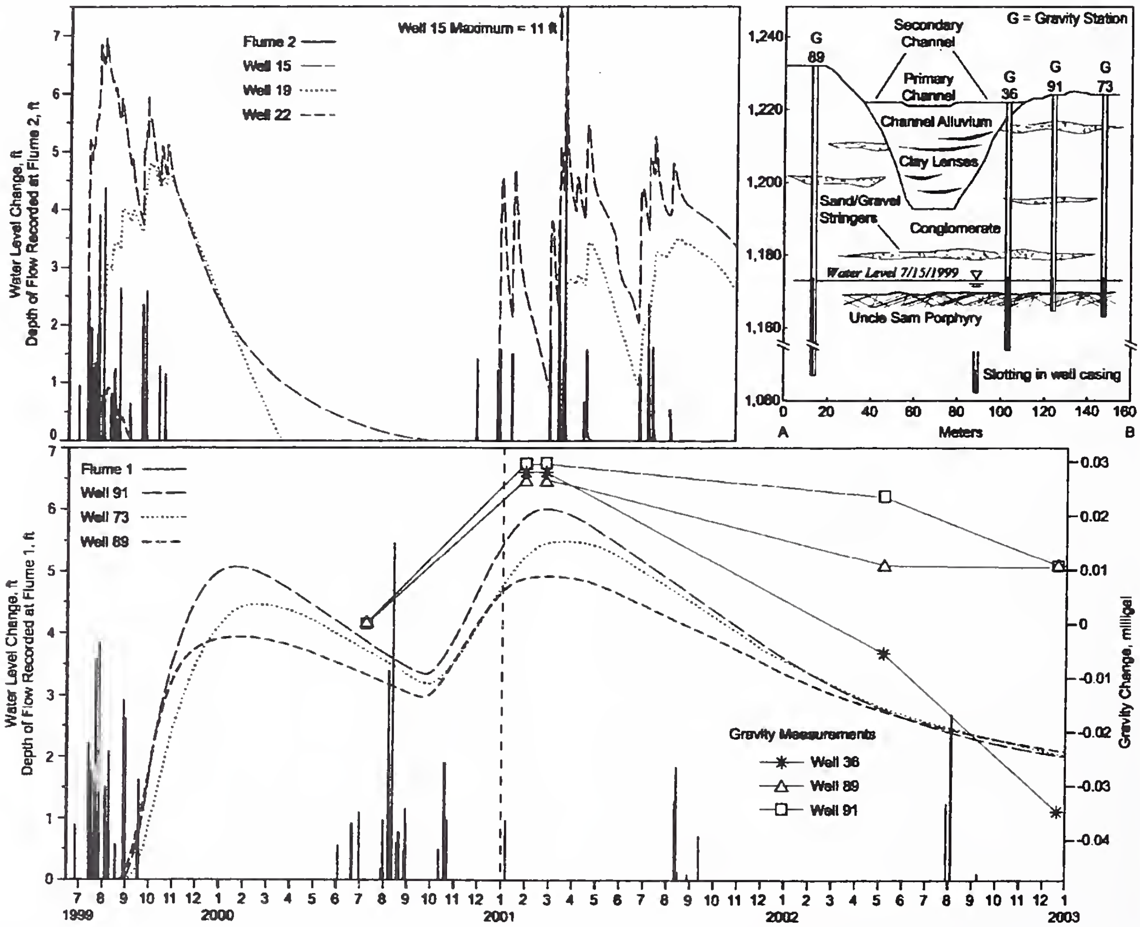

Figure 2. Well levels and flow depths at flume 2 (top) and flume 1 (bottom). Bottom figure also shows gravity measurements at flume 1. Diagram on upper right shows cross-section of well transect above flume 1

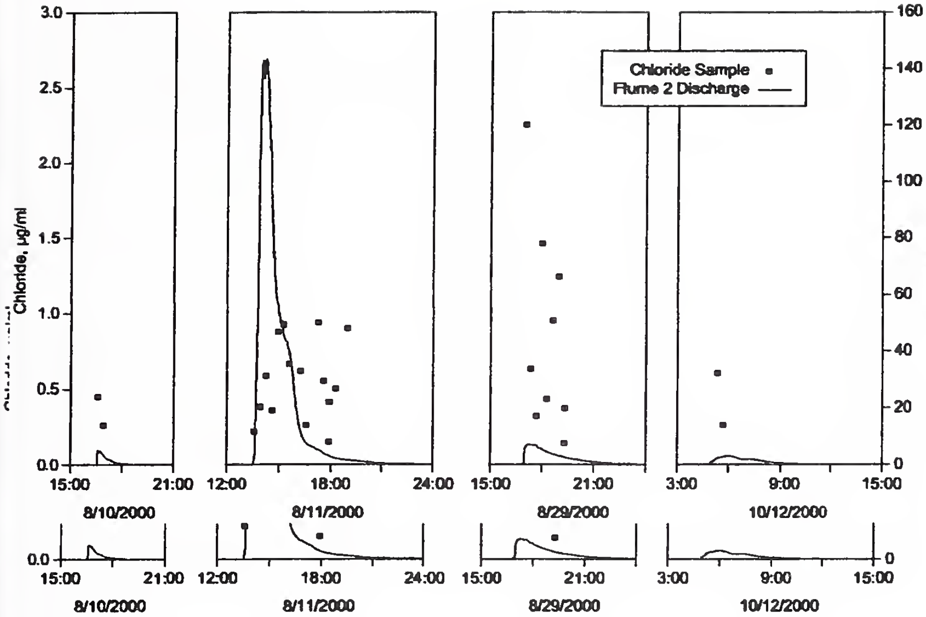

Figure 3. Discharge rate and Cl concentration at flume 2 in 2000.

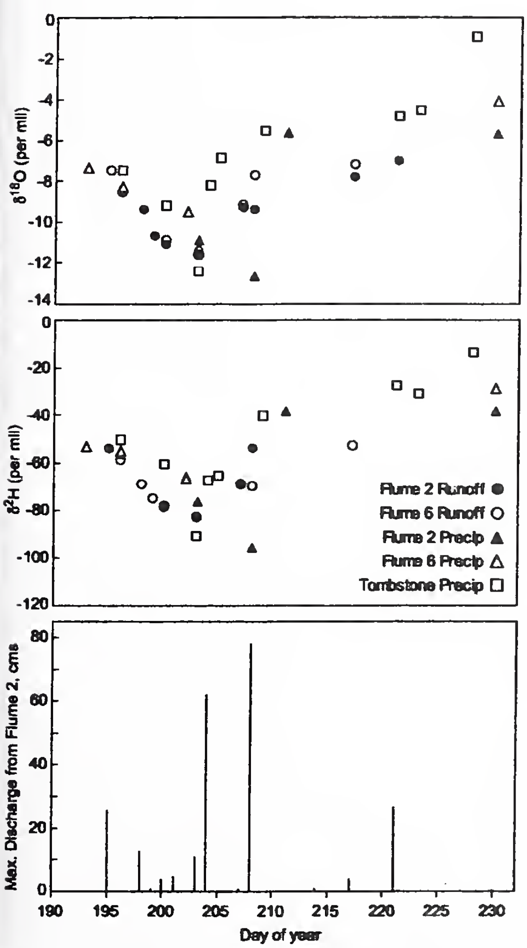

Figure 4. Oxygen and hydrogen isotope composition of rainfall and runoff during 1999, and maximum discharge rates at flume 2.

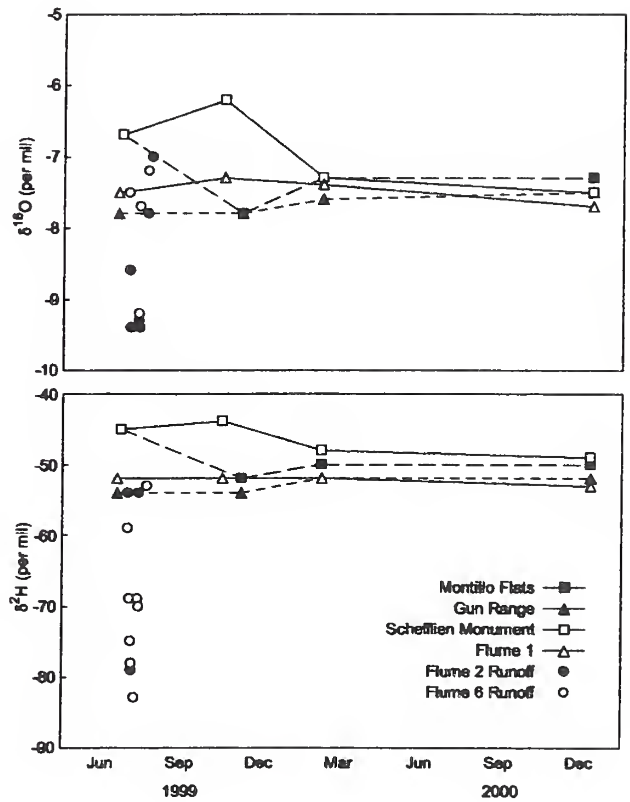

Figure 5. Oxygen and hydrogen composition of deep wells and runoff.

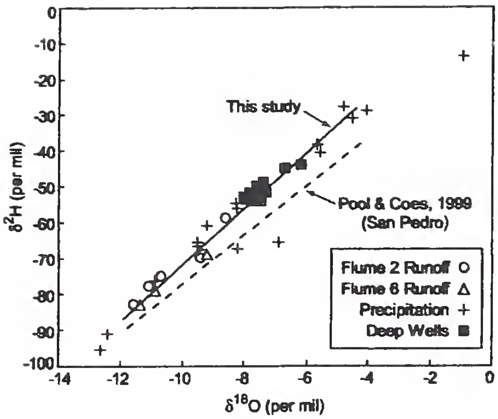

Figure 6. Relation between hydrogen and oxygen isotope composition of rainfall, runoff, and deep groundwater collected in 1999.

transmission losses. Total canopy area in the near channel zone was estimated by image processing techniques to be 10.4% (~ 103,000 m^2).

In addition to sapflow-derived rates of near channel transpiration, transpiration volumes were also computed from a nearby energy flux station positioned over a well- watered mesquite bosque (Scott et al. 2003), as sapflux techniques in mesquite are not well-tested.

Results

Space does not allow a complete presentation of results with error and uncertainty analyses. Average or mid-range values are presented in the following tables.

Table 1. Summary of recharge estimates from modeled transmission losses less abstractions (m^3).

Inputs	*1999*	*2000*
Midpoint Modeled Transmission Losses	514400	399700
Precipitation on Wetted Channel	6500	4000
Precipitation on Canopy less Interception	25800	25500
Total Inputs	546700	429200
Abstractions		
Channel Evaporation	300	800
Near Channel ET (Energy/Flux Estimate)	37500	59100
Total Abstractions	37800	59900
Total Potential Recharge	**508900**	**369300**

Table 2. Recharge estimates from groundwater mounding model recharge volume (m^3).

	1999		
Well	Low	High	Average
89	127200	250400	188800
91	107200	211000	159100
73	214000	421300	317600
	2000		
Well	Low	High	Average
89	85600	168500	127100
91	68500	134900	101700
73	138800	273300	206100

Table 3. Recharge estimates from Cl concentration change from runoff to groundwater (m^3).

Year	Midpoint Trans. Loss Runoff	Well 89 Cl Conc.& Avg. Cl Ratio	Well 73 Cl Conc. & Avg. Cl ratio
1999	514400	312600	88300
2000	399700	242900	68600

Table 4. Comparison of recharge estimates (m^3) *

Year	Trans. Loss less Abstract.	Chloride Well 89 Ratio=0.61	GW Model Average Well 89	Microgravity Change
1999	508000	313000	189000	
2000	369000	243000	127000	
Total	**877000**	**556000**	**316000**	**455000**

* Rounded to nearest 1000 m^3.

Changes in water storage were measured using microgravity methods at a single cross section between at flume 1. Measurements were done during July 2000 through December 2001, a period with few streamflow events and drainage of water that infiltrated during 1999 and 2000. Nearly all of the gravity change and storage loss occurred near the stream channel indicating vertical transport to the water table and little lateral migration through the unsaturated zone. This distribution of gravity change allowed scaling to the flume 1-2 stream reach on the basis of the ratio of average channel width and to flume 1 channel width. A preliminary estimate of total recharge over the 1999 and 2000 runoff season is 455,000 m^3. Results must be assumed to represent a portion of the previously infiltrated water because water levels (Figure 2) indicate that complete drainage of the ground-water system to pre-existing conditions had not occurred by December 2001, however, drainage of water in from the unsaturated zone beneath the channel was likely nearly complete.

Conclusions

Results indicate relatively good agreement between the average estimates from each of the methods, in that they differ by less than a factor of three. This range is not surprising given the limitations of the various methods, and the differences in time scales over which they are applicable.

Another primary purpose of this study was to assess whether recharge from ephemeral channels was a significant component of the overall San Pedro Basin water budget (Walnut Gulch is a tributary to this basin). If one crudely scales the above estimates by the overall length of channel in the basin with a support area equal to that of the drainage area at flume 2, the minimum recharge estimates for 1999 in Table 4 scale to approximately 18% of the total basin recharge estimated from a regional groundwater model (Corell et al. 1996). The maximum value in Table 4 for 1999 scales to roughly 48% of the groundwater model estimate. If the values in Table 4 are even approximately correct it is fair to conclude that ephemeral channel recharge from monsoon runoff can constitute a substantial percentage of overall basin recharge, especially during periods lacking winter runoff.

Acknowledgments

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Monitoring Water Content Changes in the Root Zone with Borehole Ground Penetrating Radar

Ty P.A. Ferre, D.R. Rucker, G. von Glinski

Abstract

The distribution of water within the root zone is critical to understanding of water and nutrient exchange among the subsurface, plant communities, and the atmosphere. While there are many available methods for nondestructive measurement of volumetric water content, no current method offers the ability to monitor rapidly to great depths with high spatial resolution. This measurement need must be fulfilled to allow for quantitative study of water and nutrient flux through the vadose zone. Borehole ground penetrating radar (BPGR) may provide this capability. BGPR can make rapid water content measurements to great depth with little need for medium-specific calibration. However, the method is not without limitations for shallow subsurface applications. This presentation will introduce the use of BGPR for subsurface water content measurement. Then, limitations regarding the shallowest attainable depth of measurement and the minimum measurement resolution will be addressed. Examples of field-measured water content profiles made along the San Pedro River will be presented to demonstrate improved methods of analysis leading to more accurate water content profiles. Finally, areas of future investigation aimed at improving the use of BGPR will be discussed.

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Keywords: ground penetrating radar, water content

Importance of Wetlands to Streamflow Generation

E.S. Verry, R.K. Kolka

Abstract

Hewlett (1961) proposed the variable-source-area concept of streamflow origin in the mountains of North Carolina suggesting streamflow was produced from water leaving saturated areas near the channel. Dunne and Black confirmed this concept on the Sleepers River watershed in Vermont (1970). Areas near the river were saturated by subsurface or interflow from adjacent upland slopes. In turn, these saturated areas fed water directly to the channel. In the northern Lake States, wetlands and lakes make up 10 to 35% of the basin. These flat landscape components are surrounded by relatively steep (10-15% slope) glacial moraine uplands. We investigated the importance of wetlands to streamflow production on watershed two at the Marcell Experimental Forest in north central Minnesota. A hydrograph separation technique for the entire watershed yielded hydrographs for water both from the upland alone and from the wetland alone. Additionally, selected direct measurements of upland runoff and watershed streamflow confirmed the timing of hydrograph peaks for the separated watershed components. The wetland produced 50 to 70% of the annual streamflow even though the wetland comprised only 1/3rd of the basin. Storm peaks from the wetland were 5 to 10 times higher than storm peaks from the upland and occurred about 1 hour before upland runoff peaked. Saturated wetlands (and lake surfaces) are the primary source of streamflow in these glacial landscapes.

Keywords: source of streamflow, peatlands, uplands, subsurface flow, interflow, hydrograph separation

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Introduction

Overland flow to streams results when rain or snowmelt exceeds the infiltration capacity of soils (Horton 1933). However, the generation of surface runoff, basin-wide, was not the source of streamflow in North Carolina forests with an intact forest floor (Hewlett 1961). Instead, Hewlett found streamflow generated from saturated areas near slope bottoms and near channels. The extent of these saturated areas changed during the year and expanded during individual storms. Thus, Hewlett coined the theory of a variable-source-area for streamflow generation. Whipkey (1965) in Ohio measured the amount of subsurface flow in mineral soils and suggested subsurface flow was the source of water saturating lower-slope areas.

Dunne and Black (1970) directly measured the areas of subsurface flow and saturated, over-land flow in the Sleepers River watershed in Vermont. They clearly demonstrated the saturated areas near the stream produced overland flow during summer storms. They also measured significant areas of subsurface flow upslope of the near-stream, saturated areas. In the Susquehanna River basin in Pennsylvania, 50- to 100-year events produced saturated flow even from sloping subsurface flow areas (Yarnal et al. 1997). Pearce et al. (1986) and Bonell (1988) have extensively reviewed the history of the variable-source-area concept and modeling efforts aimed at the processes that generate runoff in forested headwater basins. None, however, have considered the role of wetlands, with annually saturated soils, as a source of streamflow.

Study Site

We examined 20 years of streamflow record from a mixed upland/wetland basin on the Marcell Experimental Forest in north central Minnesota (Lat.

47:31:52N, Long. 93:28:07W) to determine the significance of wetlands to streamflow generation. Watershed No. 2 is a forested headwater basin 9.72 ha in size with 2/3^{rds} of the basin in mineral soils (aspen/birch forests) and 1/3rd of the basin in a centrally located black spruce, sphagnum moss, wetland (a bog in the fennoscandia terminology) (Figure 1). Figure 1 also shows the upland topography (1 m contours) and wetland topography (3 cm contours) and the location of instrumentation used in our evaluation of streamflow origin.

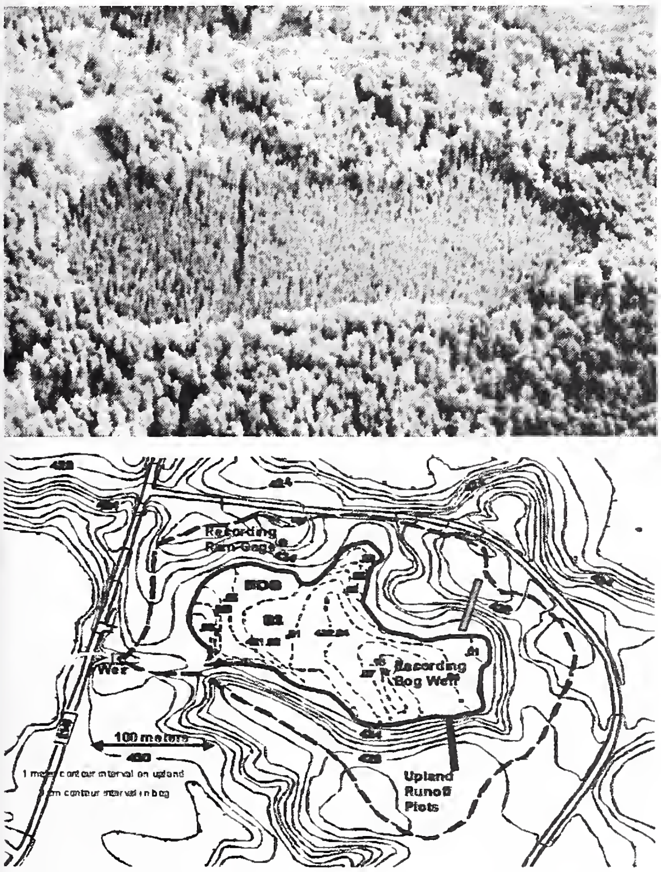

Figure 1. Aerial view of watershed S-2 on the Marcell Experimental Forest in north central Minnesota. The central area is a black spruce bog surrounded by an aspen/birch upland. In the lower map, upland contours are 1 meter and bog contours 3 cm. A recording rain gage, recording well (in the bog) and runoff plots with stage recorders in catch tanks, and the stream hydrograph at the weir were used to measure the timing and amount of streamflow originating from the upland and bog portions of the watershed.

Two sets of upland runoff plots are shown (Figure 1) one with a south aspect and one with a north aspect. At each site there are two upland runoff collection plots. One collects near-surface runoff through the forest floor (O horizons) usually very shallow flow over a frozen mineral soil. The other collects interflow or subsurface flow occurring as saturated flow in the mineral soil A and E horizons over a partially restrictive B2t horizon high in clay. The near-surface flow plots have a corrugated metal boundary, while the subsurface plots collect flow in a stainless steel well screen laid horizontally in a sand-filled trench dug into the surface of the B2t horizon about 30 cm below the surface. Contributions of the upland to total watershed streamflow are based on a hydrograph separation procedure.

Methods

The total watershed streamflow was separated into a bog-only and upland-only component using a hydrograph separation technique (Timmons et al. 1977). An analysis of total watershed hydrographs showed that logarithms of the total watershed recession leg slope were significantly higher (alpha = 0.001) during July and August than at other times. July and August recession legs represent flow periods from the bog only (water collections in upland runoff plot tanks were nil). The modal value for all July and August recession legs from 1961 through 1970 was a negative 0.21 log (base 10) of the total hydrograph recession leg in English units (cubic feet per second per day). Separation of the total stream hydrograph into an upland and wetland component is accomplished by applying the bog-only recession leg to total hydrograph peaks. On an annual basis, the bog-only recession leg is applied beginning with the first snowmelt-peak of the season in March. From that point on, the recession leg is drawn beneath the total hydrograph recession leg until another rising leg occurs. The rise (absolute amount) in the bog-only recession line is identical to the total streamflow rise measured at the weir.

The total hydrograph, rising-leg mimics the rise of the water table in the bog (from a recording well hydrograph) in timing and response. It is always identical to the streamflow weir hydrograph (see Figure 2). However, when the same amount of hydrograph rise is applied to the bog-only recession hydrograph, the peak flow is usually less than the total streamflow hydrograph rise because it begins at a lower "bog-only" value. The redrawn bog-only hydrograph beneath the total watershed hydrograph estimates water originating in only the bog wetland. Finally, the annual bog-only hydrograph (on a daily time step) is subtracted from the total watershed streamflow hydrograph to obtain daily estimates of upland-only contributions to watershed streamflow.

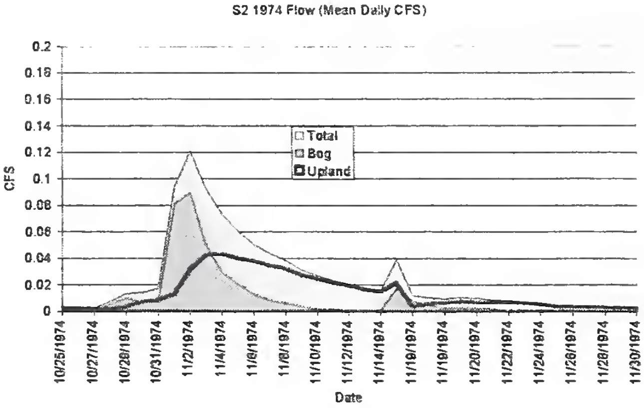

Figure 2. This one-month streamflow hydrograph for the S-2 watershed illustrates the importance of the bog wetland (dark gray) to the generation of total watershed streamflow (light gray) and the lesser importance of the mineral soil upland (black line).

In July of 1997, the upland runoff plot collection tanks were fitted with Belfort FW-I stage recorders and the data reduced to upland runoff hydrographs on an hourly time step with data read to the nearest half hour. Three July 1997 storms were analyzed at a half hour time step using the upland flow plots, bog water-table hydrograph, watershed streamflow weir hydrograph and a recording Belfort rain gage hyetograph to confirm the timing differences between upland and bog flow and the approximate total water yield from each watershed component.

A late October storm in 1974 illustrates the application of the bog-only recession leg (in dark gray) to the total watershed hydrograph (in light gay) (Figure 2). Subtraction of the dark gray from the light gray yields the upland-only, black hydrograph. The separations are based on a daily time step. Note that annual hydrographs are in cubic feet per second.

Results

Half hour upland and bog-only timing hydrographs

Three storms in July 1997 show the accumulated precipitation hyetograph, the bog well hydrograph, the total streamflow hydrograph, and two runoff-plot hydrographs for subsurface flow. On the larger, July 14 storm, the precipitation peaks at 8 AM along with the bog well and weir. In contrast the upland runoff plots peak at 9 and 10 AM for the south and north aspect respectively. A closer examination of the July 14 storm illustrates the detail of flow timing (Figure 3). The streamflow unit (cubic meters per minute) is 1000 times the unit for subsurface flow (liters per minute). However, the area of the upland 97,200 square meters is about 1000 times the area of one upland runoff plot (2 m x 49 m = 98 square meters). So the size of the hydrograph plots represent the approximate contributions to the total streamflow hydrograph.

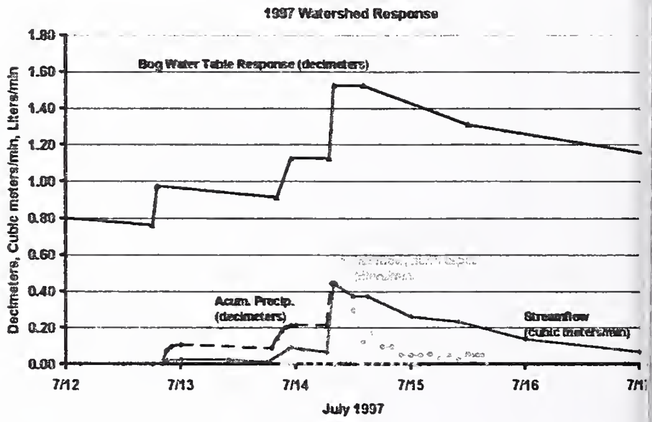

Figure 3. Direct measurements of precipitation on July 13 and 14, 1977, the upland runoff plot response and the rise of the water table in the bog show the bog responds first followed in an hour or two by the respective upland runoff plots.

The large differences between the north and south aspects reduce the combined upland runoff plot amounts to about a quarter of the total hydrograph for this particular storm. The subsurface flow for the south aspect is always less than interflow for the north aspect. Perhaps the south aspect always dries faster than the north aspect and thus has more soil water storage. Or differences in the undulation of the impeding B2t clay layer augments interflow collection on the north aspect and diverts interflow on the south aspect.

A comparison of the daily time step hydrograph separation with an hourly time step hydrograph for the same period is shown in Figure 4. The hourly hydrographs show as solid lines, from directly measured upland flow and confirm the delayed upland response compared with the bog well hydrograph and total watershed hydrograph. The annual hydrograph separation, using a daily time step, is shown with dashed lines and smoothes the hydrograph separation over several days. However, the area beneath both upland hydrograph curves (daily or hourly) is similar.

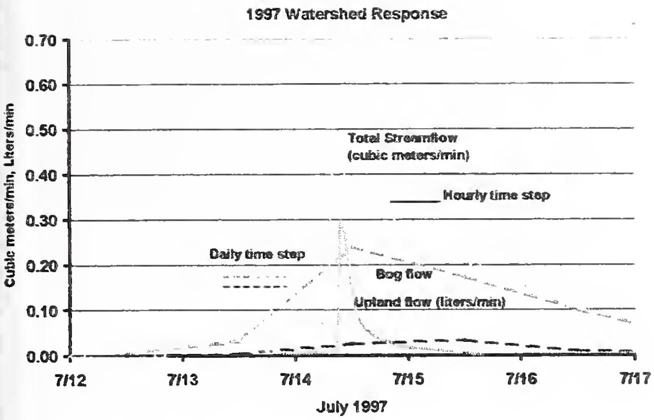

Figure 4. The July 14, 1977 storm shown with directly measured, hourly time step values for the upland runoff plots and total watershed stream flow (solid lines), and using a daily time step hydrograph separation technique for upland, bog, and total watershed streamflow (dashed lines). While the daily time step alters the actual timing, the area of total watershed and upland flow components are approximately equal for the daily and hourly time step hydrographs.

Examples of wetland-only response in annual hydrographs

The annual hydrograph separation for several years illustrates the role of the wetland ($1/3^{rd}$ of the basin) versus the upland (2/3rds of the basin) in producing the total watershed streamflow. Water years run from March 1 to February 28. In 1965, the first peak response is caused by melt of the snowpack followed by large rainstorms in May, June and late September. Throughout 1965, the bog responded first and peaked in flow rate 1 to 1.5 times the slower responding upland flow. This result occurs even though the upland is $2/3^{rds}$ of the basin (Figure 5). Note the overall streamflow level on the Y-axis of each figure (cubic feet per second).

In 1968 peak flows were generally smaller, but the basin remained wet and responsive throughout most of the year. Again upland flow lagged peatland flow, but peak flow was more comparable between the two sources even though the bog-only portion contributed more and higher streamflow peaks (Figure 6).

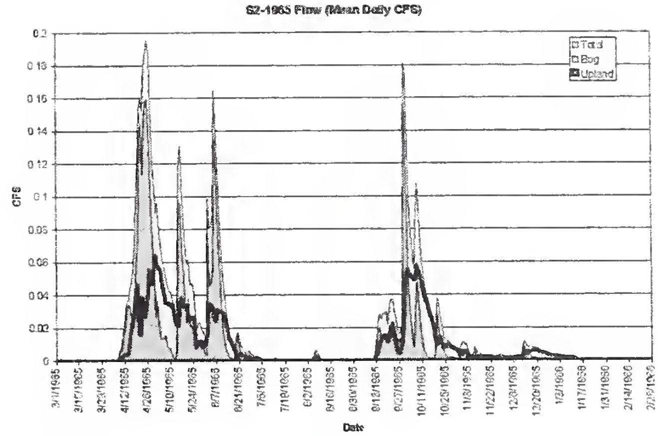

Figure 5. The 1965 hydrograph shows the bog always contributed first before the upland and had peak flows 1 to 1 ½ times the upland flow peak.

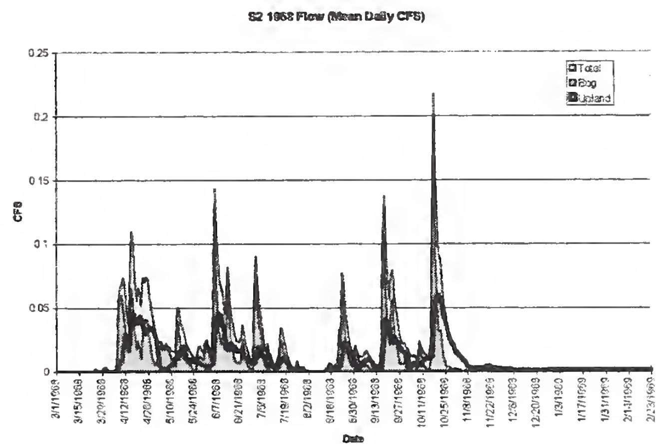

Figure 6. The 1968 hydrograph had lower streamflow than in 1965, but the bog portions again contributed first with greater peaks than the upland.

In 1966 the snowmelt and spring rain period was typical, but a dry summer stopped streamflow nearly 2 1/2 months. When a large August storm occurred, the bog responded first with a very high peak flow, 8 times the upland flow peak (Figure 7). The dry uplands provided significant soil water storage space before subsurface flow began.

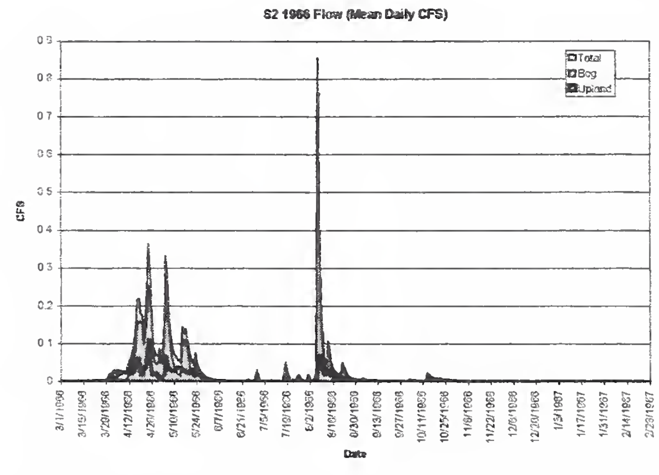

Figure 7. The 1966 hydrograph had a typical spring snowmelt period and a large summer storm when the bog responded first with a peak flow 8 times the upland peak flow.

1975 had a large snowpack and peak streamflow was high. In this spring, bog and upland contributed nearly identical spring flow volumes and nearly equal peak flows (Figure 8). A severe drought extended well into 1976 and meager snowmelt was mostly from the upland because the bog water table had dropped more than 1 meter during the drought (Figure 9).

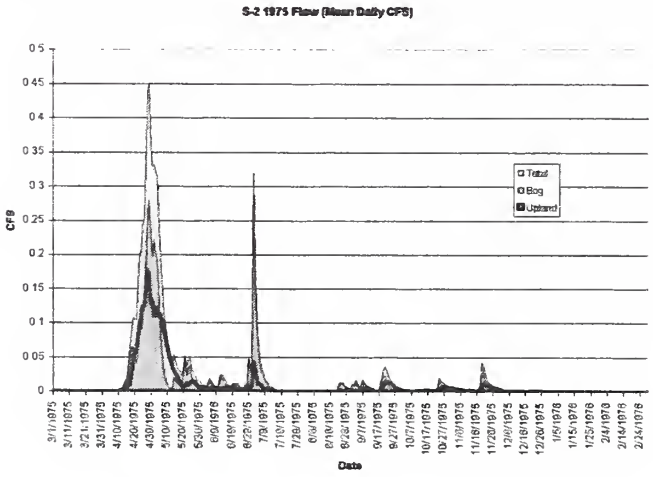

Figure 8. The 1975 hydrograph had a very large snowmelt when both the upland and bog become saturated and both had similar flow amounts and peak rates. A drought began in July of 1975 and extended through 1976.

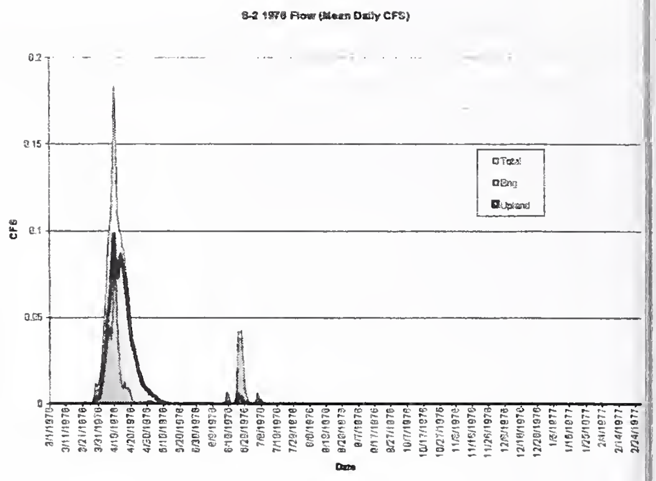

Figure 9. The 1976 hydrograph showed a small snowmelt peak, but the upland dominated because the bog water table had dropped over 1 meter during the drought.

In 1977, storage within the peat profile of the bog was quickly satisfied when spring rains broke the drought temporarily and bog-origin flow continued to dominate (Figure 10). In late September, when moderate storms, falling after leaf-fall, fully satisfied upland mineral soil water storage capacity, upland runoff gained in importance when interception on the bog black spruce was a significant factor in the peatland streamflow generation.

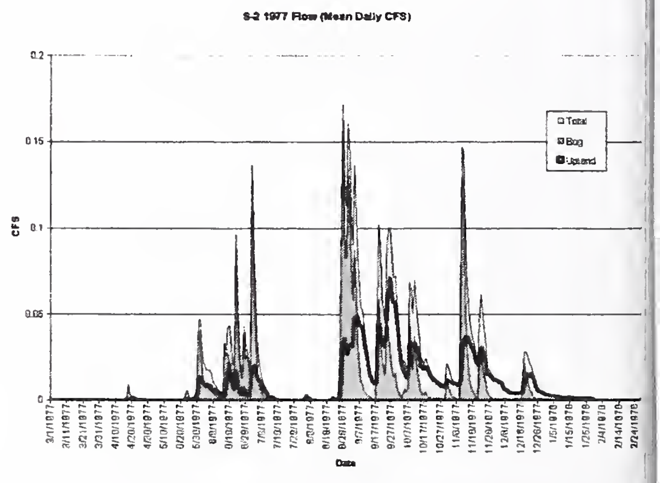

Figure 10. The 1977 hydrograph responded when spring snowmelt satisfied the large amount of soil water storage in both upland and peatland. Note however, the importance of upland streamflow in late September. Moderate storms falling after upland leaf-fall fully satisfied upland soil moisture storage, while spruce interception in the peatland reduced flow from the peatland area.

A very large, intense, convectional storm in July of 1979 (17 cm or 6.7 inches fell in less than 24 hours) immediately filled storage space in the bog, flooding the surface so only the tallest hummocks poked above water. The streamflow peak from the bog was 11 times that from the upland (Figure 11). The amount of soil water storage in the upland or wetland does affect storm response, but what is the relative importance of upland and wetland on an annual basis year after year?

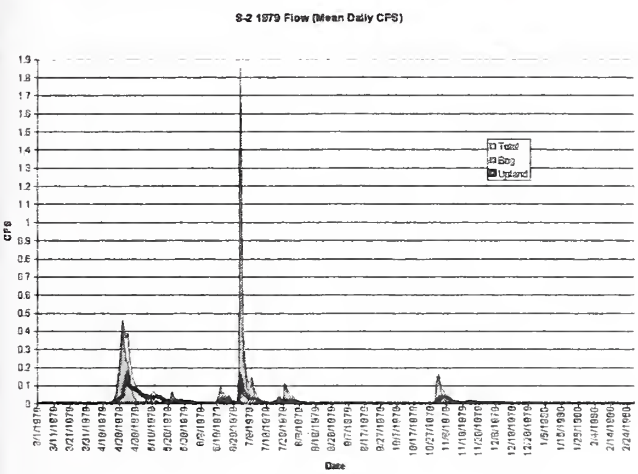

Figure 11. In 1979 a very large and intense July storm was produced mostly by bog only streamflow because the bog water table was still high, but upland soil moisture storage was high.

Average annual streamflow contributions

On average, the peatland produces 58% of the streamflow even though it occupies only 33% of the basin area. The peatland contributions range from 35 to 74% of the annual streamflow. The upland, on average, produces 42% of the streamflow even though it occupies 66% of the basin. During dry years in 1976 and less so in 1980, large amounts of soil water storage became available deep within the bog peat and the relative roles of the peatland and upland reverse (Figure 12).

The relationship of peatland and upland is obvious when annual streamflow contributions are plotted over water year precipitation (not shown). When plotted against water-year precipitation, the slope of the peatland streamflow response curve is 70% of the total streamflow response curve, and the upland streamflow response curve is 29% of the total. Thus the importance of each watershed component to total streamflow is the reverse of their relative areas.

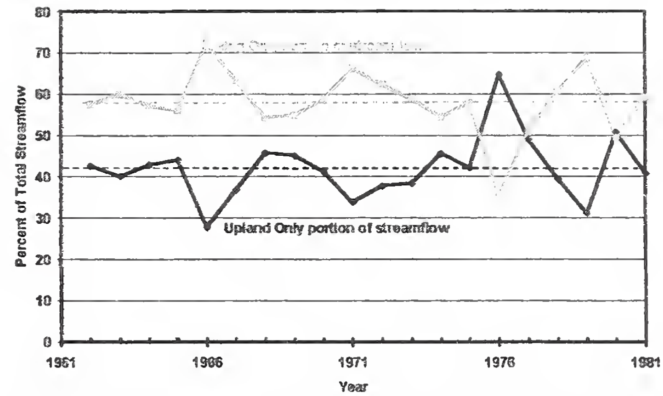

Figure 12. A twenty-year comparison of the annual amount of watershed streamflow originating from the upland (black line, average dashed) and the bog (gray line, average dashed).

Conclusions

Wetlands produce 50 to 70% of watershed streamflow even though they comprise only 1/3rd of the basin. Wetland storm peaks are 5 to 10 times greater than upland storm peaks, and upland storm peaks are delayed about an hour.

The data in this paper compares the relative contribution of upland and wetland to total watershed streamflow in a basin with moraine uplands (5 to 15% slopes) surrounding a flat wetland (black spruce bog). In this scenario the wetland is the primary producer of streamflow and primarily controls the magnitude of the storm peak. This arrangement of upland and wetland is common for Lake State pothole wetlands forming the beginning of stream systems.

Our comparison does not consider the peak streamflow response of landscapes dominated by wetlands versus landscapes dominated by steep-sloped uplands. Large landscapes with wetlands (and lakes) significantly reduce stormflow peaks at all recurrence intervals compared to landscapes with few wetlands and lakes (Conger 1971, Moore and Larson 1979, Ivanov 1981, Taylor and Pierson 1985, Roulet and Woo 1988, Johnston et al. 1990). Our evaluation of streamflow response to peak flows evaluates small basins without a groundwater, or base flow, component contributing to streamflow.
Our experience at the Marcell Experimental Forest with wetlands that do receive large groundwater inflow show similar peak flow responses on top of a substantial base flow component. Total streamflow from these groundwater-fed wetlands (fens) may be ten

times the streamflow from surfacewater-fed wetlands (bogs), yet peak flow responses are similar.

Further research looking at a longer record and examining the potential role of soil water storage can further define the role of wetlands in streamflow generation.

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Streamflow Response of an Agricultural Watershed to Seasonal Changes in Precipitation

Michael W. Van Liew, Jeanne M. Schneider, Jurgen D. Garbrecht

Abstract

Seasonal variations in precipitation on a watershed lead to variations in streamflow that in turn result in uncertainties that impede the efficient management of available water resources. This is especially true for management of reservoir storage and water releases during and at the end of the dry season when water demand is highest and streamflow supply is lowest. Anticipating streamflow amounts based on seasonal precipitation forecasts holds promise to estimate the probability of replenishment of depleted reservoir storage and helps identify best water supply management strategies related to anticipated streamflow and associated uncertainties. A study was conducted to evaluate the impact of hypothetical seasonal variations in precipitation on streamflow. The objective was to develop a prototype for streamflow response associated with a range of hypothetical precipitation forecasts. The prototype was developed for the 33 km^2 subwatershed 442 located in the USDA-ARS Little Washita River Experimental Watershed in Southwestern Oklahoma. The Soil and Water Assessment Tool was used to determine streamflow responses to hypothetical precipitation forecasts that represent changes of $\pm$ 20% and $\pm$ 40% for the fall quarter. Measured precipitation data for a period of record from 1971 to 2000 on the subwatershed was used to develop the hypothetical precipitation forecasts. Test results of this study indicate that hypothetical precipitation forecasts that are drier than normal lead to streamflow responses that approach baseflow conditions on the watershed, while forecasts that are wetter than normal lead to higher streamflow values characterized by considerable variability due to variations in storm size, duration, and intensity during the fall months. Results of this study suggest that utilization of precipitation forecasts coupled with the corresponding anticipated streamflow changes could provide sufficient risk-based information that enable water authorities to more effectively manage reservoir storage and water releases to meet water demands and downstream flow requirements.

Keywords: climate forecasts, watershed simulation, streamflow, SWAT

Introduction

Seasonal variations in precipitation have a major bearing on monthly or seasonal streamflow amounts in Southern Great Plains watersheds. Seasonal variations in streamflow, coupled with increased and competing demands for water by a growing population, place considerable pressure upon efficient management of available water resources. This is especially true for management of reservoir storage and water releases during and at the end of the dry season when water demand is highest and streamflow supply is lowest. Water resources managers and natural resources conservation agencies require better tools to more effectively assess and manage reservoir storage and water releases during and at the end of dry seasons.

Anticipating streamflow amounts based on seasonal precipitation forecasts holds promise to estimate the probability of replenishment of depleted reservoir storage. With such information, water authorities can make more informed decisions on releasing or withholding the reserve water storage at a time when water rationing is not uncommon in Oklahoma. For example, a forecast of wetter than average conditions coupled with the corresponding anticipated streamflow increases can provide sufficient risk-based information

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that may lead water authorities to delay imposing water restrictions or lift restrictions earlier in anticipation of likely increased streamflow.

The National Oceanic and Atmospheric Administration's Climate Prediction Center (NOAA/CPC) issues seasonal climate forecasts covering overlapping 3-month periods for the coming year (Schneider and Garbrecht 2003). These forecasts are issued monthly and can be viewed at the NOAA/CPC web site: www.cpc.ncep.noaa.gov. The predictive skill of these experimental forecasts has been improving as forecast techniques evolve and more data become available (Barnston et al. 1999, Mason et al. 1999, Barnston et al. 2000). The anticipated improvement in the reliability of these seasonal forecasts should produce valuable information that could be used to estimate reservoir inflow to more effectively manage reservoir storage and water releases for multiple competing demands. However, the cause and effect relationship between seasonal variations in precipitation and streamflow is not well known. A study was conducted to evaluate the impact of hypothetical seasonal variations in precipitation on streamflow. The objective of this study was to determine the streamflow response associated with a range of hypothetical precipitation forecasts for the fall quarter. This season of the year marks the beginning of the water year and corresponds to the onset of replenishment of reservoir storage. The prototype was developed for a sub-watershed located in the USDA-ARS Little Washita River Experimental Watershed in Southwestern Oklahoma.

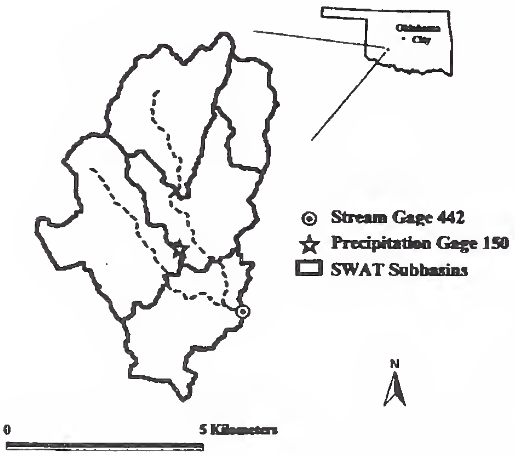

Figure 1. Location of Little Washita Experimental Watershed.

Methods

Test watershed

Subwatershed 442 of the Little Washita River Experimental Watershed (LWREW) is located about 100 km southwest of Oklahoma City and drains an area of 33 km^2 (Figure 1). The climate in the region is sub-humid to semi-arid, with an average annual precipitation during the past 40 years of about 800 mm. Average annual runoff from the watershed is 160 mm based on a period of record from 1992 to 2000. The topography of the watershed is characterized by gently to moderately rolling hills on predominantly silt loam soils. Conventional land use surveys from aerial photographs and point sampling show 54% of the watershed in rangeland, 41% in cultivation (primarily wheat and alfalfa), 1% in timber, and 4% in miscellaneous use (farmsteads, abandoned oil sites, and urban). Based on surveys (Allen and Naney 1991) and recently collected remotely sensed data, little change in land use has occurred on the watershed during the past 30 years, and for this study it was assumed that land use remained constant during the time period of study.

Model description and data input

One of the watershed loading and transport models included in the U.S EPA's Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) 3.0 is the Soil and Water Assessment Tool, referred to as SWAT (Arnold et al. 1998). This model was selected for simulating hydrologic response to precipitation on subwatershed 442. The model delineates a watershed as a number of sub-basins, which are simulated as homogeneous areas in terms of climatic forcing, but with additional subdivisions within each subbasin to represent different soils and land use types. Each of these individual land use areas is referred to as a hydrologic response unit (HRU). In this study the SCS runoff curve number option was used to estimate surface runoff from precipitation. Values of the curve number were adjusted during simulation to reflect changes in moisture conditions on the watershed (Arnold et al. 1998). Simulations conducted in this study by SWAT were performed within the ArcView Geographical Information System (GIS) of BASINS 3.0. This system includes a modular structure that contains a tool for optimizing the definition and segmentation of the watershed and network based on topography. It also consists of a tool for defining the HRUs over the watershed and an integrated user-friendly interface (Di Luzio et al. 2002).

Elevation, land use, and soil characteristics for the subwatershed were obtained from GIS data layers at a 30 by 30 m cell resolution. The elevation layer was developed from concatenated USGS DEM quads. The land use layer was obtained from a 1997 Landsat-5 thematic mapper image of the watershed, and the soils layer was obtained from STATSGO soils information (USDA-NRCS 1992) and from data reported by Allen and Naney (1991). A continuous precipitation recording gage within the subwatershed provided input for daily precipitation. Rainfall data were collected by the USDA ARS from 1971 to 2000, and streamflow data were collected by the U.S. Geological Survey at gage 442 between 1992 and 2000 (Figure 1).

Model calibration

SWAT was calibrated on subwatershed 442 by adjusting model parameters so that the measured and simulated streamflow from the 1992 to 2000 period of record agreed as closely as possible. Details of the procedure for model calibration are given by Van Liew and Garbrecht (2003) in a previous study on the LWREW. Results of the model calibration show that SWAT estimated annual runoff within $\pm$20% for 7 of the 9 years of record for subwatershed 442. A Nash Sutcliffe (Nash and Sutcliffe 1970) coefficient of efficiency was computed for measured and simulated monthly runoff for the period of record from 1992 to 2000, and indicates that simulation results were considered good.

Simulation methodology

Measured daily precipitation amounts for each of the 30 years of fall precipitation were input in SWAT to simulate hydrologic responses of the watershed. Since streamflow response depends on antecedent climatological and soil moisture conditions prior to fall months, the same antecedent conditions were simulated by the model for each fall quarter. Model simulations under dry and wet antecedent conditions must also be conducted, and are anticipated for future studies. Results of the model simulation were used to construct a flow exceedance curve and a precipitation streamflow response relationship for the fall quarter.

Results

Figure 2 displays the three-month total precipitation and flow exceedance curves for the fall quarter. The exceedance curves indicate the percent of time that a given amount of precipitation or streamflow, expressed in mm, is equaled or exceeded. Each curve is a statement of probability that a given amount of precipitation or streamflow will be equaled or exceeded. For example, there is an 80% probability that the amount of precipitation during the fall months will be equal to or greater than 93 mm, but only a 20% probability that it will be equal to or greater than 257 mm.

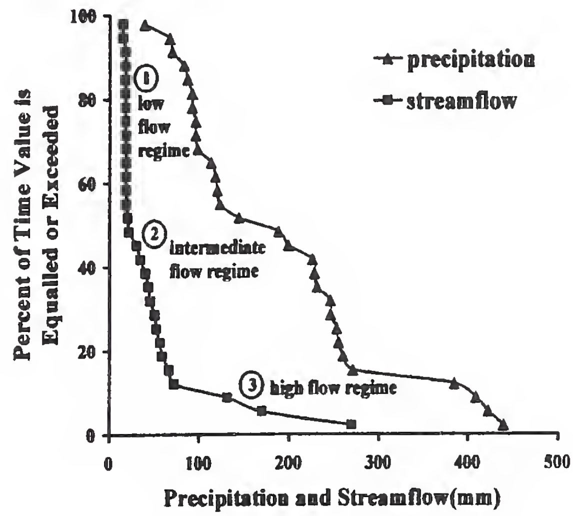

Figure 2. Precipitation and streamflow probability of exceedance curves for fall months on subwatershed 442.

Three general hydrologic responses are evident from the shape of the flow exceedance curve in Figure 2. These include 1) a low flow regime in the 55% to 100% flow exceedance range, 2) an intermediate flow regime in the 10% to 55% flow exceedance range, and 3) a high flow regime in the 0% to 10% flow exceedance range.

The three-month precipitation and corresponding streamflow data were also plotted in Figure 3 to develop a precipitation streamflow response curve. Simulation results suggest that for the 0 to 150 mm precipitation range, the precipitation streamflow relationship is very well defined, with baseflow as the dominant factor in governing streamflow. For the 150 to 300 mm range in precipitation, variation in the streamflow response reflects scatter in the data primarily associated with runoff due to varying storm sizes, intensities, and durations. Variability in streamflow response is even more pronounced in the 300 to 450 mm precipitation range, where values of streamflow range from 59 to 270 mm.

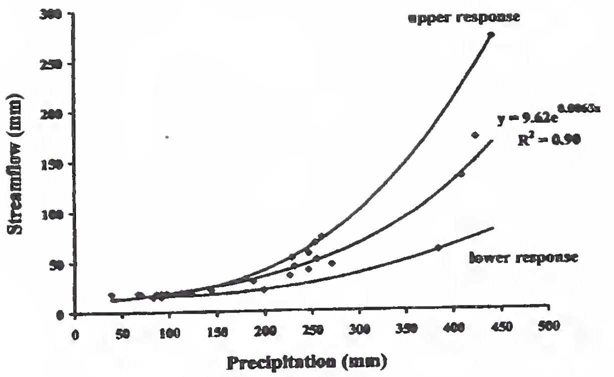

Figure 3. Precipitation streamflow relationship for fall months on subwatershed 442.

Streamflow data were fit to an exponential function:

$$y = 9.62e^{**}0.0065x \qquad (1)$$

where y = streamflow in mm and x = precipitation in mm

Figure 3 also includes two additional curves that were constructed to designate the lower and upper streamflow responses that would be expected to occur for a given amount of precipitation. Equation (1) was used to estimate the streamflow response that would be expected to occur for a given precipitation amount corresponding to a particular probability of exceedance.

Table 1. Streamflow response for various precipitation forecasts for the fall months on subwatershed 442.

Precipitation Forecast	Probability of Exceedance (%)	Precipitation (mm)	Average Streamflow (mm)	Change in Avg Streamflow (%)	Lower Flow Limit (mm)	Upper Flow Limit (mm)
No Change	20	257	51	-	27	72
	40	228	42	-	26	53
	60	118	21	-	16	22
	80	93	18	-	15	17
+20%	20	308	71	+39	39	106
	40	274	57	+36	33	79
	60	142	24	+14	17	25
	80	112	20	+11	16	20
-20%	20	206	37	-27	23	44
	40	182	31	-26	20	35
	60	94	18	-14	17	18
	80	74	16	-11	16	16
+40%	20	360	100	+96	53	156
	40	319	77	+83	42	115
	60	165	28	+33	18	30
	80	130	22	+22	17	23
-40%	20	154	26	-49	17	26
	40	137	23	-45	16	24
	60	71	15	-29	15	15
	80	56	14	-22	14	14

For this study these streamflow responses were computed under normal climatic conditions at the 20%, 40%, 60%, and 80% precipitation exceedance levels (Table 1). Other levels could also be chosen to correspond to specific management decision points. The range in expected flows as estimated from the enveloping curves is also tabulated in Table 1 (last two columns). Under normal climatic conditions, the data show that precipitation equal to 228 mm (corresponding to a 40% probability of exceedance level) results in 42 mm of runoff during the fall quarter, with a possible range from 26 to 53 mm.

Figures 2 and 3 were utilized to determine the streamflow response to hypothetical precipitation forecasts. The forecasts were derived by shifting the precipitation exceedance curve in Figure 2 to the right or to the left to reflect a forecasted increase or decrease in the odds for precipitation. For this study the forecasts consisted of changes in fall precipitation equal to $\pm$ 20% and $\pm$ 40%. The curves presented in Figure 3 were then used to estimate the average and range in streamflow responses for precipitation at the 20%, 40% 60%, and 80% exceedance levels.

For fall precipitation forecasts that are drier than normal, the expected hydrologic response tends to approach baseflow conditions on the watershed. Forecasts in this direction therefore tend to reflect less scatter associated with storm variability. Test results show that a 40% less than normal precipitation forecast, for example, would on average lead to a 22% reduction in streamflow (14 mm, no range) at the 80% probability of exceedance level, and a 49% reduction in streamflow (26 mm, range 17 to 26 mm) at the 20% exceedance level (Table 1).

For fall precipitation forecasts that are wetter than normal, the hydrologic response of the watershed becomes increasingly more variable for respective increases in the departure from normal precipitation conditions. For example, a 20% greater than normal precipitation forecast results in a 39% increase in streamflow (71 mm, range of 39 to 106 mm) at the 20% probability of exceedance level, whereas a 40% greater than normal precipitation forecast leads to a 96% increase in streamflow (100 mm, range of 53 to 156 mm) at that exceedance level. Forecasts in this direction therefore tend to reflect increased variability in streamflow response due to wider variations in storm characteristics during the fall months.

Results of these hypothetical precipitation forecasts suggest that forecasts that are drier than normal lead to streamflow responses that approach baseflow conditions on the watershed, while forecasts that are wetter than normal lead to streamflow responses characterized by increased streamflow and considerable variability. These differences in hydrologic response are important factors that would need to be considered in developing risk-based information related to water resources management.

For application to water supply considerations, the analysis described herein for the fall season could also be extended to other antecedent conditions and other seasons of the year. Using the probabilities of streamflow response for each season over the course of a water year, a multiple stage water resources planning scheme could be implemented for a reservoir to meet various competing water needs during the year (Anderson et al. 2000). The first decision of how much water to store over the course of the coming wet season would be made at the beginning of the water year in the fall. The determination of how much water that would be needed for reservoir storage would be based on water supply and flood storage considerations. With potential changes in the probability of streamflow response for the winter and spring seasons, new decisions would be made as to what additional measures should be implemented for water supply considerations and flood protection. These decisions could be updated each month as new seasonal precipitation forecasts become available during the year. A set of lookup tables could be developed to evaluate various water availability classifications ranging from very wet to very dry conditions. These tables could be used to assess the impacts of storing, releasing, and conserving water for each water availability classification. In turn, a method that reflects risk and uncertainty associated with climatological forecasts would be available for water resources authorities to balance limited water supplies during the dry season with various competing demands.

Summary and Future Research

A study was conducted to evaluate the impact that seasonal variations in precipitation have on streamflow. A precipitation exceedance curve was constructed from 30 years of historical precipitation data for a 33 km^2 subwatershed of the Little Washita River Experimental Watershed. The Soil and Water Assessment Tool was then used to generate hydrologic responses to the historical record in order to develop prototype streamflow exceedance curve and precipitation streamflow response relationships. The precipitation exceedance curve and the precipitation streamflow response relationships were in turn used to determine streamflow responses to hypothetical precipitation forecasts that represent changes of $\pm$ 20% and $\pm$ 40% for the fall months. Results of this preliminary investigation reflect two types of uncertainty associated with the estimation of streamflow response from precipitation forecasts. One type of uncertainty is related to the nature of the forecast itself as defined by the precipitation probability of exceedance curve. The other type of uncertainty relates to the impact that storm characteristics such as size, duration, and intensity have on runoff. Both of these uncertainties must be considered in making risk based assessments of water resources as related to the replenishment of reservoir storage during a given season of the year.

This preliminary investigation was conducted only for the fall months of the year on subwatershed 442 under near average antecedent precipitation conditions. Model simulations also need to be completed for dry, average, and wet antecedent conditions for each of the seasons of the year. In this study the impact of hypothetical precipitation forecasts was based on computing relative changes in the 30 year historical precipitation record for the watershed. Studies anticipated for the future will utilize climate generation techniques to develop precipitation exceedance curves that represent NOAA/CPC precipitation forecasts. Results of these studies will also need to be extended to other watersheds in the Southern Great Plains to evaluate the effectiveness of this method as a tool to more effectively manage reservoir storage and water releases for competing demands.

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Stochastic Daily Rainfall Generation in Southeast Arizona: An Example from Walnut Gulch Experimental Watershed

Huey-hong Hsieh, Jeffry Stone, D. Phillip Guertin, Donald D. Slack

Abstract

The thunderstorm rainfall in southeast Arizona has high spatial and temporal variation. Using areal-averaged rainfall for watershed modeling had caused over or under estimations in erosion and runoff peak. This is a common problem usually faced by watershed researchers. This study had presented a series of analyses to identify spatial characteristics of thunderstorm rainfall based on statistical analyses on Walnut Gulch Experimental Watershed (WGEW) rainfall records. A stochastic daily summer rainfall generator was constructed and calibrated based on derived statistical characteristics and selected events. This rainfall generator can be used for advanced hydrological modeling.

Keywords: thunderstorm rainfall, rainfall generator, hydrological modeling, spatial characteristics

Introduction

Thunderstorm rainfall in semi-arid area has very high spatial and temporal variability (Osborn et al. 1993). Knowledge of the spatial characteristics of thunderstorm rainfall is important for the increasing demands of distributed hydrological modeling. Rainfall data from the semiarid USDA-ARS WGEW are used to investigate the spatial characteristics of thunderstorm rainfall in southeast Arizona and to develop a daily thunderstorm rainfall generator.

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Methods

This study was involved in two major tasks: the first part was the characterization of the spatial patterns of thunderstorm rainfall, which were derived form the statistical analyses on the 40-year WGEW summer rainfall records. The second part was the developments and calibrations of a daily thunderstorm rainfall generator based on the statistical characteristics derived from previous step. The following sections describe the methods in more details.

Characterization of the thunderstorm rainfall patterns

The first objective of this research is to identify the spatial characteristics of the daily summer thunderstorm rainfall on WGEW and TW in order to provide information for the construction of the thunderstorm generator. The spatial characteristics of the thunderstorm rainfall were clarified through the analyses of the historical rainfall events recorded by a dense rain gage network on WGEW and TW. Several distinct characteristics of thunderstorm rainfall were examined, which include: distribution of storm center location, distribution of maximum rainfall depth within a storm cell, shapes and orientations of storms, relation between maximum rainfall depth and storm coverage, and transition probability.

Developments and calibrations of the daily thunderstorm rainfall generator

A stochastic daily thunderstorm rainfall generator was constructed based on the statistical characteristics derived from the previous analysis. The rainfall generator involved the following steps: 1) Generation of a dry/wet sequences using the transition probability derived above, 2) Generation of locations of storm center using derived distribution, 3) Generation of maximum rainfall depth within a storm cell using derived distribution 4) Generation of storm coverage using relation of maximum depth and storm coverage derived above, and 5) Computations of rainfall depth on rain gage location using exponential spread function spreading outward from the storm center.

The simulated results were compared with observations and several adjustments of parameters were made to reduce the differences between simulations and observations.

Events and Data Characteristics

WGEW is located in southeast Arizona and has an area of 148 km^2. It ranges in elevation from 1650 m in the east to 1200 m in the west. The rain gage network consists of 93 weighing bucket recording gages, which record cumulative depth of precipitation on a continuous time base. Location of WGEW and the rain gage network on WGEW are shown in Figures 1 and 2.

Figure 1. Location of WGEW.

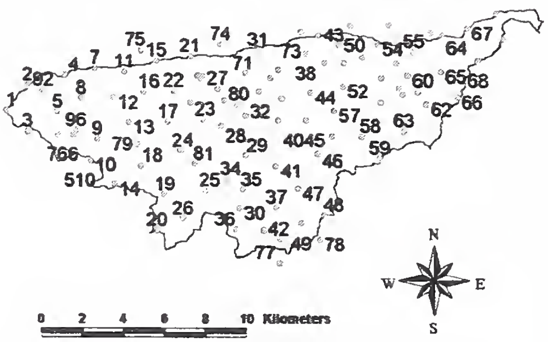

Figure 2. Rain gage network on WGEW.

The comparison study is limited to single events that only one storm event was recorded during a day by event related rain gages. The rain gage network consisted of 93 weighting recording type gages that recorded cumulative depth of precipitation on a continuous time base. The summer (July - September) rainfall records from 1960-95 were converted into an Access database and the selected events were retrieved from the database

Results

Results from this research consist of two parts: the first part is the statistical characteristics of the thunderstorm rainfall in southeast Arizona from the analyses of the rainfall data and the second part is the results of the simulations from the constructed daily rainfall generator. The following articles discuss the results separately.

Statistical characteristics of thunderstorm rainfall

In order to construct a stochastic daily thunderstorm generator, the statistical characteristics of thunderstorm rainfall are examined to provide information for the generator. This study examined several properties of the thunderstorm rainfall, which include: storm occurrence, spatial patterns of storm centers, distribution of maximum rainfall depth within a storm cell, storm shapes and orientations, and relationships between storm coverage and maximum rainfall depth. The following sections describe the approach applied to identify these characteristics.

Storm occurrence

Monthly and bi-weekly transition probabilities (P(D/D) and P(W/W)) and probability of wet (P(W)) at each gage location were calculated. Table 1 presents the average probabilities of 93 gages. The average monthly P(D/D) and P(W) of July and August have no significant differences. On the other hand, the average bi-weekly P(W/W) and P(W) have significant higher values comparing to other periods. The bi-weekly probabilities suggested that the second half of July (July 16-31) is the wettest period in summer. Hence, transition probability was computed on a bi-weekly basis.

Since a bi-weekly simulation period is appropriate, the bi-weekly transition probabilities and probability of wet of the entire watershed were calculated. Table 2 presents the bi-weekly transition probabilities and probability of wet for the entire watershed. The results indicate the storm occurrence has a higher frequency during the last two weeks of July and first two weeks of August than other wet periods, which is consistent with previous studies (Rodriguez et al. 1987).

Table 1. Average monthly and bi-weekly transition probabilities and probability of wet (in percentage) from 93 gages.

Period	Jul	Aug	Sep	Jul 1-15	Jul 16-31
P(D/D)	76.10	76.78	86.63	79.86	68.67
P(W/W)	47.29	40.14	36.48	45.27	51.86
P(W)	23.90	23.22	13.37	25.05	39.06

Period	Aug 1-15	Aug 16-31	Sep 1-15	Sep 15-30
P(D/D)	75.59	80	83	93
P(W/W)	42.98	36	35	35
P(W)	31.28	24	22	11

Table 2. Bi-weekly transition probabilities and probability of wet (in percentage) for the entire watershed.

Period	P(W)	P(W/W)	P(D/D)
Jul 1-15	56	69	61
Jul 16-31	79	82	35
Aug 1-15	67	76	51
Aug 16-31	55	65	61
Sept 1-15	51	67	66
Sept 16-29	25	56	86

Spatial patterns of storm centers on WGEW

In order to identify the spatial patterns of storm centers on WGEW, the locations of storm centers from each selected events were derived and aggregated to a map, which is shown in Figure 3. The nearest-neighbor analysis (NNA) (Davis 1986) was performed to identify the spatial patterns of points on a map.

The NNA compares characteristics of the observed set of distances between pairs of closest points with those that would be expected if the points were randomly placed. The characteristics of a theoretical random pattern can be derived from the Poisson distribution. If the edge effect of the map is ignored, the expected mean distance between nearest neighbors is

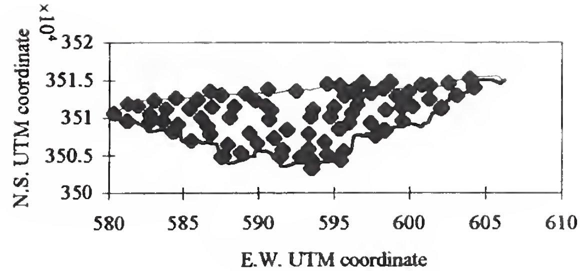

Figure 3. Storm centers derived from single events (374 events from 1970-90).

$$\bar{\delta} = \frac{1}{2}\sqrt{A/n} \quad (1)$$

where A is the area of the map and n is the number of points.

The expected and observed mean nearest-neighbor distances can be used to construct an index R to identify the spatial patterns of points on a map (Davis, 1986).

$$R = \bar{d}/\bar{\delta} \quad (2)$$

where $\bar{d}$ is the observed mean distance between nearest neighbors and R is the near-neighbor statistic, range from 0 for a distribution where all points coincide and are separated by distances of zero to 1.0 for a random distribution of points to a maximum value of 2.15 for a uniform distribution. Table 3 presents the results of the nearest-neighbor test. The results show all the R indices fall in the second category that is between 1 to 2.15 and represents a spatial pattern of random distribution, which is a spatial Poisson distribution.

Table 3. Nearest-neighbor analysis for WGEW storm centers.

	Area(sq. km)	n	delta	dbar	R
gages	153	93	640.38	1101.93	1.72
1970-75	162	107	614.47	826.70	1.35
1977-78	154	43	947.48	1285.22	1.36
1980-84	153	120	563.75	706.24	1.25
1985-90	152	104	604.55	881.60	1.46

Distribution of maximum rainfall depth within a storm cell

The rainfall depths are partitioned into several categories according to the amount of the depth. The count of each sub-category is calculated accordingly. The histogram of the depth in each sub-category verses the relative count is made to visualize the possible distributions and the Kolmogorov-Smirnov (K-S) test (Wilks 1995) is performed to test for the possible distributions. The maximum rainfall depth of each event was estimated as the maximum rainfall depth of each event from the observation. Table 4 presents the K-S test for maximum rainfall depth within a storm cell. The tests accepted both lognormal and Gamma distributions. Since the lognormal distribution requires less parameter estimations, hence was used for the generations of the maximum rainfall depth within a storm cell.

Table 4. K-S test for distribution of maximum rainfall depth within a storm cell.

Distribution	Jul 1-15	Jul 16-31	Aug 1-15
Gamma	>0.15	>0.15	>0.15
Lognormal	>0.15	>0.15	>0.15
Exponential	>=0.025&<=0.01	>=0.025&<=0.01	<0.01

Distribution	Aug 16-31	Sept 1-15	Sept 16-30
Gamma	>0.15	>0.15	>0.15
Lognormal	>0.15	>0.15	>0.15
Exponential	<0.01	<0.01	>=0.025&<=0.01

Storm orientations and shapes

The lengths of the major (a) and minor (b) axes and orientations were measured directly from the derived rainfall surfaces. Forty-eight events from the derived rainfall surfaces that had storm centers inside the watershed boundary were selected for the analysis. The ratio $r = a/b$. Table 5 presents the summary statistics of storm orientation and ratio of the lengths of the major to the minor axes. The mean of the ratio of the major to the minor axes is 1.54, which indicates the shape of storm on WGEW is elliptical rather than circular (consistent with Fogel and Duckstein 1969). As for orientations, the K-S test was performed to test for the possible distributions (Table 6).

Table 5. Summary statistics of storm orientation and ratio of the lengths of the major to the minor axes.

	ratio	orientation
mean	1.54	91.40
Std	0.37	38.27
skewness	0.96	0.06
min.	1.08	0
max	2.50	170
count	39	48

Table 6. K-S test results for the distribution of the storm orientations on WGEW.

Distribution of null hypothesis	P-value
Gamma distribution	>0.15
Lognormal distribution	>0.15
Normal distribution	>0.15

Relation of the maximum rainfall depth within a storm cell and storm coverage

A scatter plot of the storm coverage verses corresponding maximum storm depth using logarithmic scale (Figure 4). The plot shows a linear trend between the depth and coverage after a logarithmic transformation. A regression analysis was performed to obtain the linear relationship between the depth and storm coverage. The R^2 is 0.46 and the slope is 0.93. From the regression, the storm coverage was expressed as $A = 3.18d_{max}^{0.93} + \varepsilon$, where A is the storm coverage in km^2, d_{max} is the maximum rainfall depth within a storm cell in mm and ε is the error term. The error term is obtained from the error between the prediction from the regression and observation.

Stochastic daily thunderstorm generation

The following section describes the models and algorithms for the various components of the rainfall generator based on the derived statistical characteristics.

Precipitation occurrence

The method uses a two-state Markov chain to generate the number and distribution of precipitation events. The six bi-weekly transition probabilities were calculated and used to provide a transition from one period to another. Random sampling of the bi-weekly distribution is then used to determine the occurrence of a wet or dry day probabilities.

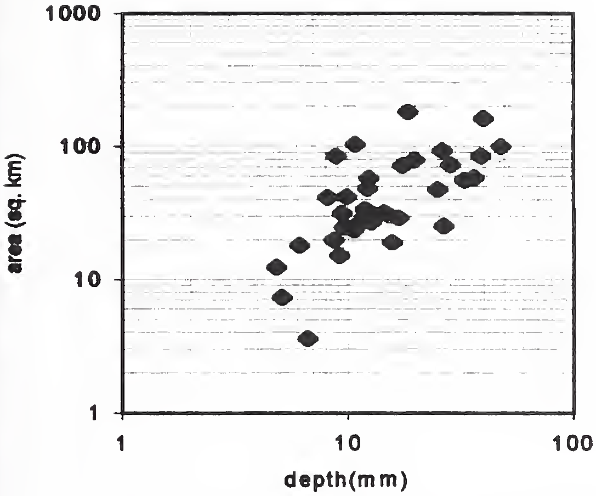

Figure 2. Maximum rainfall depth within a storm cell vs. storm coverage in logarithmic scale.

Precipitation depth

A lognormal distribution is used to represent the maximum precipitation depth within a storm cell. The form of this equation is

$$x = \frac{\log X - \mu}{s} \tag{3}$$

where x is the standard normal deviate, X is the raw deviate, and μ and s are the mean, standard deviation of the raw deviate after a logarithmic transformation, respectively. The mean and standard deviation of the logarithmic daily depths were calculated for each two weeks. Then, to generate a daily depth for each wet day occurrence, a random normal deviate is drawn and the raw variate, X (daily depth), is calculated using Eq. 3.

Storm coverage

The storm coverage is calculated using the corresponding precipitation depth and a random error term. The form of this equation is

$$A = 3.18X^{0.93} \pm \varepsilon \tag{4}$$

where A is the storm coverage in km^2, X is the generated precipitation depth in the storm center (mm) and ε is the random error introduced from the predicted errors of regression of storm coverage and precipitation depth. A uniform variate between 0 and 1 is generated to introduce the random error. The random error is added to the deterministic equation if the random variate is greater than 0.5, otherwise the random error is subtracted from the equation. Then, the storm coverage is calculated using Eq. 4.

Storm orientation

A normal distribution is used to represent the orientation of the major axis of the elliptical shape of the cell. The form of this equation is

$$x_o = \frac{X_o - \mu_o}{s_o} \tag{5}$$

where x_o is the standard normal variate, X_o is the raw variate, and μ_o and s_o are the mean, standard deviation of the raw deviate, respectively. The mean and standard deviation of orientations were calculated in previous chapter. To generate an orientation for each wet day occurrence, a random deviate x_o is drawn and the raw variate, X_o (orientation), is calculated using Eq. 5.

Storm cell axes

The lengths of a generated elliptical storm cell axes (a and b) can be calculated from the storm coverage A. Assuming the ratio of the major to minor axes is r, the form of this equation is

$$A = \pi r b^2 \tag{6}$$

where A is the storm coverage of a wet day event in m^2 and b is the length of the minor axes in m, respectively. A uniform distribution is used to represent the ratio. The form of the equation is

$$r = \frac{R - \mu_r}{s_r} \tag{7}$$

where r is the standard uniform deviate, R is the raw variate, μ_r and s_r are the mean, and standard deviation of the raw variate, respectively. The mean and standard deviation of the ratio were calculated in the previous chapter. Then, to generate ratio for each storm, a random uniform deviate is drawn and the raw variate, R (ratio), is calculated using Eq. 7. Since a = Rb, a and b can be calculated using Eq. 6.

Storm center
A uniform distribution is used to represent the wet day event storm center locations. The form of the equation is

$$z = \frac{Z}{33750} \tag{8}$$

where z is the standard uniform deviate between 0 and 1, Z is the location of storm center. A rectangle simulation space covering the study area (WGEW) which consisted of 33,750 square cells with a resolution of 100m is used for simulations. An index ranges from 1~33,750 is assigned to each cell. To generate the location for each wet day occurrence, a random uniform deviate is drawn and the raw variate, Z (storm center location) is calculated using Eq. 8.

Precipitation depths within storm coverage
The precipitation depth at any location within a storm cell is calculated using a spread function. The spread function is a relation of relative precipitation depth at an arbitrary location with respect to the depth at the storm center and is usually related to the absolute distance between those two points. Two types of spread functions are evaluated in the model: linear and exponential (Fogel and Duckstein 1969) spread functions. The forms of the equations are

$$P_z = \frac{d-x}{d} P_{max} \quad \text{(linear)} \tag{9}$$

$$P_z = P_{max} \exp\left(-0.27\pi\left(\frac{d}{1600}\right)^2 \exp(-0.0264 P_{max})\right) \quad \text{(exponential)} \tag{10}$$

where P_z is the precipitation depth (mm) at z, x is the distance between the storm center and z, d is the distance (m) between the boundary of the storm coverage and the storm center along the direction of z and storm center and P_{max} is the precipitation depth (mm) at storm center. Parameters used in equation 4.1.8 are adapted from Fogel and Duckstein (1969). To calculate the precipitation depth at any location within a storm cell, the distance between x and z is obtained, and the precipitation depth P_z is calculated using Eqs. 9 and 10, respectively.

A Fortran code was developed to facilitate the thunderstorm generation. Due to page limitation, the results are summarized in the following sections. For more details, please contact the authors.

Summary and Conclusion

Summary

This research study examined the spatial characteristics of the daily summer thunderstorm rainfall in the southeast Arizona. The following statistical characteristics of daily thunderstorm rainfalls have been identified from an analysis of the WGEW data: the storm center locations on WGEW have a Poisson distribution, the maximum depth within a storm cell has a lognormal distribution, the shape of a storm cell is elliptical with an average major axis length to the minor axis length ratio of 1.55 and the orientation of a storm cell is primarily NW or NE. The storm coverage and the maximum rainfall depth within a storm cell have a linear relationship after a logarithmic transformation. Storm occurrences have higher frequencies during the last two weeks of July and the first two weeks of August than other wet periods and there was no significant trend of transition probability with elevation. The stochastic daily thunderstorm generator is able to produce the statistical daily thunderstorm rainfall characteristics on WGEW.

Conclusion

The daily thunderstorm rainfall generator provides a distributed thunderstorm generator for southeast Arizona. The research of the temporal variation during storms can be further studied.

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Estimation of Ephemeral Streamflow Duration Using Temperature Methods in the Upper San Pedro River Basin, Arizona

Bruce Gungle

Abstract

A new method for interpreting the onset and cessation of streamflow in ephemeral streams using temperature sensor data is being utilized in the Sierra Vista subwatershed of the Upper San Pedro River Basin. Previous detection methods involved a moving standard deviation window technique and visual inspection of thermographs. The method presented here uses a rapid temperature drop greater than a designated threshold value to indicate flow onset and the following low temperature inflection point to indicate flow cessation. The temperature-drop threshold value is dependent on the mean thermal wave amplitude preceding the temperature drop. The amplitude is a function of the sensor burial depth and the antecedent soil water content in the sediments. The temperature-drop – low-temperature inflection point method was tested using a sensor buried 30 to 33 centimeters below the streambed surface and 10 meters downstream from a U.S. Geological Survey streamflow-gaging station in an ephemeral wash. Using the optimum temperature-drop thresholds of 0.25°C and 0.30°C, the method correctly identified 85 percent of all flows and had an 8 percent false positive detection rate. The average timing error of flow onset was 37 minutes with a standard deviation of 110 minutes, and the average timing error of flow cessation was 4 minutes with a standard deviation of 232 minutes.

Keywords: ephemeral flow, flow duration, temperature, thermograph

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Introduction

Identification of intermittent and ephemeral streamflow using temperature sensors has been accomplished by visual inspection of thermographs (Constantz et al. 2001) and by using statistical techniques, particularly standard deviation windows centered on the time of flow events (Blasch et al. 2002, Lawler 2002). Both methods are based on the premise that the presence of streamflow in an ephemeral channel reduces the amplitude of the diurnal thermal wave propagating through the sediments.

Using the visual inspection method, periods of flow are characterized by those sections of the thermograph where the amplitude of the diurnal temperature signal is visibly dampened (Constantz 2001). When the in-stream temperature data are compared graphically to temperature data from a nearby site out of streamflow (benchmark thermograph) where little dampening has occurred, a flow signal is readily identifiable. In addition, during periods of flow there is generally a change in the shape of the residual wave form (stream thermograph minus benchmark thermograph).

The standard deviation method uses the standard deviation of the temperature signal over a moving window of 1 to 12 hours (Blasch et al. 2002) or a static window of 24 hours (Lawler 2002), and those standard deviation values that exceed a specified threshold value are interpreted as the onset and cessation of flow. Using an optimized, 6-hour, moving standard deviation window (0 percent false positive and 0 percent false negative detections) on data gathered at Rillito Creek in Tucson, Arizona, Blasch et al. (2002) found the average timing error was 95 minutes at flow onset and 310 minutes at flow cessation. Using a static, 24-hour standard deviation window with different summer and winter thresholds, Lawler (2002) was able to correctly identify

78 percent and 80 percent, respectively, of the periods of flow of the San Pedro River near Palominas, Arizona. This improved to 80 percent for both summer and winter periods using a daily amplitude threshold.

Methods

This study uses a new technique for the identification of flow onset and cessation that is suited for the small drainages and short ephemeral flows ($t < 24$ hours) that are common to most tributary stream channels in the study area—the Sierra Vista subwatershed of the Upper San Pedro River Basin in southeastern Arizona. In a previously dry streambed, the onset of flow can be identified by a sharp drop in temperature of 0.20°C or more over 15 minutes (Figure 1). The minimum temperature drop required to identify flow onset will vary with the mean amplitude of the thermal wave. The thermal wave amplitude is a function of sensor burial depth (deeper burials will show a smaller temperature drop at flow onset) and antecedent soil water content in the sediments surrounding the sensor (the greater the volume of antecedent water the smaller the temperature drop at flow onset). Flow cessation is identified by the low-temperature inflection point that follows the sharp temperature drop (Figure 1).

The utility of this method for determining streamflow was tested over the summer of 2002 using a TidbiT© temperature sensor that had a precision of 0.1°C buried 10 meters downstream from a USGS streamflow-gaging station in Greenbush Draw. Greenbush Draw is an ephemeral tributary of the San Pedro River tributary at the upper end of the Sierra Vista subwatershed. On May 2, 2002 the sensor was buried approximately 33 centimeters below the streambed surface, under 25-27 centimeters of clay with some sand, and 6-8 centimeters of sand with some clay. Thirteen ephemeral flow events were recorded at the Greenbush Draw gaging station during the summer of 2002, and the sensor was recovered approximately 30 centimeters below the streambed surface on September 17, 2002. The temperature sensor recorded every 15 minutes.

The temperature-drop – low-temperature inflection point method was subsequently applied to temperature data previously gathered from other ephemeral streams in the Sierra Vista subwatershed (Figure 2). Results of one such application—the Woodcutters Canyon drainage on the west side of the subwatershed—are presented below and in Table 2.

Results

The optimal temperature-drop threshold to be used for flow detection is selected on the basis of maximizing correct flow detection while minimizing false negative and false positive detections of flows, and minimizing the time between predicted and actual flow onset and cessation. A false negative detection is one in which flow occurs but is not detected. A false positive detection is the reverse; flow does not occur, but flow is indicated using the temperature-drop – low temperature-inflection point method. For the purposes of this study, streamflow-gaging station data are used to determine when flow occurs.

The data are compiled in Table 1, where time of temperature drop, time of low-temperature inflection point, and elapsed time between the two are compared to onset of flow, cessation of flow, and duration of flow, respectively. Temperature-drop thresholds tested ranged from 0.20°C to 1.0°C. Using temperature-drop thresholds of 0.20°C, 0.25°C and 0.30°C, 11 of the 13 flows (85 percent) were correctly identified (15 percent false negative detection). The percent false positive detections for the 0.20°C temperature drop threshold is 69 percent. This decreases to 8 percent (1 false detection out of 12 flows detected) for both the 0.25°C and 0.30°C thresholds. Thus, for purposes of flow detection, the 0.25°C and 0.30°C thresholds are optimum. The mean difference between the time of actual flow onset and the time of temperature drop is low (37 minutes) as is the standard deviation (less than 2 hours). The mean difference between the time of flow cessation and the time of the low-temperature inflection point is very low (4 minutes), but the standard deviation is high (nearly 4 hours).

Field application

Table 2 presents the results of this interpretive technique applied to Woodcutters Wash on the western side of the Sierra Vista subwatershed of the Upper San Pedro River Basin (Figure 2). Approximately 3 miles separate the upstream and middle sensors, and approximately 5.5 miles separate the middle and downstream sensors. Nearly half of the flows were recorded at more than one site, and three of the flows, including those resulting from the synoptic scale weather system of April 2001, were recorded at all three sites. More flows were recorded at the upstream and middle sites than at the downstream site. The upstream and middle sites, however, are within the city of Sierra Vista. Hence, urban runoff may play a role in the large number of flows recorded at these two sites that are not recorded at the downstream site.

Table 2 also highlights a number of the difficulties encountered when using the temperature-drop – low-temperature inflection point technique to interpret stream flow. First, uninterpretable data such as occurred with the middle sensor between July 29, 2001 and August 17, 2001 are the result of the sensor being pulled up to the surface during a flow event. By anchoring the sensor at the desired depth, most such occurrences can be avoided.

Second, modest localized precipitation events at road crossings can result in flow indicated at a sensor installed downstream from the crossing. Because such events represent small magnitude street drainage rather than large magnitude flow events, they give an inaccurate representation of what is occurring along the entire wash. This is the case with the middle sensor, and the two flows recorded at only that site may have been the result of localized urban drainage rather than true wash flow. This problem is resolved by moving the sensor upstream from the road.

Burial depth, which affects the mean amplitude of the thermograph, is a third factor that can affect flow interpretation. Although the control data from Greenbush Draw are for a sensor buried 30 to 33 centimeters below the streambed surface, most of the sensors throughout the subwatershed have been buried shallower than this. As a result, the mean amplitude of the thermal wave in most subbasin thermographs is commonly greater than at the Greenbush Draw research site, and a larger temperature-drop threshold is required when screening data to minimize the number of false positives. A series of sensors buried at multiple depths downstream from the Greenbush Draw gaging station have recently been installed for the purpose of quantitatively determining the optimum temperature-drop thresholds for various sensor depths and mean amplitudes.

Conclusion

Constantz et al. (2001), Lawler (2002), and Blasch et al. (2002) have demonstrated that temperature can be used to estimate the occurrence and timing of ephemeral and intermittent flow events. The temperature-drop – low-temperature inflection point method offers a simpler method of ephemeral streamflow analysis than does the standard deviation method. It requires fewer parameters (two: a temperature drop that exceeds a threshold value, and a following low-temperature inflection point) and fewer and less complicated computations (subtraction to determine flow duration) than does the moving standard deviation window method (five parameters; determination of various thresholds and filters, approximation of the thermal and hydraulic parameters for the site, and the calculation of the moving standard deviation). It is not clear at this time, however, how the two methods compare in detecting the onset and cessation of flow.

In addition, it is unlikely that the temperature-drop – low-temperature inflection point method can be used to indicate the onset of long term flows that are not the result of a significant precipitation event (e.g., Lawler 2002), nor the cessation of flows that last for more than 24 hours. Combining this method with the visual inspection method may prove effective for instances of longer term flow (more than 24 hours) in ephemeral channels. Also, the temperature-drop – low-temperature inflection point method does not appear to be effective in separating discrete flows that occur in rapid succession (within approximately 24 hours or less) on the basis of the Greenbush Draw data. In situations where sensors can be easily installed at the necessary depths for optimization, the moving standard deviation method of Blasch et al. (2002), optimized to detect all flows with no false detections, may be effective in combination with this method.

Acknowledgments

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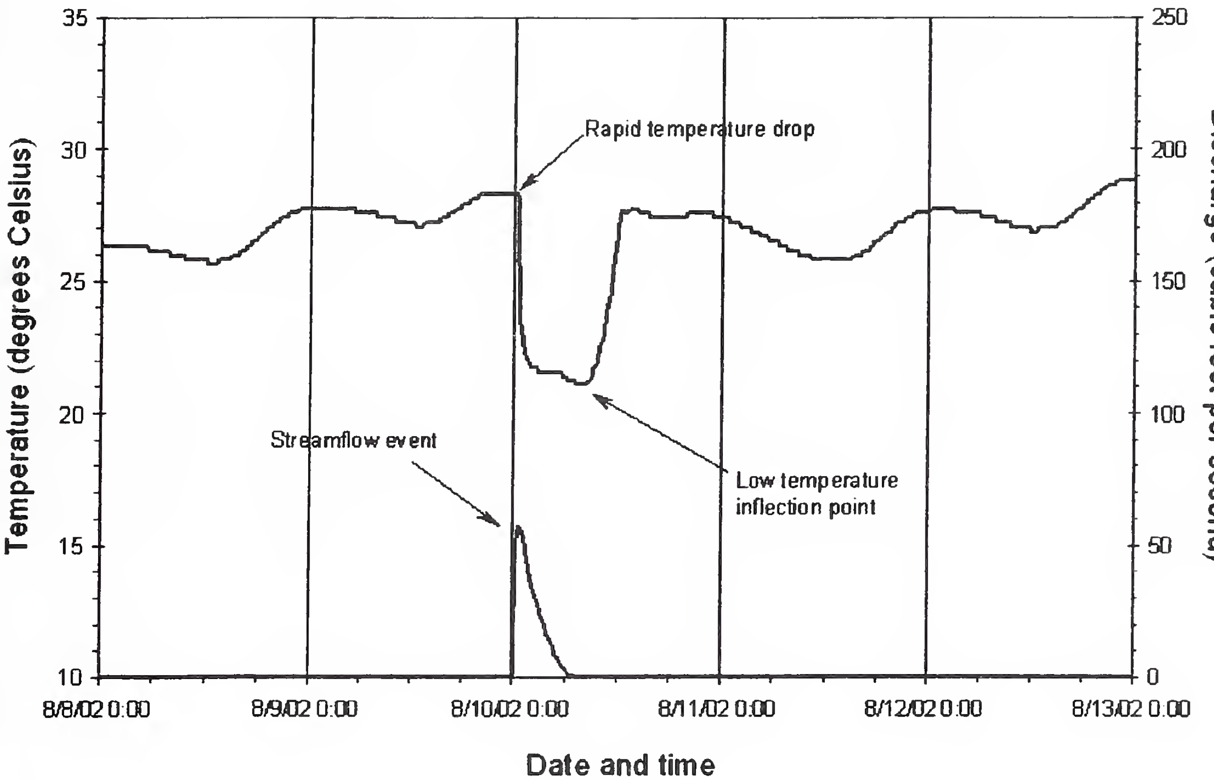

Figure 1. Greenbush Draw thermograph and streamflow-gaging station discharge record for 5 days in August 2002. The temperature sensor was 10 meters downstream from the streamflow-gaging station and buried 30–33 centimeters below the streambed surface.

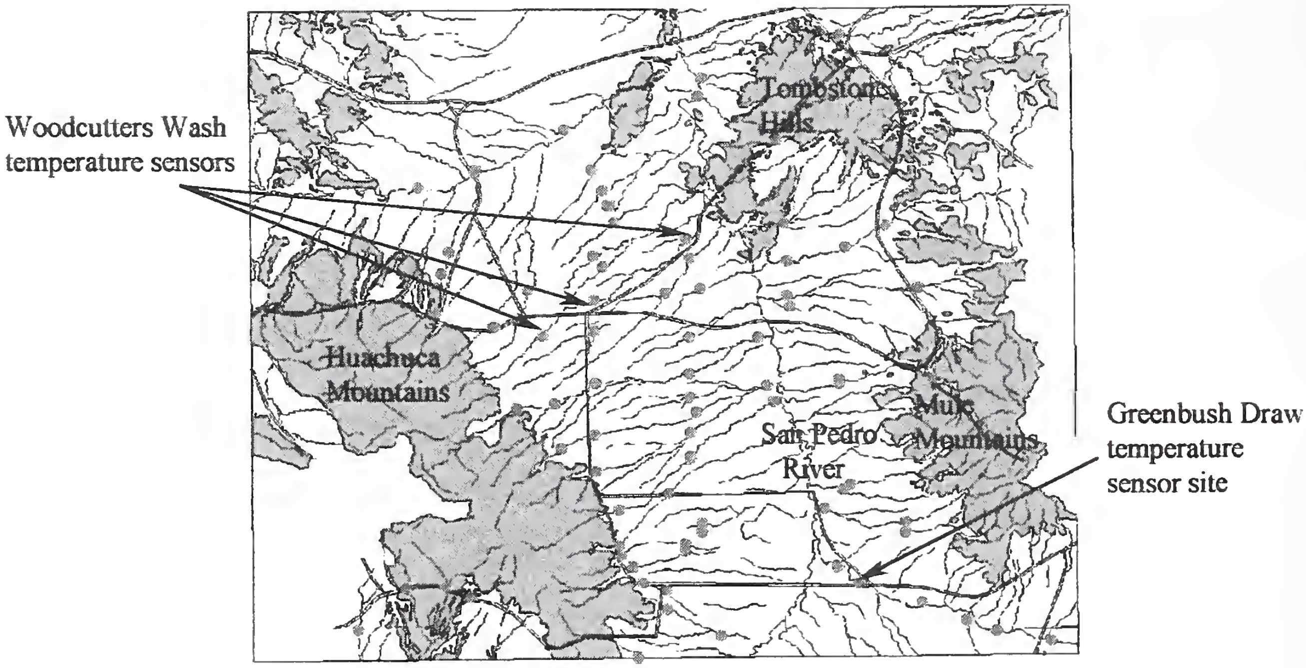

Figure 2: Sierra Vista subwatershed of the Upper San Pedro River Basin. Dots are location of temperature sensors.

Table 1. Temperature sensor flow detection at Greenbush Draw for various temperature drop thresholds compared to flow detection at Greenbush Draw gaging station. The temperature sensor is 10 meters downstream from the gaging station and 30-33 centimeters below the streambed surface. Values in bold are for the optimal temperature-drop threshold for flow detection (correct flow detection is maximized, whereas false negative and false positive flow detection and the time between predicted and actual flow onset and cessation are minimized).

Temperature drop threshold (°C)		0.20	**0.25**	**0.30**	0.35	0.40	0.45	0.50	0.55	0.60	0.65	0.70	0.75	0.80	0.85	0.90	0.95	1.00
Flows correctly identified	N:	11	**11**	**11**	9	8	8	8	7	7	7	7	6	6	6	6	6	6
(of 13 possible)	Percent:	85	**85**	**85**	69	62	62	62	54	54	54	54	46	46	46	46	46	46
False negative flow identification (flows missed / total actual flows)	Percent:	15	**15**	**15**	31	38	38	38	46	46	46	46	54	54	54	54	54	54
False positive flow identification (false flow identifications / total flows identified)	Percent:	69	**8**	**8**	10	0	0	0	0	0	0	0	0	0	0	0	0	0
Time of flow onset minus time of temperature drop (minutes)	Mean:	37	**37**	**37**	33	6	6	2	-4	-4	-4	-4	-7	-7	-12	-15	-15	-15
	Standard Deviation:	110	**110**	**110**	78	40	40	44	45	45	45	45	48	48	46	44	44	44
Time of flow cessation minus time of low-temperature inflection point (minutes)	Mean:	4	**4**	**4**	2	-34	-34	-34	11	11	11	10.7	-32	-32	-32	-32	-32	-32
	Standard Deviation:	232	**232**	**232**	251	243	243	243	224	224	224	224	212	212	212	212	212	212
Duration of flow minus time from temperature drop to low temperature inflection point (minutes)	Mean:	-33	**-33**	**-33**	-32	-39	-39	-36	15	15	15	15	-25	-25	-20	-18	-18	-18
	Standard Deviation:	231	**231**	**231**	204	222	222	220	185	185	185	185	166	166	170	169	169	169

Table 2. Flows interpreted using temperature sensors at three locations along Woodcutters Wash, upstream (left) to downstream (right). The reach covers approximately 13.5 kilometers (8.5 miles). Flows recorded at more than one sensor site are in bold. [MST, Mountain Standard Time; cm, centimeters; °C, degrees Celsius]

Woodcutters at 7th Street (upstream sensor) TidbiT serial number: 375010				**Woodcutters at Rt. 90 (middle sensor) TidbiT serial Number: 377793**				**Woodcutters at Moson Road (downstream sensor) TidbiT serial number: 377788**			
	Date and time of flow onset (MST)	Date and time of flow cessation (MST)	Flow duration (minutes)		Date and time of flow onset (MST)	Date and time of flow cessation (MST)	Flow duration (minutes)		Date and time of flow onset (MST)	Date and time of flow cessation (MST)	Flow duration (minutes)
1	**4/6/01 1:18**	**4/6/01 1:48**	**30**	**1**	**4/5/01 22:20**	**4/6/01 10:50**	**750**	**1**	**4/6/01 4:47**	**4/6/01 7:47**	**180**
2	**6/19/01 18:48**	**6/20/01 9:18**	**870**	**2**	**6/19/01 20:20**	**6/20/01 7:50**	**690**				
				3	6/25/01 13:30	6/25/01 15:30	120				
								4	6/26/01 11:00	6/26/01 16:45	345
				5	**7/7/01 13:30**	**7/7/01 14:45**	**75**	**5**	**7/7/01 14:15**	**7/7/01 14:30**	**15**
				6	7/9/01 18:45	7/9/01 20:15	90				
								7	7/16/01 15:30	7/16/01 15:45	15
8	**7/24/01 21:18**	**7/25/01 1:18**	**240**	**8**	**7/24/01 21:00**	**7/25/01 8:45**	**705**				
9	7/25/01 18:48	7/26/01 0:18	330								
10	**7/28/01 16:48**	**7/29/01 5:18**	**750**	**10**	**7/28/01 23:00**	**7/29/01 6:15**	**435**	**10**	**7/28/01 17:15**	**7/28/01 17:30**	**15**
11	8/5/01 18:48	8/5/01 20:48	120	*	NA 7/29/01 - 8/17/01 uninterpretable						
12	8/11/01 20:48	8/12/01 3:48	420	*	NA						
13	**8/13/01 19:18**	**8/14/01 1:48**	**390**	*	**NA**			**13**	**8/13/01 19:00**	**8/14/01 8:45**	**825**
14	8/16/01 19:48	8/16/01 21:18	90	*	NA						
15	8/17/01 20:18	8/18/01 4:18	480	*	NA			15	8/17/01 21:00	8/18/01 1:15	255
16	8/29/01 17:48	8/29/01 23:18	330	16	8/29/01 18:30	8/30/01 8:00	810	16	8/29/01 20:00	8/30/01 7:15	675
	Burial depth: 20 - 36 cm				Burial depth: 20 cm				Burial depth: 15-20 cm		
	Temperature drop threshold: 0.5 °C				Temperature drop threshold: 0.6 °C				Temperature drop threshold: 0.6 °C		

Hydrologic Characteristics of the Little River Experimental Watershed

David Bosch, Joe Sheridan, Randy Williams

Abstract

The USDA-ARS, Southeast Watershed Research Laboratory (SEWRL) in Tifton, Georgia has collected over 30 years of hydrologic and climatic data from the 334 km^2 Little River Watershed (LRW). The SEWRL collects hydrologic and water quality data representative of the Gulf-Atlantic Coastal Plain region of the southeastern United States. The LRW is typical of the heavily vegetated, slow-moving stream systems in the region. Hydrologic data are available from up to eight watersheds ranging in area from 2.6 to 334 km^2. LRW long-term hydrologic budgets have established that approximately 30% of the watershed precipitation leaves as streamflow. This includes both surface runoff and shallow groundwater flow that contributes to streamflow. Field studies indicate the surface runoff component varies from 7 to 20% of precipitation while shallow return flow varies from 3% to 22%. Peak flow and minimum flow characterizations using standard USGS procedures show considerable variability in the streamflow. Flow distribution curves and basic statistical characterizations of extreme events on the watersheds have been determined. Generalizations relating watershed yield to watershed drainage area along with expressions relating annual streamflow to annual precipitation illustrate differences between this region and other areas of the U.S. The SEWRL is currently testing predictive relationships and physically-based watershed models to simulate the streamflow in the region.

Keywords: streamflow, watersheds, runoff

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Erosion II

Comparison of the USLE, RUSLE1.06c, and RUSLE2 for Application to Highly Disturbed Lands

George R. Foster, Terrence E. Toy, Kenneth G. Renard

Abstract

RUSLE1.06c and RUSLE2, recently released erosion prediction models, are described. These land-use independent models are well suited for application to highly disturbed land. Cover-management subfactors make possible the land-use independence. Similarities and differences with the USLE are discussed.

Keywords: soil erosion, erosion prediction, erosion control, conservation planning, rainfall, overland flow

Introduction

The USLE, RUSLE1, and RUSLE2 are widely used to estimate rill and interrill erosion that occurs on overland flow areas. These equations apply where mineral soil is exposed to the erosive forces of raindrops and water drops falling from vegetation and surface runoff occurring as Hortonian overland flow. These equations share features proven in conservation planning over four decades.

Highly disturbed lands include construction sites, highways, reclaimed surface mines, landfills, military training sites, and similar lands where mechanical operations disturb the soil and vegetation to leave the land vulnerable to rill and interrill erosion. The disturbance period is often brief followed by an extended recovery where permanent vegetation develops. Cropland is a special case where a sequence of mechanical operations is periodically repeated. Rill and interrill erosion also occurs on wildlands, pasturelands, rangelands, and other undisturbed lands. These lands do not experience the mechanical disturbance common to cropland or highly disturbed lands. However, infrequent renovation to encourage forage production often involves mechanical disturbance. Extensive removal of vegetation by livestock and wild animal grazing and fire that removes vegetation and litter subject these lands to rill and interrill erosion.

The USLE (Universal Soil Loss Equation), released in the early 1960s, was developed for cropland (Wischmeier and Smith 1965). Later it was extended to other land uses (Wischmeier and Smith 1978, Dissmeyer and Foster 1980). RUSLE1 (Revised Universal Soil Loss Equation) was released in the early 1990s and evolved to the current RUSLE1.06c released in mid 2003 (Renard et al. 1997, USDA-ARS-NSL 2003). RUSLE1 is land-use independent and applies to any land use having exposed mineral soil and Hortonian overland flow. RUSLE2 was also released in mid 2003, and it too is land-use independent (USDA-ARS-NSL 2003).

The USLE is an index based, empirically derived model. RUSLE1 and RUSLE2 are hybrid models that combine index and process-based equations. RUSLE2 expands on the hybrid model structure and uses a different mathematical integration than does the USLE and RUSLE1.

Purpose of USLE, RUSLE1, and RUSLE2

The purpose of the USLE, RUSLE1, and RUSLE2 is to guide conservation planning. The equations are used to estimate erosion based on site-specific conditions for

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erosion control alternatives. The erosion estimates are judged against a criterion and those practices that meet the criterion provide satisfactory erosion control for the site. All three equations estimate average annual erosion. The usual soil conservation objective is to protect the soil from excessive erosion, even when the main objective is to control sediment leaving the site. Excessive erosion degrades the landscape, reduces soil productivity, increases the difficulty of establishing and maintaining vegetation, inconveniences mowing, and produces sediment that cause downstream damage.

The validity of a model is judged by how well it serves its intended purpose (Toy et al. 2002). Accuracy is important, but most important is the conservation planning decision. Two models that yield the same conservation decision perform equally well. Other considerations are resources required to use a model, availability of input values, ease of use, and robustness. A model may give accurate estimates, but if it is difficult to use, users often sacrifice accuracy for ease of use. Demonstrating that a particular erosion model is more accurate than another is very difficult given the variability in and limited extent of the erosion research data, especially for highly disturbed and wild lands.

Neither the USLE, RUSLE1, or RUSLE2 should be used solely to evaluate overall site environmental or ecological well-being. These equations estimate soil erosion rates, nothing more. The user interprets the erosion estimates according to the user's purpose. Application of these models to wildlands has been criticized. Sometimes the criticism is misdirected to the models rather than to how erosion estimates are used. Erosion rate, even if known with 100% accuracy, is not the sole indicator of ecological well being.

Basic Equations

Sediment production

Detachment and transport are combined in these models as a sediment production term. The equation for sediment production on a uniform slope is:

$$a_i = r_i k_i l_i s_i c_i p_i \qquad (1)$$

where: a_i = erosion rate (spatial average for the slope length λ) for the *ith* storm, r_i = storm rainfall erosivity, k_i = soil erodibility factor, l_i = slope length factor, s_i = slope steepness factor, c_i = cover-management factor, and p_i = support practices factor. Storm erosivity r (EI) is the product of the storm's energy and its maximum 30-minute intensity. Storm energy is closely related to storm amount. The EI variable captures the two most important rainstorm characteristics that determine erosivity, storm amount and a measure of peak intensity. Soil erodibility k is erosion from the unit plot per unit erosivity. A unit plot is 22.1 m long on a 9 percent steepness, periodically tilled up and down slope to break the soil crust and to control the weeds, and maintained in continuous fallow for several years. Time is needed for the effects of previous land use to dissipate and to measure erosion from both moderate and large storms. The unit plot is used to empirically determine soil erodibility as a function of inherent soil properties where the effects of land use have been removed. The product *lscp* adjusts erosion for the unit-plot condition, which is the product *rk*, to erosion for the actual field condition.

Deposition

The USLE does not compute deposition. RUSLE1 and RUSLE2 compute deposition on concave slopes, at dense vegetative strips, in terrace channels, and in sediment basins using process-based equations for transport capacity and deposition. The equation for transport capacity is:

$$T_c = k_t q_p \sin(\theta) \qquad (2)$$

where: T_c = transport capacity, k_t = a transport coefficient that decreases as hydraulic resistance increases from ground cover, vegetative retardance, and surface roughness, q_p = characteristic runoff rate, and θ = slope angle. The product $q_p\sin(\theta)$ is directly proportional to runoff's total shear stress raised to the 1.5 power. Shear stress is divided into two parts, the part dissipated on ground cover, vegetation, and surface roughness and the part that erodes and transports sediment. The term k_t reduces total shear stress to the shear stress active in sediment transport.

The equation used to compute deposition is:

$$D = (V_f/q_p)(T_c - g) \qquad (3)$$

where: D = deposition rate, V_f = fall velocity of the sediment, and g = sediment load. A single deposition coefficient is used in RUSLE1 to represent the sediment. This coefficient varies with soil texture so that RUSLE1 computes deposition as a function of soil texture. The coefficient is not varied along the slope as deposition enriches the sediment load in fines. RUSLE2 divides the sediment into five particle classes based on soil texture. RUSLE2 treats each particle class individually with interaction among the classes. RUSLE2 computes deposition as a function of soil texture and how deposition changes sediment characteristics along the slope, which is turn affects computed deposition. RUSLE2 computes an enrichment ratio for the sediment leaving the end of the slope. Enrichment ratio is the ratio of specific surface area of the sediment to specific surface area of the soil subject to erosion.

Integration of equation 1

USLE

The discovery that erosion is linearly proportional to storm erosivity facilitated the development of the well known USLE:

$$A = RKLSCP \tag{4}$$

where: A = average annual erosion, R = erosivity factor, K = soil erodibility factor, LS = topographic factor, C = cover-management factor, and P = support practices factor. Average annual values are used for each factor to compute erosion.

Only the C-factor value results from a temporal integration as:

$$C = \sum \left(f_j c_i \right) \tag{5}$$

where: f_j = the temporal distribution of erosivity during the year and j = an index for a "crop stage" time step. Experimental erosion data were used to determine cover-management factor (c_j) values by crop stage period (soil loss ratios, Table 5, AH537, Wischmeier et al. 1978). Crop stage periods mark crop development and events like primary tillage, seedbed preparation, and harvest that change cover-management conditions. Values for C are increased when the most erosive period coincides with the period when cover-management conditions are most vulnerable to erosion. Once computed, C factor values for an erosivity distribution zone are placed in tables for use in equation 4.

Erosivity values for the USLE and RUSLE1 were determined from 22-years of weather data from about 1935 to 1957 for the eastern U.S. Erosivity values were computed for storms equal to and greater than 12.5 mm and were summed for each year. The average annual value for erosivity is the R-value used in equation 4. Mapped R-values provide an erosivity index by location. Erosivity varies during the year. The temporal erosivity distribution, f, was empirically determined for half-month periods and mapped by zones in the U.S.

Experimental data were also used to determine LS-factor values for slope length and steepness and P-factor values for support practices. Soil erodibility K-factor values were obtained by plotting erosion from a particular soil in the unit-plot condition versus storm erosivity. The slope of this line through the origin is the soil erodibility K-factor value for that soil. The USDA-Natural Resources Conservation Service (NRCS) assigned and cataloged K-factor values for many soils across the U.S. With the exception of the interaction between erosivity and cover-management, all USLE factors are independent of each other.

RUSLE1

RUSLE1 uses equations to compute half-month values for the cover-management factor. All RUSLE1 versions until the recently released RUSLE1.06c computed half-month values for soil erodibility for the eastern U.S. These RUSLE1 versions compute erosivity-weighted values for K and C using equation 5. RUSLE1.06c assumes a constant K-factor value. RUSLE1 considers a limited interaction among the factors in equation 1. The relationships for LS and ground cover effect vary with the ratio of rill to interrill erosion, which in turn varies with soil texture, slope steepness, and cover-management variables.

RUSLE2

RUSLE2 computes average annual erosion using:

$$A = S \sum \left(r_k k_k l_k c_k p_k \right) \tag{6}$$

where: k = index for day of the year. The mathematical integration in RUSLE2 differs fundamentally from that in the USLE and RUSLE1. Average annual factor values are multiplied in the USLE and RUSLE1. Instead, RUSLE2 multiplies the factor values for each day to estimate daily erosion values, which are summed for average annual erosion. This difference results in as much as a 20% difference in average annual erosion values between RUSLE2 and the USLE and RUSLE1. RUSLE2 uses basic variables rather than the *RKLSCP* factors to compute erosion. Although RUSLE2 does not use these factors to compute erosion, it computes values for them and demonstrates their interaction. Which formulation is best?

RUSLE2 is mathematically superior to the USLE and RUSLE1. Also, RUSLE2 is much more powerful than either the USLE or RUSLE1 and uses better relationships to compute factor values. Use RUSLE1.06 for applications where the USLE equation structure, equation 4, is desired. Do not use the USLE because the RUSLE1.06c equations are superior to the USLE equations. Do not use RUSLE1.06b or older versions of RUSLE1 because RUSLE1.06c was modified to give values comparable to RUSLE2 values (USDA-ARS-NSL 2003).

Recent Developments

Erosivity, precipitation, and temperature

Input climate values for monthly erosivity, precipitation, and temperature were developed from modern climate data from 1960-1999. Fifteen-minute precipitation data were analyzed to determine erosivity density values. Erosivity density is the ratio of monthly erosivity to monthly precipitation. Erosivity density varies spatially and temporally. Erosivity density is higher in the southern U.S. than in the northern U.S. Summer erosivity density is greater than winter erosivity density in the eastern U.S. The converse is true along the most western part of the U.S. Erosivity density does not vary with elevation up to about 3,000 m, the extent of the data. Erosivity density was mapped throughout the continental U.S. Monthly erosivity density is multiplied by monthly precipitation to obtain monthly erosivity at a location. Monthly precipitation and temperature values for any U.S. location are available in the NRCS PRISM database. The PRISM precipitation and temperature values vary spatially in mountainous areas. The new erosivity values are much better than previous values.

RUSLE2 uses 10 yr-24 hr precipitation amounts to compute runoff. A new map of 10 yr EI values for the eastern U.S. was developed for use in RUSLE1.06c.

Soil erodibility

The NRCS assigned K-factor values cannot be used for mixed soils typical of highly disturbed lands. The RUSLE2 modified soil erodibility nomograph is used to estimate K-factor values for mixed soils and subsoils where the surface layer has been stripped away without mixing the soils. The effect of the soil structure in the standard nomograph (Wischmeier et al. 1978) is inconsistent with accepted science regarding the relationship between erosion, texture, and structure.

Topography

The S factor in RUSLE1 and RUSLE2 is based on a much larger data set than the S factor in the USLE. The RUSLE relationship better fits data from highly disturbed lands than does the USLE relationship.

The exponent n in the slope length L factor $(\lambda/22.1)^n$ in RUSLE1.06c varies with land use and soil texture. This exponent in RUSLE2 is computed with equations that are functions of slope steepness, soil biomass, soil consolidation, ground cover, and soil texture.

Cover-management

Cover-management represents how cultural management practices that involve mulch, vegetation, and soil condition affect erosion.

Subfactor method

Both RUSLE1.06c and RUSLE2 use subfactors to compute temporal cover-management factor values. Erosion occurs when erosive agents exert physical forces on the soil that exceed internal resisting forces that hold the soil particles in place (Toy et al. 2002). Vegetative cover above the soil surface; litter, stones, and other material on the soil surface; and surface roughness reduce erosive forces. Physical, chemical, and biological properties modified by land use and land use condition affect soil resistance to erosion. The subfactors capture how major variables affect these external and internal forces.

A strength of RUSLE1 and RUSLE2 is that they are land-use independent, made possible by the subfactor method. Both models treat land use and land use condition as a continuum. A freshly graded and seeded surface mine reclamation site is like a recently tilled and seeded cropped field. Over time, the site evolves to a pasture, range, or wild land like condition. Land use in western South Dakota alternates between crop-land and rangeland as farming economics shift. Previous land use affects erosion with the current land use. A part of a military training ground is undisturbed like rangeland at Fort Hood, Texas or forestland at Fort Benning, Georgia. Another part of the grounds is highly disturbed like a construction site with a very rough soil. All sorts of conditions exist between these extremes. An erosion prediction model derived from cropland data and another derived from rangeland data are unlikely to give common estimates at the boundary between land use conditions. Land users may not know the correct erosion estimate, but they recognize and question inconsistent erosion estimates. Both RUSLE1.06c and RUSLE2 provide the expected consistency.

The subfactor method was originally developed to extend the USLE to undisturbed land (Wischmeier 1975). The USLE subfactor method considered how cover-management conditions above the soil surface, on the soil surface, and within the soil surface affected erosion. Values for this procedure are given in Table 10, AH537 (Wischmeier and Smith 1978). These values give poor results and should not be used. Table 10, AH537 does not consider surface roughness, does not represent properly soil biomass as a function of vegetation type or production level, and does not represent properly the combination of rock, litter, and other ground cover. Also, Table 10 cannot be used for mechanically disturbed land.

The subfactor variables used in both RUSLE1.06c and RUSLE2 include percent canopy cover and fall height; surface roughness; ground cover provided by stones, litter, basal area, live vegetation touching the ground, and other material on the soil surface; plant community type; average annual plant production; and time since the soil has been mechanically disturbed. Plant community type determines the ratio of effective root biomass to average annual above ground plant production. The overlap of canopy over ground cover and the overlap of litter over stones are taken into account. The subfactor equations in RUSLE2 are more detailed than those in RUSLE1.06c. For example, the relationships in RUSLE2 consider the distribution of roots with depth and where soil-disturbing operations distribute material within the disturbance depth.

The time invariant C factor method in RUSLE1 uses effective average annual input values to compute a C-factor value. RUSLE2 and the time variant C-factor method in RUSLE1 computes the accumulation of a litter layer on the soil surface and the accumulation of soil biomass. Sources of soil biomass include live and sloughed dead roots, plant material moved into the soil by insects, and material mechanically incorporated into the soil. These biomass pools are a function of precipitation and temperature at a location, plant production level, litter fall, root sloughing, decomposition characteristics of the biomass, and burial characteristics of mechanical operations.

Soil loss ratio values in Table 5, AH 537 (Wischmeier and Smith 1978) and literature values for conservation tillage were used to partially calibrate the subfactor equations. Literature values for the effect of rangeland conditions on erosion, including data collected by the USDA-Agricultural Research Service (ARS) in the Walnut Gulch watershed, Nevada Test Site, and other locations were also used. The WEPP rangeland data were used to develop values for the ratio of effective root biomass to annual plant production. Experiments were conducted at more than 10 locations across the western U.S. Analysis of the ARS-NRCS Range Study Team data was attempted with limited success.

The procedure was to back calculate the effective below ground biomass values using measured erosion and measured values for other variables in the subfactor equations. Measured root biomass values do not work well for estimating effective below ground biomass. Collecting and accurately measuring root biomass is very difficult, not all roots are equally effectiveness in reducing erosion, and research has not determined the relation of erosion to root characteristics. Also, the presence of organic compounds from decomposition of sloughed (dead) roots and litter brought into the soil by insects is not represented by measured root biomass.

C factor values for construction sites

Values for the C and P factors are available in various technical publications for applying the USLE to construction sites. These values are quite inconsistent, which means that some of them are erroneous and should not be used. RUSLE1.06c represents the current state of scientific knowledge and research data (Toy and Foster 1998). Comparable relationships are used in RUSLE2. A project is underway with the Wisconsin Department of Natural Resources to further refine RUSLE2 for application to construction sites.

Support Practices

Support practices include contouring (ridging), barriers (vegetative strips, silt fences), flow interceptors (diversions), sediment basins, and subsurface drainage. These practices affect erosion by affecting runoff. The 10 yr EI in RUSLE1 and 10 yr-24 hr precipitation in RUSLE2 are used to compute runoff using the NRCS curve number method. The curve number is related to a cover-management condition index in RUSLE1 and is computed in RUSLE2 with equations that are functions of ground cover, soil biomass, surface roughness, and soil consolidation. The effectiveness of contouring is computed as a function of runoff and slope steepness. Critical slope length, the location where contouring fails, is computed as a function of the shear stress applied to the soil. Both RUSLE1.06c and RUSLE2 use runoff in process-based equations to compute deposition caused by concave slopes, barriers, and low-grade channels using equations 2 and 3.

Deposition depends on the characteristics of the sediment reaching the support practice. Less deposition occurs if the sediment is fine. For example, less deposition occurs in a terrace channel or a sediment basin if a dense grass strip immediately upslope of the channel or basin deposits sediment that enriches the sediment load in fines. RUSLE2 computes this effect of upslope deposition, but RUSLE1.06c does not.

Computer Programs

The RUSLE2 computer program includes an exceptional graphical user interface. The user can customize the interface to their preferences by choosing screen arrangement, units, significant digits, and the complexity of the inputs and outputs. The program's computational engine maximizes the power of the RUSLE2 hybrid model structure.

The RUSLE1.06c program maintains the simple USLE index structure. However, accommodating interactions among the factors is inconvenient in this structure. The same information must be entered at multiple places in the RUSLE1 program. RUSLE2 represents detailed interactions with simple data entry. RUSLE1 is limited in the complexity of field situations that it can represent. RUSLE2 can analyze very complex hillslope shapes and spatial arrangements of soil, cover-management, and support practices on the slope.

Neither RUSLE1.06c nor RUSLE2 is a simulation model. The user describes the field condition using RUSLE program features. The models use this description to compute erosion. Both models must be told almost everything, including when frost kills vegetation. This approach, while seemingly crude and awkward, improves accuracy, power, and flexibility.

Conclusion

RUSLE2 is modern powerful, easy-to-use erosion prediction technology. USLE and RUSLE1 users, and perhaps users of other models, should shift to RUSLE2 for estimating rill and interrill erosion rates needed for conservation planning on all land uses. Readily available databases facilitate the adoption of RUSLE2. RUSLE1.06c is recommended for those users who wish to continue to use the USLE structure. The equations in RUSLE1.06c are much better than those in the USLE and previous RUSLE1 versions.

Acknowledgments

RUSLE2 was developed jointly by the USDA-Agricultural Research Service (ARS), the USDA-Natural Resources Conservation Service (NRCS), and the University of Tennessee. The Illinois State Water Survey, NRCS, ARS, and University of Tennessee analyzed the weather data to obtain new erosivity values. Additional information on RUSLE1.06c and RUSLE2 can be obtained from USDA-ARS-NSL (2003).

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Sediment Yield from Semiarid Watersheds

M. H. Nichols, K. G. Renard

Abstract

Stock tanks on the United States Department of Agriculture – Agricultural Research Service Walnut Gulch Experimental Watershed were instrumented in the mid-1960s with the goal of quantifying sediment yield from small rangelands watersheds. Periodic topographic surveys of stock tanks at the outlet of four watersheds ranging in size from 35 to 92 ha are used to compute sediment yield. Unit sediment yield from the four watersheds studied ranged from 0.4 to 2.8 m^3/ha/yr. Computed sediment accumulation is used in conjunction with observed precipitation and runoff data to relate the variability in sediment yield to the variability in climate. These data will be useful to land managers, decision makers, and scientists concerned with semiarid rangeland sediment yield.

Keywords: sediment yield, semiarid, rangeland, watershed

Introduction

Soil movement is of considerable interest to rangeland managers. Healthy ecosystems in properly functioning watersheds depend on maintaining soil onsite. Vegetation loss is often accompanied by erosion and transport of eroded sediment. In addition to productivity loss on uplands, eroded soil can have significant impacts on downstream water quality, and sediment deposition can reduce reservoir storage capacity.

Soil loss and movement in watershed uplands is difficult to measure, and may go unnoticed until it is a severe problem. Deposition is often easier to identify and measure. Water supply reservoirs, or stock water tanks, are found throughout the rangelands in the southwestern United States. These reservoirs collect sediment as well as runoff water, and can be monitored to assess sediment yield. Within the Walnut Gulch Experimental Watershed (Renard and Stone 1982, Renard et al. 1993), stock tanks behind earthen dams provide sites for sediment accumulation measurement. The objectives of this paper are to briefly describe the sediment measurement methods and to present a summary of sediment yield from four stock tank watersheds.

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Study Site and Methods

The Walnut Gulch Experimental Watershed (WGEW) is located in the transition zone between the Sonoran and Chihuahuan Deserts in southeastern Arizona. Twenty-two stock tanks on the watershed collect surface runoff that is used to water livestock. Twelve of the stock tanks have been instrumented to evaluate the interactions and effects of various soil and vegetation complexes on local runoff, water yield, and sediment production (Figure 1). Sharp-crested weirs are located in the spillways of four of the stock tanks. These four sites provided data for this paper.

Figure 4. Stage gage for measuring water level at stock Tank 223 on the Walnut Gulch Experimental Watershed.

The watersheds above the four stock tanks range in size from 35 ha to 92 ha and are underlain by a coarse-grained Quaternary and Tertiary alluvium shed from the Dragoon Mountains (Gilluly 1956). Vegetation, soil, and geology of each watershed were summarized from GIS layers developed at the Southwest Watershed Research Center (Table 1). Historically, the primary land use on the WGEW has been cattle grazing. The complex interactions between vegetative cover, underlying geology, and land use result in variation in sediment yield among small watersheds (Lane et al. 1997).

Table 1. Characteristics of selected stock tank watersheds.

Stock Tank Number	2002 Tank Volume (m^3)	Dominant Soil Type	Dominant Vegetation Type
208	7700	McAllister-Stronghold complex	Black Grama, Curly Mesquite
215	7200	Tombstone very gravelly fine sandy loam	Whitethorn Acacia, Creosote Bush, Tarbush
216	6600	Stronghold-Bernardino complex	Black Grama, Curly Mesquite
223	2900	LuckyHills-McNeal Complex	Whitethorn Acacia, Creosote Bush, Tarbush

Sediment accumulation and measurement

Sediment yield is the amount of eroded material that moves from a source to a downstream control point, such as a reservoir, per unit time (Chow 1964). The fate of eroded material within a watershed is influenced by hydrologic, topographic, vegetative and groundcover characteristics. Eroded particles may be transported to the watershed outlet, or they may be deposited and stored within the watershed. Stock tanks trap sediment at an outlet point, where topographic measurements of the dry stock tank surface can be taken to quantify sediment yield.

Periodic topographic surveys of the stock tanks on the WGEW are conducted to quantify the amount of sediment transported off the watershed above the stock tank and to compute reductions in tank storage capacity. As part of a nationwide sedimentation survey, methods for measuring the volume of sediment in small reservoirs were established in 1935 by USDA Soil Conservation Service (SCS) personnel (Eakin 1939, Brakensiek et al. 1979). Although surveying equipment has evolved, the general procedures remain unchanged and are currently in use by the Natural Resources Conservation Service (NRCS) and other federal agencies (SCS 1983).

Topographic surveys of dry tank surfaces consist of measuring the location and elevation of a sufficient number of points within the tank to map the surface shape. Tank surfaces are surveyed up to spillway elevation, or up to a level inclusive of the highest water level achieved during the period between surveys.

During the 1950s and early 1960s, a plane table was used to conduct surveys at Walnut Gulch. A level and stadia rod replaced the plane table in the 1970s and since 1993, a Sokkia Set 3CII Total Station has been used to characterize tank surface topography. Data are stored electronically and Surfer (Golden Software 1994) is used to generate stage-volume curves and contour plots and to compute volumes.

For each tank, the volume at sequential elevations is computed and plotted against the elevation to produce a stage-volume curve (Figure 2). The total tank capacity is the volume computed at the level of the spillway. Throughout the summer thunderstorm season, runoff transports sediment into the tank. As the tank fills with sediment the stage-volume relationship changes and a new survey is required to update the plot. Changes in volume between successive surveys can be attributed to the influx of sediment during the runoff season. Tank capacity is maintained by periodic sediment removal. Surveys before and after cleanouts are used to account for the material removed.

Following plane table and stadia surveys, collected data were plotted by hand, and a planimeter was used to compute the area enclosed by a contour. Tank volumes were calculated by computing volumes between successive contours and summing over the range of elevations. Recently, each of the hand-plotted maps was digitized, and elevations were adjusted using vertical control benchmarks to establish a common coordinate systems and datums for each of the four tanks. Surfer was used to re-generate contour plots and to re-compute volumes. Thus for each tank, a common datum, electronic data format, and computational method were used to quantify sediment accumulation. The calculated amount of sediment accumulated is reported as a volume in units of cubic meters. Units of m^3/ha provide information on the sediment yield relative to the watershed area.

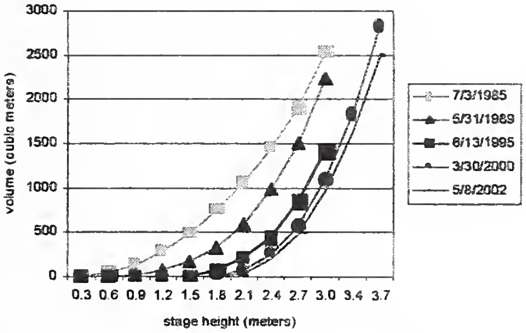

Figure 5. Tank 63.223 stage-volume curves generated for five different years.

Precipitation and runoff

Runoff-generating precipitation in southeastern Arizona is generally the result of high intensity, short duration airmass thunderstorms during the months of July, August, and September (Osborn 1983). Approximately 2/3 of the total annual precipitation on the WGEW occurs during the summer "monsoon" season (Nichols et al. 2002). Precipitation is recorded at 100 raingages distributed across the entire 150 km^2 watershed (Figure 3). Specific raingages associated with runoff within each tank watershed were determined based on Theissen weighted area coverages.

Each stock tank is instrumented to monitor water level. A vertical culvert pipe with slots at the bottom for water access acts as a stilling well. An instrument box on top of the stilling well pipe contains a water level recorder, which is connected to a pulley and a float that rests on the water surface. Analog recorders (Brakensiek et al. 1979) on the tanks were converted to electronic potentiometer systems in 1999.

Recorded water levels are used to calculate runoff. The relationship between water level and tank volume changes as sediment is deposited (Figure 2). Outflow over the spillway is monitored with sharp crested weirs. Spill volumes are computed using standard weir formulae (Brakensiek et al. 1979). In the absence of a spill, water depth is converted to volume based on the stage-volume relationship computed from topographic survey data.

Results

Overall average annual sediment yield from the four watersheds ranged from 36 to 142 m^3/year (Table 2). The total sediment yield for the period of record at each tank ranged from 1060 m^3 to 5670 m^3, although the period of record ranges from 29.0 years to 45.6 years.

Annual sediment yield values provide a general basis of comparison between watersheds. Sediment yield is influenced by hydrologic, geomorphic, and watershed characteristics. Sediment yield is directly related to runoff (Figure 4). During drought years when no runoff-producing precipitation events occur, sediment yield is zero. In contrast, high velocity flows associated with high intensity precipitation events can transport and deposit large sediment amounts. At tank 223, sediment yield ranged from a low of 1.2 m^3/ha/yr during the 1965 - 1975 time period and a high of 6.0 m^3/ha/year during the 2000 – 2001 period (Table 3). **However, caution must be exercised in comparing rates computed over differing time interval lengths. As the length of time between surveys increases, computed annual sediment yield rates can mask the variability.**

The annual variability in sediment yield is a reflection of the variability in precipitation and runoff. Theissen weights were assigned to raingages to determine the spatial contribution of measured precipitation over each watershed. Precipitation recorded at gage 23 contributes to runoff in stock tank 23. Figure 5 is a graph of annual rainfall at gage 23 for the time period 1953 – 1996 and illustrates the typical variability in precipitation on the WGEW. In general, the unit rate of sediment yield decreases as drainage area increases. Branson et al. (1981) presented a graph illustrating the relationship between sediment yield and drainage area based on the work of several researchers. The decrease in sediment yield can be explained in part by increases in deposition and sediment storage within the channel network with increasing watershed size. In addition, precipitation in semiarid areas like the WGEW is usually not spatially uniform over the basin. As watershed area increases, relative spatial coverage of precipitation decreases. Sediment yields from the four watersheds presented in this paper are plotted on the same graph (Figure 6). The relationship is consistent with the previously reported studies.

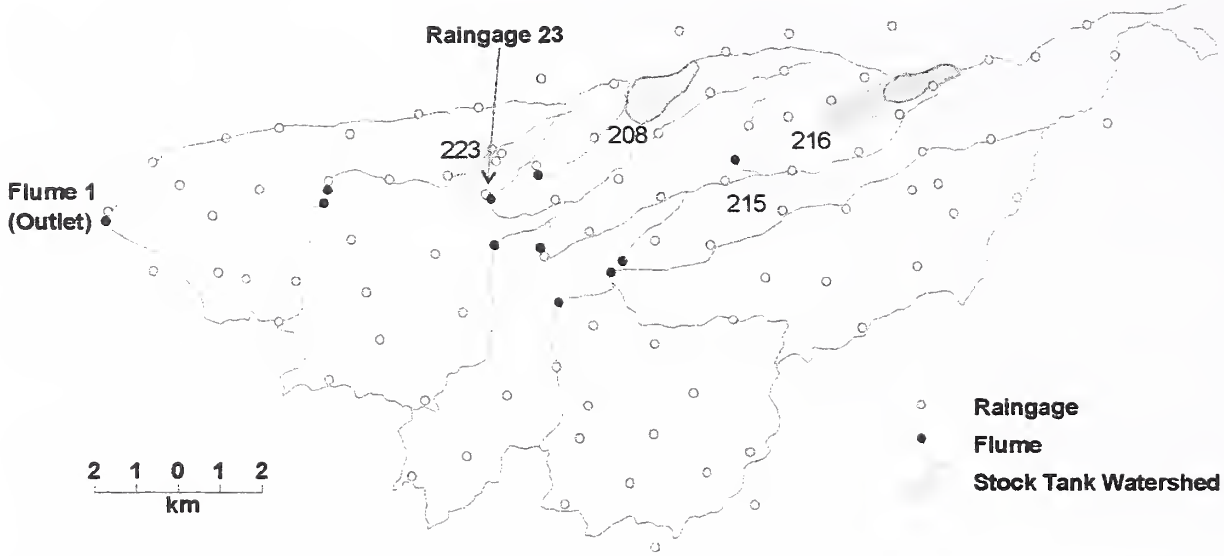

Figure 3. Walnut Gulch Experimental Watershed stock tank location map.

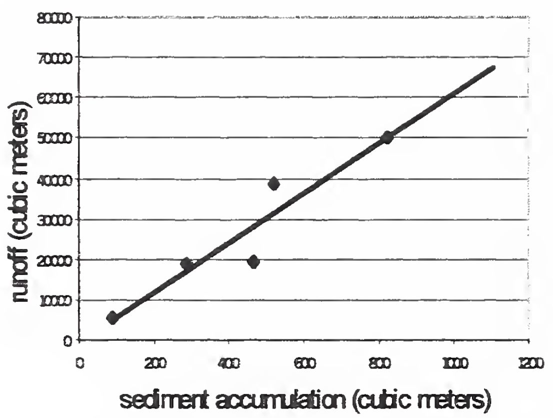

Figure 4. Relationship between runoff and sediment accumulation in Tank 63.223. R2 = 0.90.

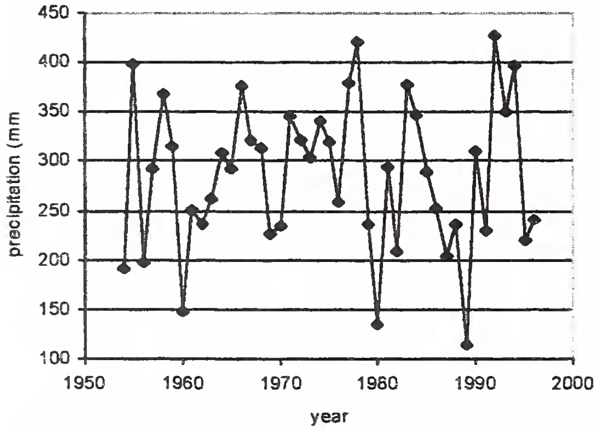

Figure 5. Annual precipitation at rainage 23.

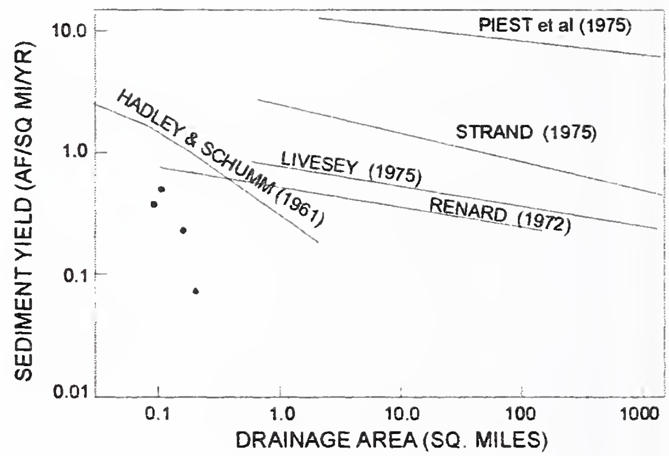

Figure 6. The relationship between sediment yield and watershed area including 4 WGEW stock tank watersheds (Branson et al. 1981, Figure 6-24).

Conclusions

Sediment yield from semiarid watershed is highly variable because precipitation and runoff are highly variable. One of the objectives of long-term sediment accumulation monitoring is to evaluate trends in sediment yield in relation to land management. However, conditions of stable sediment yield from which to compare are atypical. The variability suggests that average annual sediment yield rates may not provide sufficient information to interpret causes and effects of upland land management.

Table 2. Summary of sediment accumulation.

Stock Tank Number	Drainage area (ha)	Period of Record	Year of Record	Volume of Accumulated Sediment (m^3)	Sediment Yield (m^3/yr)	Unit Sediment Yield (m^3/ha/yr)
208	92.2	1973 - 1984	29.0	1057	36	0.4
215	35.2	1966 - 1984	35.9	2936	82	2.3
216	84.2	1962 - 1996	39.9	5667	142	1.7
223	43.8	1956 – 2002	45.6	5658	124	2.8

Table 3 Summary of Sediment yield in Stock Tank 223.

Survey Date 1	Survey Date 2	Fractional Years	Sediment Yield (m^3)	Annual Sediment Yield (m^3/ha/yr)
10/11/1956	6/27/1963	6.712	1672	5.7
6/27/1963	6/7/1965	1.948	464	5.4
6/7/1965	6/4/1975	9.997	518	1.2
6/4/1975	4/30/1985	9.912	1106	2.5
4/30/1985	7/3/1985	CLEANOUT		
7/3/1985	5/31/1989	3.912	286	1.7
5/31/1989	6/13/1995	6.038	823	3.1
6/13/1995	11/11/1996	1.416	89	1.4
11/11/1996	3/30/2000	3.384	267	1.8
3/30/2000	5/8/2001	1.107	290	6.0
5/8/2001	5/8/2002	1.000	144	3.3

Continued monitoring of sediment yield is necessary to obtain long-term records sufficient to incorporate variability when assessing trends. Sediment yield data play a key role in simulation model calibration and validation. Additional work to further quantify the spatial variability of sediment yields will indicate the watersheds where sediment production is the highest, and can be used to identify those areas where remediation efforts will have the greatest impact on reducing erosion.

Acknowledgments

The authors would like to thank the staff of the SWRC who contributed to the development of the Walnut Gulch database. The contributions of William Flack to field surveying, and updating datums and data reduction procedures are gratefully acknowledged. The reviews of Drs. Mark Nearing and Evan Canfield are greatly appreciated.

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Tracking Erosion and Sediment Re-distribution in a Small Watershed

Mark A. Nearing, Viktor Polyakov, Martin Shipitalo

Abstract

Current erosion and sediment monitoring systems are limited to sediment leaving a watershed outlet or to spatial distribution of loss using single tracer techniques. A new method is introduced to track erosion, translocation, and re-deposition of sediment in a small watershed, thus allowing a complete, spatially distributed, sediment balance to be made as a function of morphological landscape elements. A 0.68 ha watershed in Coshocton, OH with a silt loam soil and an average gradient of 8% was divided into six morphological units: lower channel, upper channel, lower hillslope, upper hillslope, interfluve, and toeslope. The six units were tagged with six rare earth element (REE) oxides in a powder form. Sediment leaving the watershed was collected, and soil translocation was evaluated using spatially distributed sampling of the soil surface. While the average soil loss on the watershed between May 2nd and November 8th 2001 was 6.1 t ha-1, local rates varied between 46 t ha-1 of loss to 50 t ha-1 of gain through deposition. The advantage of the multiple tracer technique was the ability to distinguish between multiple sources of sediment both exiting the watershed and re-deposited within the watershed. For the first time it was possible to itemize the sediment budget at any specific landscape element as three components: 1) the soil from the element that left the watershed with runoff, 2) soil from the element that was re-deposited on lower positions, and 3) soil originating from the upper positions and deposited on the element.

Keywords: sediment tracers, spatial distribution, sediment re-distribution, Coshocton OH

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Experimental Drainage Basins in Israel: Rainfall, Runoff, Suspended Sediment and Bedload Monitoring

Jonathan Laronne, Judith Lekach, Hai Cohen, Yulia Alexandrov

Abstract

Within the hyper-arid to semiarid areas of Israel three experimental drainage basins varying in area, climate, monitoring duration and type have been being research-monitored.

Lessons derived from a large number of published and ongoing research projects on these experimental basins focus on runoff and sediment in drylands. The effect of the spatial distribution of rainfall on runoff generation becomes increasingly important with aridity. Rainfall angle on hillslopes and storm intensity and direction derived from rainfall recorders and radar backscatter are crucial for explanation of runoff response. Runoff hydrographs have more bores, shorter-duration peaks, briefer recessions, longer dry periods, and are more variable in flood volume and flood peaks with increased aridity. Suspended-sediment fluxes, yields and concentrations are high in the semiarid realm, reaching maxima at the beginning of a flood season and after long, dry spells. Bedload fluxes are exceptionally high from dryland basins where hillslopes are minimally vegetated and where bedload transport takes place in channels lacking an armor layer. The bedload/ suspended-sediment load ratio increases with aridity and may rise to 70%. The depth of channel bed activity is indicated by a fluvio-pedogenic unit. Hillslope to channel connectivity is high in drylands. In the hyperarid region the finer suspended-sediment is derived from hillslopes while the sandy fraction derives from the channel bed.

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Keywords: bedload, suspended sediment, runoff, arid

Introduction

The research watersheds are those of Nahal (stream in Hebrew) Yael, subdivided into five sub-basins, Rahaf-Qanna'im (main and tributary, respectively) and Eshtemoa. The basins have precipitation, runoff, sediment and fluviomorphological records. Each was conceived for differing purposes, but all share the common two objectives for continuous monitoring of water, sediment and morphology:

1) Many hydrological issues may be approached if, and only if, there are prototype databases on a wide spectrum of hydrological processes; and
2) There is a need for long-term records to assess large floods and subsequent hydrologic and geomorphic recovery.

This paper introduces these research watersheds as well some of the multitude of results that have arisen from prolonged monitoring. Whereas results from the Yael and Eshtemoa have been published in various outlets, those from Rahaf-Qanna'im have not (Cohen's dissertation).

The Research Watersheds

The Nahal Yael Watershed has been in operation for more than 35 years, is the smallest of the three and is located in the most arid region. The Rahaf and Eshtemoa are similarly sized, but vary in climate, soil cover and tectonic setting (Table 1). The Yael's instrumentation was novel when introduced in the 60's and onwards; that in the Eshtemoa and Rahaf is modern, with varied electronic sensors and digital recorders.

Table 1. Characteristics of the Israeli Research Watersheds. SD denotes standard deviation.

Watersheds	Yael	Rahaf (Qanna'im)	Eshtemoa (Yatir)
Drainage basin area (km^2)	0.6	79	112
Annual rainfall (mm)	27	50-130	280
Climate	hyper-arid	Arid	semi-arid
Soils	desert reg, rock outcrops, colluvium	Desert lithosols, reg, coarse alluvium	rendzina, loess in valleys
Rock	granite, schist, amphibolite	Dolomite, limestone, some chert	limestone with Nari, dolomite, chalk, chert
River pattern a outlet	shallow braided	Canyon, flat alluvial to braided	single thread, straight to meandering
Width of channel at station (m)	7	Variable, 30 at station	6
Mean bed slope	0.05	0.05 (0.017 Rahaf outlet and 0.027 at Qanna'im)	0.0075
Banks	alluvial, bedrock	Bedrock (limestone, marl, conglomerate), alluvium	loess, alluvium
# of events each year	0.6 arriving at outlet	2	3.1
SD of above	0.5	2	4.8
Max recorded discharge ($m^3 s^{-1}$)	3.7	775 (1987 estimate)	84
Mean annual runoff volume ($10^6 m^3$)	?	0.25 (min. estimate)	0.65
SD of above ($10^6 m^3$)		0.36	0.90
Typical suspended sediment concentration (mg l^{-1})	40,000 (only during stage rise)	1,000-55,000	34,000
Typical bedload discharge (kg $m^{-1} s^{-1}$)		0.1-10	0.3-2
Mean annual sediment yield (t $km^{-1} yr^{-1}$)	170	150 (estimate)	472

Results from the multi-year monitoring of water and sediment at these research watersheds is herein presented. Rather than being all inclusive, these are illustrated with examples.

Monitoring Methods

Rainfall was monitored using miniature accumulating gages, tipping bucket recorders as well as radar backscatter images acquired at intervals as short as 5 minutes. Runoff was measured in flumes (Yael), or else by continuous water depth monitoring using pressure transducers and surface velocity using floats up to bankfull discharge. Longitudinal water surface slope was monitored using a set of pressure transducers along the banks (Eshtemoa and Rahaf). Suspended sediment concentration was sampled manually, or automatically by a rising stage sampler (Yael) (Lekach and Schick 1982) or by a pre-programmable pump sampler, as well as by hand sampling (Alexandrov et al. 2003a). Average suspended sediment concentration was also calculated based on reservoir sedimentation, dividing the mass of fine sediment deposited during an event by event runoff volume . Continuous suspended sediment concentrations were obtained from the deployment of turbidity sensors. The latter have a good response up to 80,000 mg l^{-1}; some may be calibrated in the lab and in the field up to 150,000 mg l^{-1}. Bedload was monitored continuously and automatically using a set of Birkbeck type slot samplers (5, 2 and 1 respectively at Eshtemoa, Rahaf and Qanna'im). These typically have a sampling efficiency of 100% until they are almost entirely full (Laronne et al. 2003). These samplers are excellent at separating bedload from suspended sediment for any size of sediment finer than the slot width.

Rainfall

Rainfall is very variable in hyperarid areas, even more so than in the semiarid realm. This is more pronounced during rainfall events (Figure 2), a derivative from the local convective origin of most rainstorms, giving rise to rainfall spottiness (Sharon 1972). Measurements of rainfall using a dense array of gages in the Yael has shown it to vary with aspect and wind velocity on hillslopes. Indeed, the difference between meteorological and hydrological rainfall is often considerable and significant with relevance to runoff production (Sharon 1980).

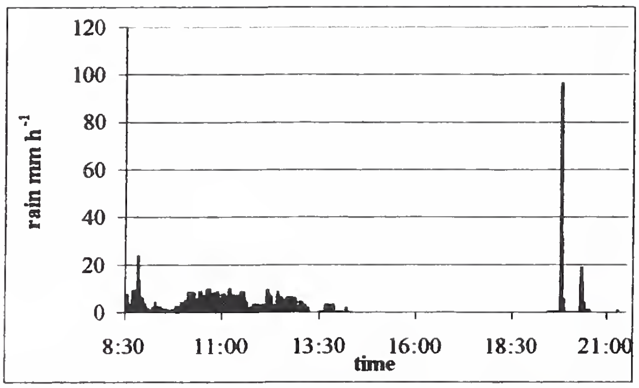

Figure 1. Short term rainfall intensity at Nahal Yael for locally-termed Event 12. It created two rises; the first was generated by medium rainfall intensities, the second by high intensity (96 mm hr^{-1}) lasting 4 min (see Figure 2).

Runoff

Similar to rainfall, runoff is very variable in time and space. Dryland channels are ephemeral, typically flowing merely 0.5-5 times annually. When they do flow, the rise is steep (Figure 2), often arriving as a bore (Reid et al. 1994). Longitudinal water surface slope varies considerably during a flow event, but the variation is not large enough to generate large differences in bedload discharge at rising and falling stages (Meerovich et al. 1998).

Suspended sediment

Unlike the relative ease of obtaining rainfall and runoff data in dryland watersheds, sediment monitoring is more difficult to obtain, because in the past it necessitated the presence of personnel at the site. Because rainfall and, therefore, runoff are spotty in nature, this has proven to be cumbersome, leading to no good databases (Luna Leopold, personal communication).

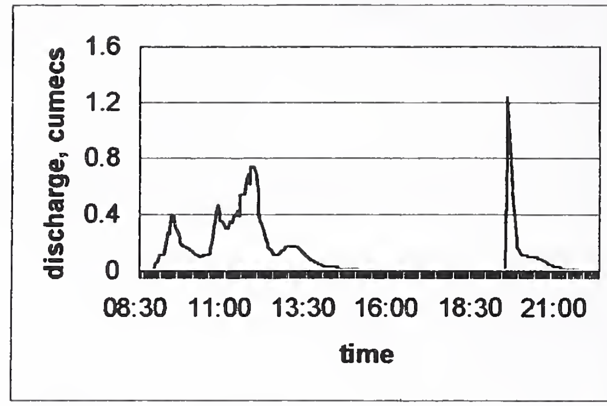

Figure 2. Discharge hydrograph of event 12 at Nahal Yael (see Figure 1). Note the bore on the second rise.

The sources of suspended sediment may be the hillslopes or else the coarser, sandy fraction derived from the channel bed (Figure 3). By tracking storms using radar backscatter, we have shown that the source of suspended sediment may be attributed to tributary inputs in the form of piggy-back rises, or else due to temporal increases in rainfall intensity.

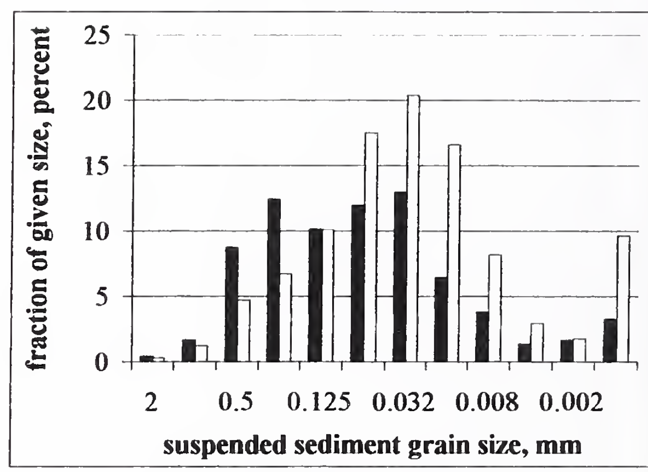

Figure 3.Grain size distribution of suspended sediment at an upper site (white) and at a lower site (black) in Nahal Yael. The source of sediment in the upper site are mainly the hillslopes, further downstream the alluvial channel contributes the coarser component.

During a flow event, the concentration of suspended sediment varies hysteretically with water discharge (Figure 4). There are group-types of response, and overall these relationships are complex. In general, concentrations of sediment are high (Table 1; see also

Figure 4), at times exceptionally high, more than 200,000 mg/l. The high concentrations typify rises, especially after a long dry spell, during which much sediment has been weathered and is readily available.

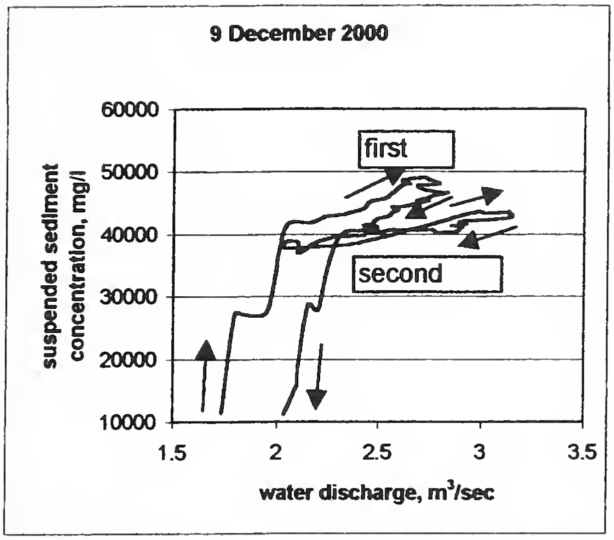

Figure 4. Variation of suspended sediment concentration with water discharge on December 9, 2000 at Nahal Eshtemoa. Concentrations were calculated based on calibration of turbidity. Note the high concentrations and the similar clockwise responses in both rises.

Although the suspended sediment-water discharge relationships are complex within events, the relationships are simple and well defined when averaged from entire events. This holds for concentration vs. flow volume and flow peak (Alexandrov et al. 2003b).

Bedload

Bedload was studied at the Yael using reservoir deposits, tagged clasts and scour chains. The sediment budgets at this hyperarid watershed vary temporally (Lekach and Schick 1993). It has been demonstrated that bedload moves in a scour layer that has a maximum depth, below which a stable soil horizon has developed (Lekach et al. 1998).

Bedload discharge in semiarid channels such as the Eshtemoa and the smaller Yatir are exceptionally high, orders of magnitude higher than in their more humid counterparts (Reid and Laronne 1995). This has been shown to occur due to the unlimited supply of coarse sediment that is unarmored (Laronne et al. 1994). Bedload discharge responds directly to shear stress, such that the response is sympathetic, remindful of the relationships derived from flume studies. Indeed, the conditions of no armor development in such channels is often modeled in flumes. Our results demonstrate that bedload discharge varies in accordance with several flume-based equations, such as the 1948 Meyer Peter equation (Reid et al. 1996). Figure 5 demonstrates the bedload response mentioned above.

Conclusions

In Israel, the drainage area of monitored watersheds is limited, hence 3-4 additional representative basins covering areas of 300, 1000, 2000 and 8000 km^2 will likely be implemented in the next decade.

National and regional hydrological research needs will dictate future global monitoring in experimental, research basins. International collaboration may bring about considerable cost reduction by exclusion of monitoring aspects that can be evaluated based on the monitoring in other, similar conditions. Advanced international collaboration on validation and calibration of and consistency in monitoring means, as well as syntheses of lessons derived from international collaboration, such as from an International Watershed Research Network, are required for maximizing our understanding of water and sediment basin responses in varied global regions.

Acknowledgments

Research results reported in this paper are based on grants from more than a dozen funds. Asher Schick brought forwards the notion of research basins in Israel via the Nahal Yael. Ian Reid has been principal co-investigator on the Nahal Eshtemoa.

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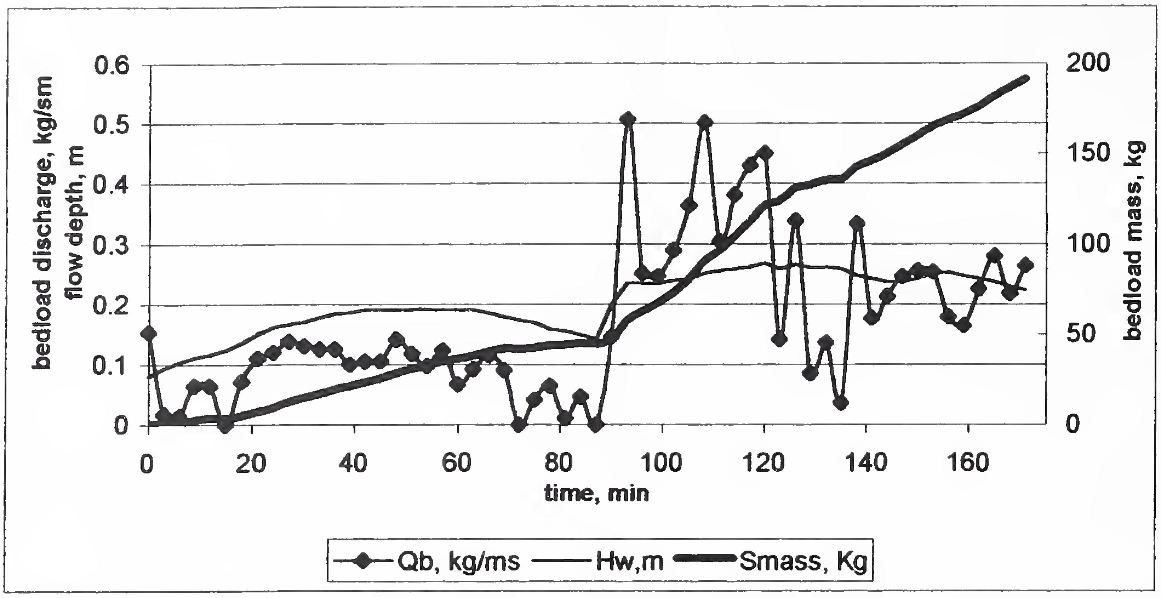

Figure 5. Bedload discharge (3 minute averaged) and cumulative bedload mass as monitored by the Nahal Eshtemoa left bank sampler on January 16, 1997. Observe that bedload discharge responds well to water depth (i.e. to channel average shear stress) and that it is very high even in this shallow flow. Bedload discharge averaged for shorter times is considerably higher.

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The Use of Used Tires in Water Systems

Stuart A. Hoenig

Abstract

Old tires have many uses in the control of erosion, both in and out of water. They are pollution-insect free, good insulators, fire resistant when filled with soil and last an estimate of 250 years.

Keywords: tire, erosion control, pollution free, long lasting

Introduction

The United States has a vast number of used tires and they increase at roughly 300 million per year. Some are burned in smelters or ground up for surface covering. This process is handicapped by the high strength steel that is part of the tires. Burning introduces toxic materials into the air that are difficult to capture.

It seems more effective to use them in their original condition. They can be used for animal fences on farms, as subgrades or barriers for road construction, bullet stops on rifle ranges, off-shore for wave control and fishing. Wave barriers around lakes and fish habitat have proved to be successful. Other applications are construction of houses and water-erosion control.

Many studies have shown that whole tires do not present smell or pollution problems, are very long lasting in dry soil or fresh and salt water. When filled with soil or baled (to be discussed below) they are very resistant to fire, insect or noise penetration.

Erosion Applications

In the West rain falls in short and often intense storms. The area of the storm may be small but the rapid increase of the water and speed of the flow causes severe erosion.

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The question is "what can be used for erosion control?" Rock barriers are effective but require significant time to build, if stones are not available they must be trucked in.

Concrete has been used but it is expensive and some skill in concrete construction is required. The cost and requirement for hiring skilled help makes concrete impractical for many applications.

In contrast tires are generally free from the County involved and they are easier to haul than stones. In some cases the County will actually haul them to the construction site. The author has had some experience in tires for erosion control Figures 1 and 2 show a tire dam built some 5 years ago. It has performed very well, water is retained in the soil and grass can grow. The water passing the dam is free of soil particles and the rancher involved is quite happy with it.

The construction was aided by Mr. Joseph Minyard a graduate student and Pima County probationers. The first step was the removal loose soil in the arroyo; we went down until hardpan was reached. The first layer of tires was laid and pinned to the soil with 18-inch soil bolts. The tires were then filled with 1" stones and plastic ties were used to attach the next layer. (Plastic ties were specified by a state agency and they have stood up well to the intense sunlight.) Several figures show the tie and fastening process and the dam in cross-section (Figures 3, 4, and 5); the final dam was 5 tires high.

Results

The dam has stopped soil from eroding in the area; a nearby road that was blocked by loose sand after every storm is now always passable. The soil level above the dam has risen and there is grass where there was nothing but bare sand.

Comments

The use of baled tires is now quite common; 100 passenger car tires are used. A photo of the baling process is shown in Figure 6. The bales are 4.5 x 5.5 x 2.5 feet; they weigh some 2,000 lbs. and must handled by front loader or small crane. The bales are tied together by steel wire put in during the baling process, it is interesting to note that after a year in baled condition the wires can be removed and the tires stay in place. Chain can be inserted to fasten the bales together producing very strong walls.

Small bales have been made for special purposes, they can be placed at the center of a gabion and then surrounded by stones. The system is much lighter than a gabion filled with stones.

The application of bales for banks and breakwaters at lakes is shown in Figure 7. The nice thing about bales is they are fast to use provided you have power machinery. If the bales are unsightly they can be covered by adobe.

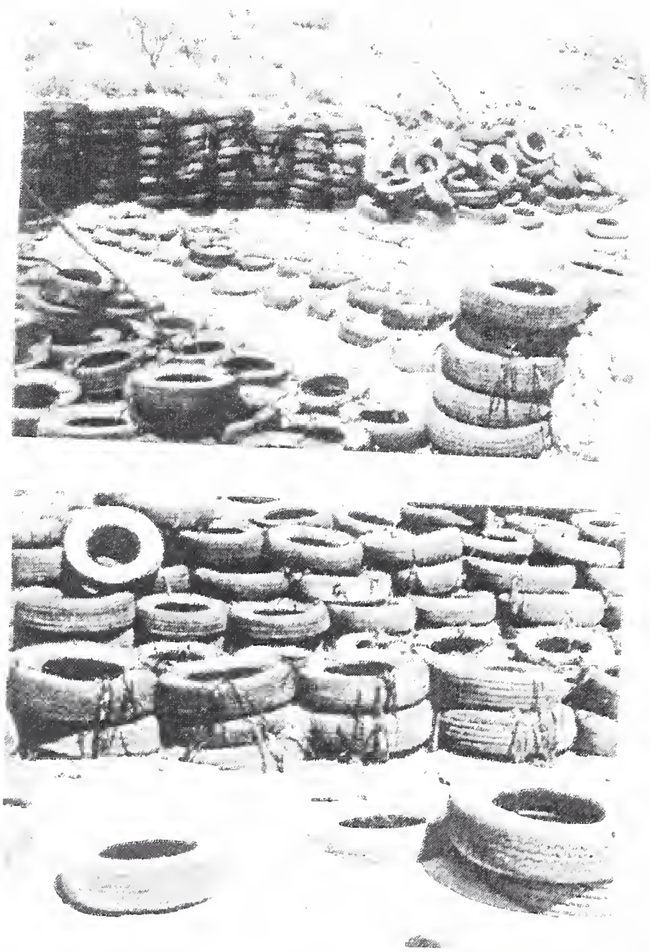

Figure 1. Erosion control dam under construction.

Figure 2. Dam under construction

Figure 3. Three-dimensional view of dam.

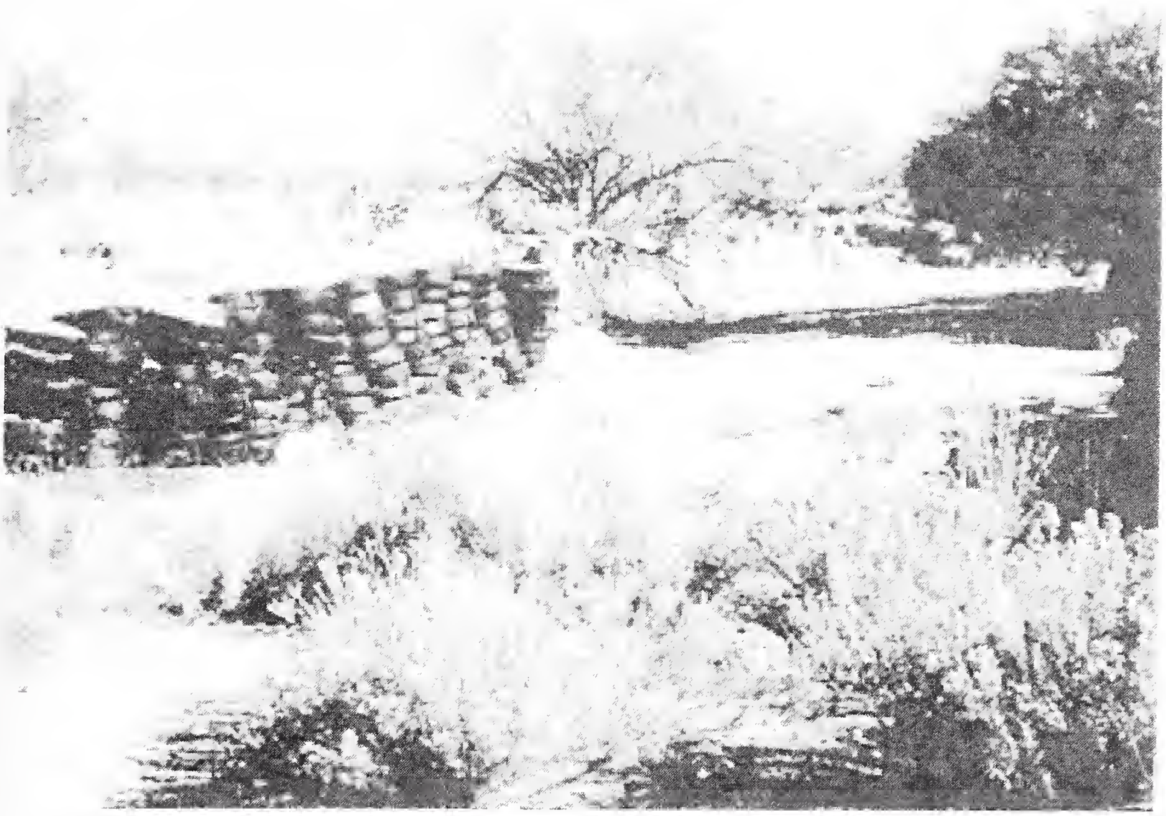

Figure 4. Dam six months after construction; the soil is now five tires high.

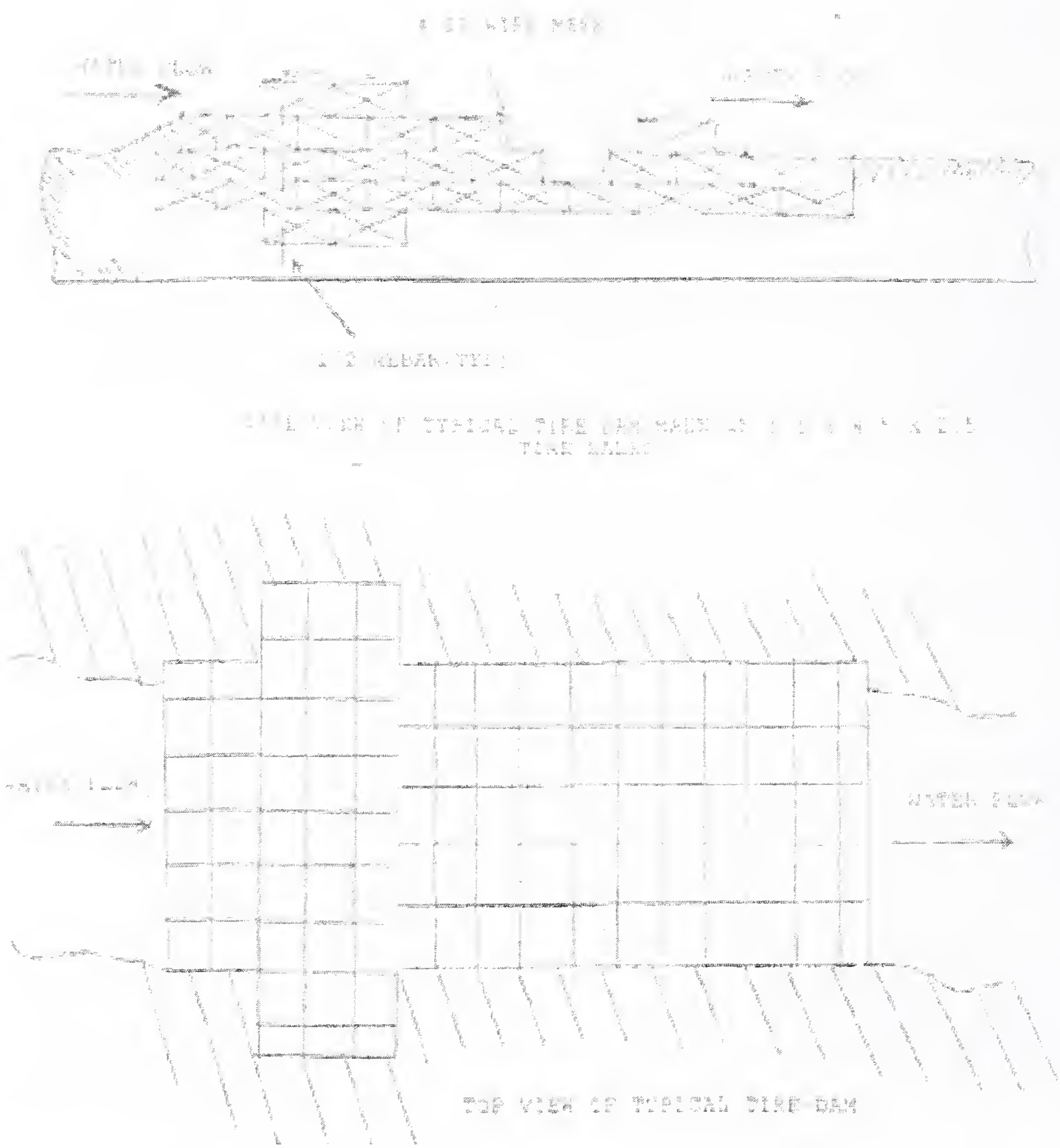

Figure 5. Dam (using bales) shown in side and top views.

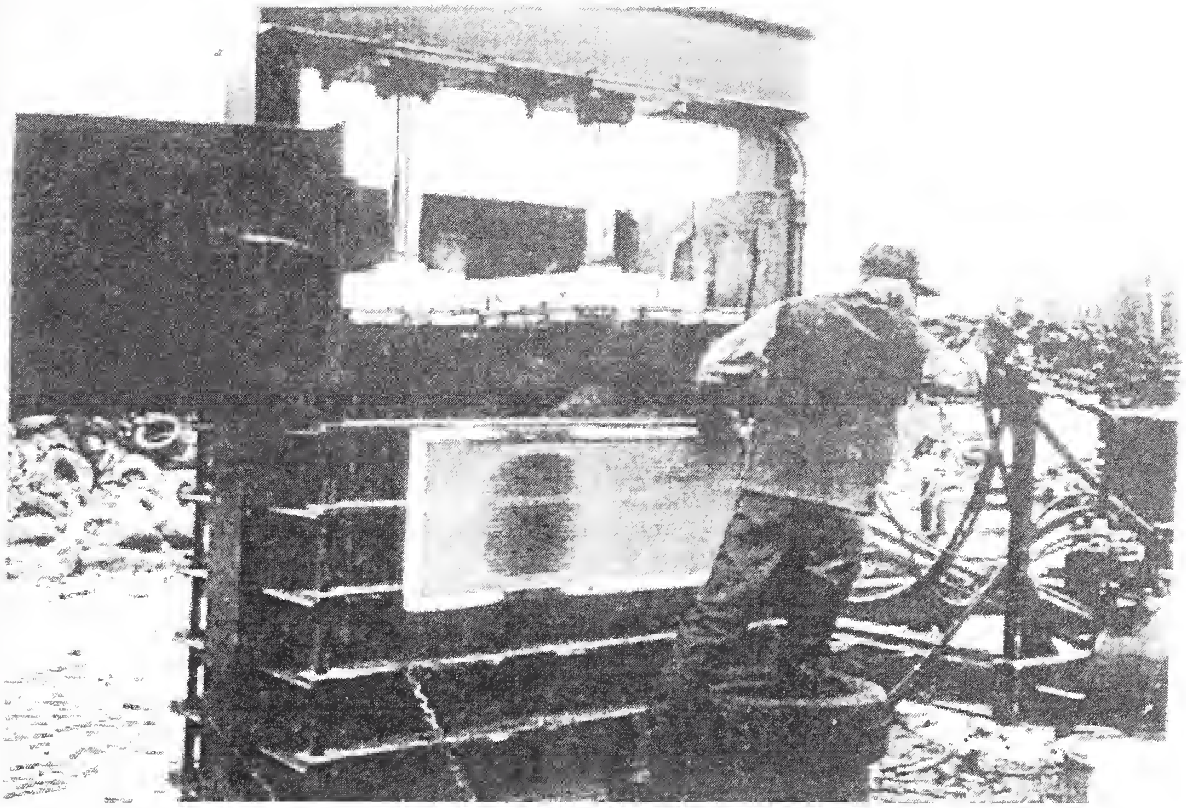
Figure 6. The tire baling process.

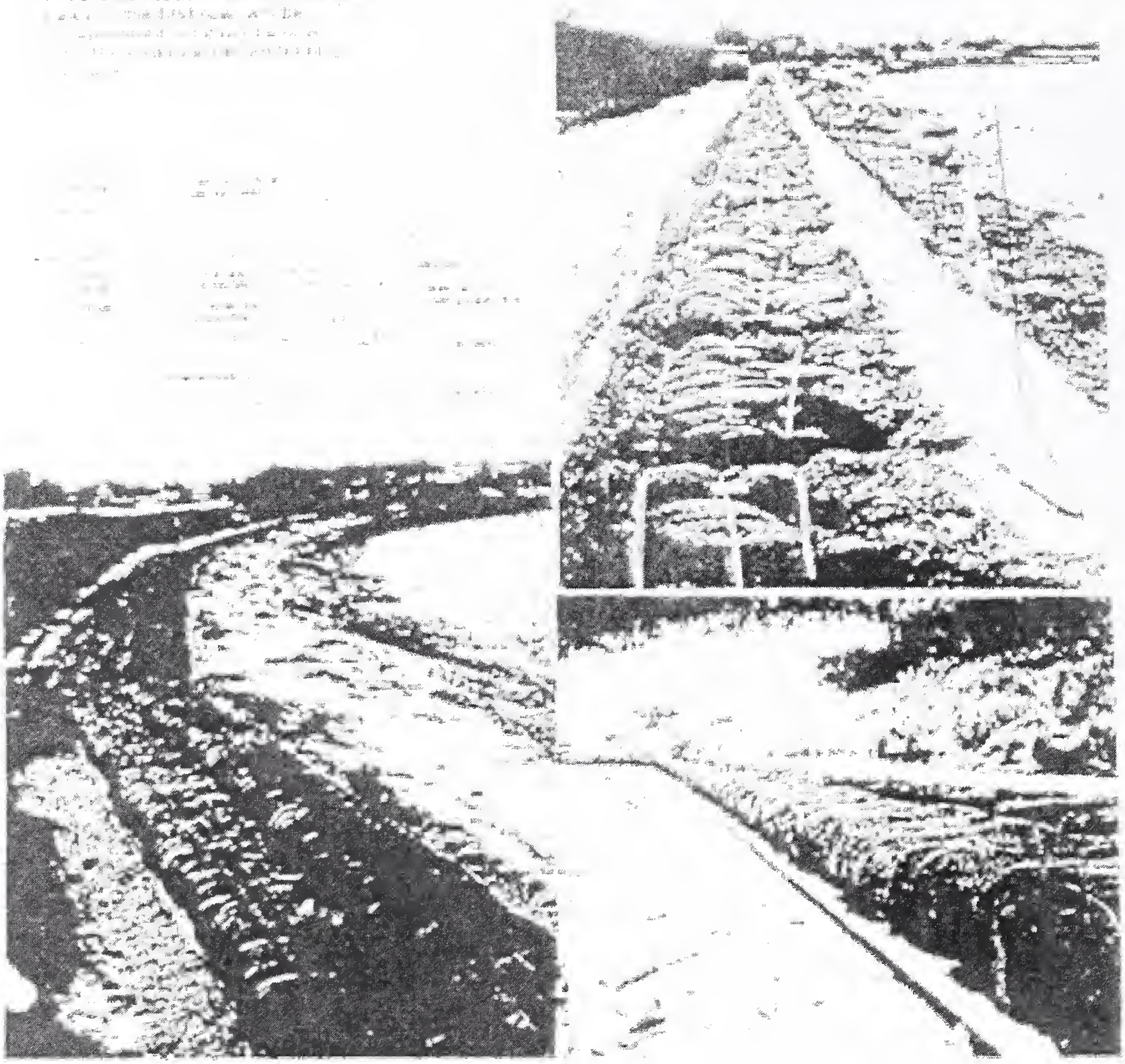

Figure 7. Application of bales along the shore of a lake

Effect of Peak Flow Increases on Sediment Transport Regimes Following Timber Harvest, Western Cascades, Oregon

Gordon Grant, Shannon Hayes, Sarah Lewis

Abstract

A persistent and often contentious debate surrounds evaluating effects of forest harvest activities on streamflow. Despite decades of paired-watershed studies at small experimental catchments world-wide, the jury is still out on the magnitude, persistence, and mechanisms responsible for peak flow changes following timber harvest. Recent studies examining long-term streamflow data from the H.J. Andrews Experimental Forest reached conflicting conclusions on the magnitude and causes of peak flow changes. But no studies have evaluated the geomorphic response to observed peak flow changes-- a question of great interest in interpreting potential downstream consequences of forest management on channels and ecosystems. Since the relation between sediment transport and discharge typically follows a power law, small increases in discharge can translate into large increases in sediment transport. But interpreting the geomorphic effects of peak flow increases is confounded by the fact that timber harvest typically influences both the hydrologic regime and sediment supply of a watershed, making it difficult to isolate the peak flow effect alone. Here we report on a novel approach to this problem using paired-watershed data to predict streamflow response in the absence of cutting. We combine this predicted hydrology with observed relations between discharge and sediment transport to disentangle the relative effects of changes in hydrology and sediment supply. Results indicate that while peak flow increases alone can account for modest increases in both suspended and bedload transport, the peak flow effect is dwarfed by the increased supply of sediment that accompanies timber harvest.

Keywords: sediment transport, peak flows, timber harvest, paired watershed

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Stream-bed Scour and Fill in Low-order Ephemeral Stream Channels

D. Mark Powell, R. Brazier, M. Nichols, J. Wainwright, A. Parsons

Abstract

Reach scale patterns of scour and fill have been studied in three, low-order dryland ephemeral stream channels with sandy bed materials. Depths of scour within the reaches are highly variable though scour is generally matched by equal amounts of fill. As a result, there is little net change in bed elevations over the flood season. A preliminary examination of the data using serial correlation and spectral analysis reveals no systematic pattern within the reaches although interpretation of the results is constrained by the limited length of the spatial series. For many of the events studied, the exponential model provides a satisfactory approximation for the distribution of scour and fill depths and may provide a basis for predicting sediment exchange depth distributions, at least to a first approximation, in low-order sand-bed streams.

Keywords: scour and fill, sand-bed channels, dryland rivers, flash floods

Introduction

Scour and fill refer to fluctuations in the vertical position of an alluvial stream-bed during a flood event. The fluctuations occur in response to the entrainment (or scour) and deposition (or fill) of bed material and reflect both the redistribution of sediment within the channel and the short-term hydraulic adjustments that help a river to maintain a quasi-equilibrium channel form (Andrews 1979). As a result, scour and fill processes have been of longstanding interest to geomorphologists, engineers and aquatic ecologists seeking to understand the morphodynamics of alluvial rivers (Haschenburger 1999) the stability of artificial structures such as bridge piers, pipelines and water abstraction points (Chang, 1988) and the role of flood disturbance in structuring lotic ecosystems (Lapointe et al. 2000, Rennie and Millar 2000).

Although scour and fill are characteristic of all alluvial rivers, they are of particular importance in dryland environments where there is commonly an almost unlimited supply of sandy-gravely material that is readily entrained by infrequent, but intense, flooding. However, there is considerable uncertainty about the dynamics of scour and fill in dryland fluvial systems. Some studies have suggested that scour tends to occur in reaches that are narrower and deeper, while wider and shallower reaches tend to aggrade (Lane and Borland 1954). This is thought to reflect the control of channel width on unit discharge and as a result, predictive relationships between unit discharge and scour depths have been identified (Leopold et al. 1966). Other studies, however, have shown that patterns of scour are independent of channel morphology (Emmett and Leopold 1965) and to be consistent with the migration of bed forms (Foley 1978).

Uncertainty over channel behaviour arises because few studies have featured measurements of sufficient density to characterise the magnitude and distribution of scour and fill occurring within a reach. Gauging station measurements are limited because they are restricted to observations made at isolated cross-sections whilst other studies are constrained by the low density of data acquisition (e.g. Leopold et al. 1966). In this paper, we present an analysis of event-based measurements of scour and fill made with sufficient

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spatial resolution to capture reach-scale variability and pattern. The purpose of the study is to characterise the variability of scour and fill at the scale of the channel reach. The measurements were made in three steep, low-order, sand-bedded ephemeral stream channels as part of a wider study investigating erosion by water in dryland catchments.

Study Area and Methods

The study was undertaken at Walnut Gulch, the Experimental Watershed of the United States Department of Agriculture, Agricultural Research Service in southeastern Arizona (USDA-ARS 2003). The catchment consists of grass and shrub covered piedmont sands and gravels. The climate is semi-arid and the channels flow ephemerally in response to intense, short-lived and highly localised convective storms during the summer months.

Measurement efforts were concentrated in three reaches within Lucky Hills, a 43.7 ha sub-catchment of the main watershed. One reach was located on the main channel and two reaches were located on tributary channels (upper and lower). All three reaches were straight, single thread and relatively narrow and steep with planar beds of medium-coarse sand with a small gravel fraction (Figure 1).

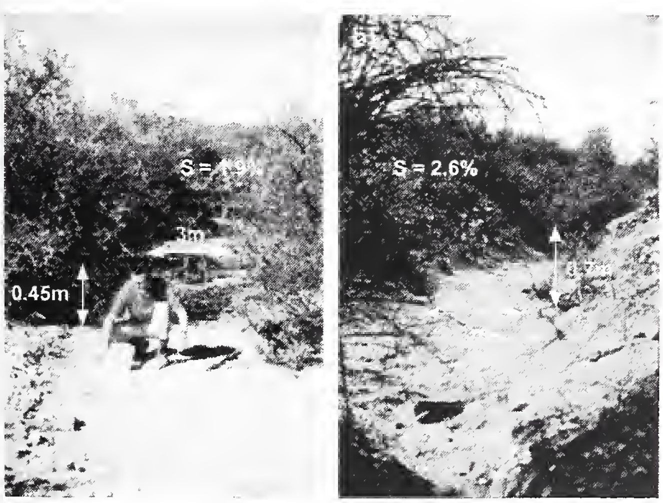

Figure 1. Downstream views of (a) the main channel and (b) the upper tributary study reaches. In (a), the author is holding two scour chains.

Scour and fill data were collected using lengths of metal-linked chain (Laronne et al. 1994). Each chain was inserted vertically in the streambed with a length of chain left exposed at the channel surface (Figure 2). After each flood, the elbow of the chain was located. The difference in the length of chain above the elbow before and after a flood yielded the depth of scour. When fill occurred, the distance between the elbow and the post flood bed gave the depth of fill. Once these measurements had been taken, the chain was reset in anticipation of the next flood.

Between three and five chains were installed at equally spaced distances across each cross-section in the relatively wide main channel and lower tributary reaches. In the narrower upper tributary reach, chains were installed in an alternating sequence of one in the channel centre and two at left and right locations to ensure that adjacent chains did not become entangled. Accordingly, the number of chains installed in each reach ranged from about 45 in the upper tributary reach to nearly 100 in the main channel and lower tributary reaches. Chain densities varied between 0.36-1.3 m^{-2}, values that exceed those in previous studies by two-three orders of magnitude (Rennie and Millar 2000).

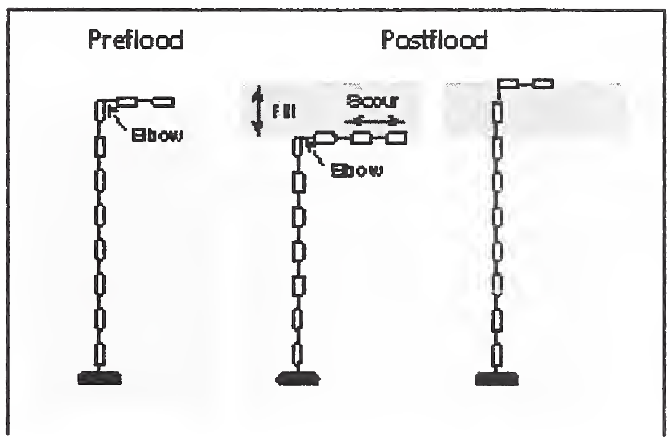

Figure 2. Schematic illustration of scour chain defining a net fill event.

Flow stage was measured using a combination of gantry mounted ultra sonic depth recorders and rudimentary crest stage recorders. Discharge was estimated from the stage records using Manning's flow resistance equation.

Spatial Patterns of Scour and Fill

Cross-sectional average depths of scour, fill and net elevation change recorded in the main channel during a bankfull event are shown in Figure 3a. Scour was observed over entire reach length, though the depths varied from over 15 cm to less than 5 cm. The reach average scour depth was 6.5 cm. Depths of fill range over similar values, though fill just exceeds scour. As a consequence, there was a reach-average net elevation gain during this flood of just under 2 cm. Other reaches show similar results. Figure 3b shows the pattern of scour and fill and net elevation change recorded in the lower tributary reach during the same event. Here, unit discharges were less than 1/10th that recorded in the main channel and as a result, bed activity was less intense. However, there was still significant variability in depths of scour and fill and as before, depths of scour and fill were approximately equal at each cross-section.

A preliminary analysis of the spatial pattern of stream-bed scour provides some evidence for oscillatory behaviour. Theoretical analyses of turbulence suggest that flow in a straight, uniform channel generates alternate zones of faster and slower flow with an average spacing of 5-7 times the channel width (Yalin 1992). The pseudo-cyclic distribution of scour depths may, therefore, reflect the interaction between alternate faster and slower areas of flow with the mobile bed sediment. We are currently investigating this hypothesis using serial correlation and spectral analysis

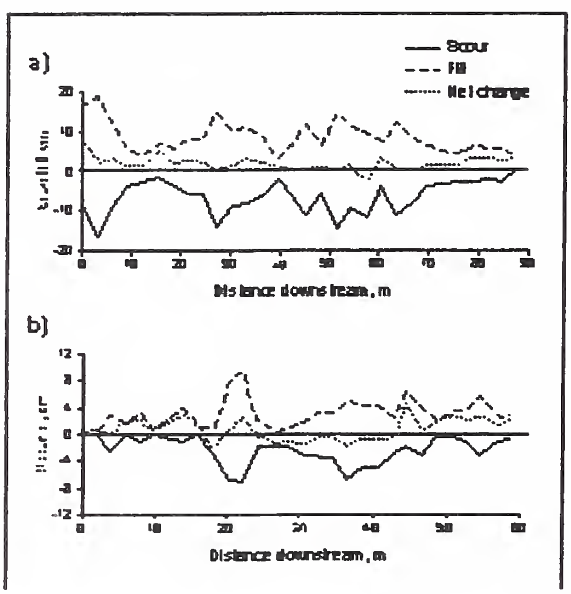

Figure 3. Spatial patterns of cross- sectional average scour, fill and net change in (a) the main channel ($Q_p = 4.5\ m^3\ s^{-1}$) and (b) the lower tributary reach ($Q_p = 0.2\ m^3\ s^{-1}$).

techniques to define the scale and regularity, or otherwise, of variations in scour depths. Early results, unfortunately, are equivocal, largely because of the limited length of the spatial series.

Reach Scale Variability in Scour and Fill

The reach scale variability in scour and fill can most readily be investigated using frequency distributions. Distributions of scour depths obtained from the main channel for five representative flow events (characterised by peak discharge Q_p) are shown in Figure 4. Scour depths are monotonically distributed. At low flow events ($Q_p = 0.2\ m^3\ s^{-1}$) bed activity extends to depths less than 8-10 cm. In contrast, maximum observations of scour equalled 30 cm during the flood with the largest peak discharge ($Q_p = 4.5\ m^3\ s^{-1}$). Overall, the channel bed experienced an increasing depth of activity over an increasing proportion of the bed as peak discharge increased. Consequently, mean depths of scour increased with peak magnitude.

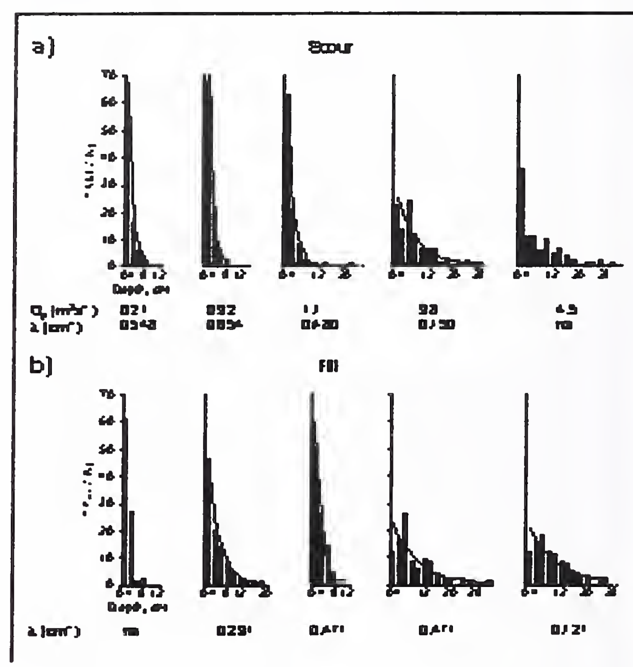

Figure 4. Frequency distributions of scour (a) and fill (b) for five runoff events in the main channel (n = 96). Statistically significant exponential distributions are shown by the fitted lines (ns = not significant).

The distributions shown in Figure 4 have been modelled by the exponential function. The exponential density function is defined as $F(x, \lambda) = \lambda e^{-\lambda x}$ in which F(x) is the proportion of the channel bed scouring or filling to a given depth increment x in centimetres and λ is the model parameter that is the inverse of the distribution mean.

Application of the Chi Square goodness-of-fit test indicates that the exponential function fits four of the five scour depth distributions at a significance level of 0.05 with parameter values of between 0.65 and 0.15 (Figure 4a). The exception is the largest flood where there is an over-abundance of observations in the first class and insufficient observations in the second class. Figure 4b shows the corresponding distributions of fill. Four of the five fill distributions conform to the exponential distribution at the 95% confidence level. The exception here is one of the smaller events.

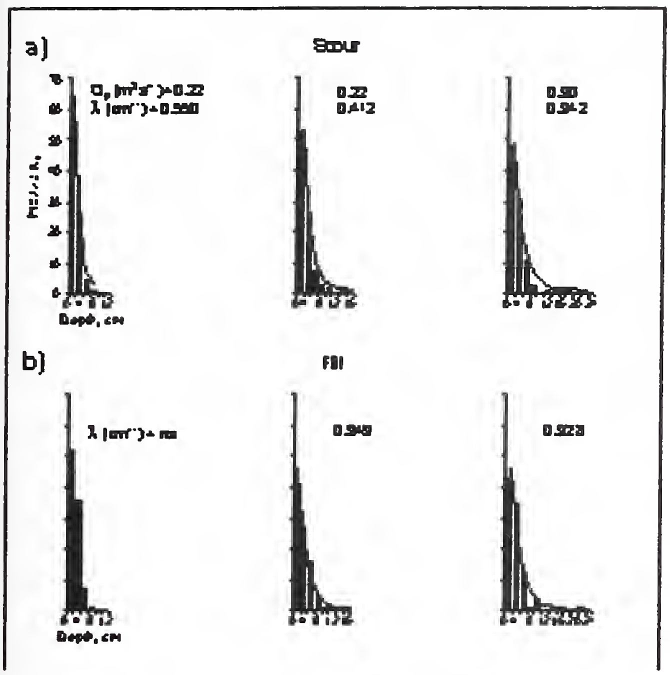

Figure 5. Frequency distributions of scour (a) and fill (b) for five runoff events in the lower tributary reach (n = 95). Statistically significant exponential distributions are shown by the fitted lines.

Other sites show similar distributions of scour and fill depths. Figure 5, for example, shows the distribution of scour and fill for three flow events recorded in the lower tributary reach. The flows are about the same magnitude as the lowest flows recorded in the main channel and bed activity extends to comparable depths - less than 8 cm, in general, with a limited number of channel locations experiencing up to 22 cm of activity at the slightly higher flow.

More significantly, it is seen that the shape of the scour and fill distributions are similar to those recorded in the main channel with five of the six distributions conforming to the exponential model ($\alpha < 0.05$; $0.33 < \lambda < 0.55$).

In all, 56 frequency distributions of scour and fill were recorded over the three-year study period (28 for scour, 28 for fill). Of these, 15 were characterised by only two bins and thus could be modelled successfully by several probability distribution functions. Of the remaining 41 distributions, 26 (63%) can be fitted by the exponential function. This suggests that the exponential model may give a satisfactory approximation of sediment exchange depth distributions in sand-bed streams where scour is not restricted by sediment availability.

Conclusions

This study has investigated reach scale patterns of scour and fill in sandy, dryland streams. Although significant variability in the depth of scour and fill has been recorded at the scale of the channel reach, the depths of scour and fill are approximately equal at each measurement location. There is some evidence of spatial organisation in scour depths though the limited length of the spatial series hampers quantitative confirmation of this. The exponential density function describes the spatial variation in event-based scour and fill depths for the majority of the events examined. Given that stream powers were generally greatly in excess of entrainment thresholds, this finding is somewhat surprising since it implies that significant areas of the streambeds experienced little, if any, bed activity during an event. This may reflect the fact that much of the data was collected during floods of low-medium intensity. Confirmation of the appropriateness of the exponential model for characterising the distribution of scour and fill depths in sand-bed streams awaits further testing using data obtained at higher magnitude flows in higher order channels.

Acknowledgements

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Hydrology II

Infiltration and Runoff: Point and Plot Scale

Ginger Paige, Jeffry Stone

Abstract

Point scale measurements of infiltration and plot scale measurements of infiltration and runoff were made on three, 2 m by 6 m, rainfall simulator plots on an instrumented sub-watershed within the USDA-ARS Walnut Gulch Experimental Watershed (WGEW). Point measurements were made at three different pressure heads using a tension infiltrometer. Plot scale infiltration and runoff measurements were made using a variable intensity rainfall simulator at a range of intensities and two different soil moisture conditions. A distributed, process-based hydrologic model was used along with measured plot characteristics to determine distributed infiltration parameters using the plots as micro-watersheds. The objective of the study was to determine if tension infiltrometer measurements would lead to similar estimates of infiltration and hydraulic conductivity as the rainfall simulator measurements. Differences in infiltration rate and calculated hydraulic conductivity values were found between the two methods. The implications of measurement method, scale, and the complexity of hydrologic processes are discussed.

Keywords: infiltration, runoff, scale, rainfall simulator

Introduction

The relationship between hydrologic processes and scale is one of the more complex issues in surface water hydrology. Infiltration processes are often measured at the point or plot scale while landuse managers and hydrologic models often are interested in rainfall-runoff processes at the hillslope or watershed scale. The measurement of the infiltration process and quantification of its spatial variability is difficult due to inherent differences with the measurement methods and the scales at which they are applied (Merzougi and Gifford 1987, Paige and Stone 1996). In general, the spatial variability of infiltration decreases with increasing measurement scale (Sisson and Wierenga 1981) and its importance and impact on runoff and erosion decreases as the magnitude of the rainfall increases (Goodrich 1990).

Goodrich et al. (1996) presented a good relationship among hydraulic conductivity estimates from tension infiltrometer, rainfall simulator, and small catchment measurements on the Lucky Hills brush dominated rangeland site. These estimates were determined from different studies conducted on the same rangeland site, but not the same locations. For this paper, two of those measurement methods were used to measure infiltration on the same locations on a semiarid rangeland watershed. The methods, a tension infiltrometer and a variable intensity rainfall simulator measure infiltration at different scales and under different conditions.

The objective of the study was to determine if point measurements of infiltration distributed over a rainfall simulator plot using a tension infiltrometer and rainfall –runoff measurements from a rainfall simulator on these same plots would yield similar estimates of infiltration and hydraulic conductivity. Tension infiltrometer measurements were made within three of five rainfall simulator plots on an instrumented grassland sub-watershed. The results from the two methods are compared and evaluated with each other in the context of rainfall-runoff process at the hillslope and watershed scales.

Methods

The research for this study was conducted on Kendall watershed 112 within Walnut Gulch

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Experimental Watershed (WGEW). Kendall 112 is a zero order grassland watershed of 1.91 hectares with an average slope of 9.4%. Kendall 112, as well as the entire WGEW, is within the Natural Resources Conservation Service (NRCS) Major Land Resource Area (MLRA) 41-3. It is classified as a Loamy upland - Limy slopes. Loamy upland is the dominant classification with inclusions of Limey slopes. The soils are mapped as an Elgin-Stronghold complex. In general, the soil complex is classified as a gravelly fine sandy loam with slopes ranging anywhere from 3 to 30 percent (NRCS 1993). The average measured soil bulk density is 1.40 g/cm^3.

Lane et al. (1995) identified three overland flow paths, one on each of the three hillslopes within the sub-watershed. Each profile originates at the upper boundary of the hillslope and terminates at the outlet of the watershed. Infiltration measurements were made along profile 1 using a disc permeameter to determine the variability of saturated hydraulic conductivity (Ks) along the profile (Gallo 2000). The resulting infiltration rates and Ks values from the ponded infiltration measurements were very high (70 to 330 mm/h).

Rainfall simulator experiments were conducted on five 2 m by 6 m rainfall simulator plots using a variable intensity rainfall simulator that applies intensities between 50 and 178 mm/h. Plots 1-3 were installed along profile 1 and plots 5 and 6 were installed on an adjacent hillslope, along profile 2. The vegetative canopy and surface ground cover were measured at 480 points on each plot. Two rainfall simulator runs, a dry run under initial soil moisture conditions and, one hour later, a wet run, were conducted on each plot using the prototype of the Walnut Gulch Rainfall Simulator (WGRS) (Paige et al., in review). For each simulator run, the rainfall application was continuous and started with the higher intensities and decreased incrementally to 50.8 mm/h. Each rainfall intensity was applied until steady state runoff was maintained for a minimum of 5 minutes. The steady state infiltration rate was calculated for each rainfall intensity by subtracting the observed steady state runoff rate from the applied rainfall intensity (Paige et al. 2002).

Point scale infiltration measurements were made on three rainfall simulator plots along profile 1 (Plots 1-3) using a tension infiltrometer. Measurements were made at three different negative supply heads, 3 cm, 5 cm, and 10 cm and were made at a minimum of three locations down the length of each plot using all three pressure heads. The measurements were made on "soil" areas within each plot. Loose gravel and litter were removed, being careful not to disturb the soil surface. Infiltration rates were measured continuously at a single location starting with a 10 cm negative pressure head. Once steady-state infiltration was observed, the pressure head was changed. Initial and final soil moisture measurements were made using gravimetric samples.

Hydraulic conductivity parameters

Hydraulic conductivity parameters were determined using the steady-state infiltration rates from the infiltrometer and rainfall simulator measurements. The method used to calculate the hydraulic conductivity from tension infiltrometer infiltration measurements was presented in Reynolds and Elrick (1991). The unsaturated hydraulic conductivity, $K(\Psi)$, is determined from two or more measurements (Q_1, Q_2, Q_3,...) made at different supply heads (ψ_1, ψ_2, ψ_3,...) at the same location.

The hydrologic simulation model KINEMAT, a research version of the KINEROS2 (Smith et al. 1995), using the Green-Ampt Mein-Larson (GAML) equation (Mein and Larson 1973). The model was used as a tool to determine the effective hydraulic conductivity term (K_e) using the data from the rainfall simulator experiments (Paige et al. 2002).

Two different sets of K_e values were determined for each plot. Each plot was parameterized and modeled as a single plane using a plot average K_e values. The plots were also parameterized using a strip model approach, with the flow length of the planes oriented parallel to the direction of flow. In this case, the parameters and K_e values for each plane were based on the measured plot cover characteristics and the observed runoff rate. For the bare soil areas, the K_e value was determined from the observed time to ponding. In both cases, the K_e values were determined using the measured runoff volume from the dry runs and validated using the runoff volumes for the wet runs. Details of the methods used to determine the hydraulic conductivity values and plane discretizations for each of the plots using the model were presented in Paige et al. (2002).

Results and Discussion

There was a large range in infiltration rates from the point scale measurements made using the tension infiltrometer. The infiltration rates are lowest for the 10 cm pressure head and increase with decrease in negative pressure head as one would expect (Table 1). The infiltration rates not only varied among plots and among pressure heads but the Coefficient of Variability (CV) of the replicates ranged from 0.01 to 0.76. Plot 3 had the lowest average infiltration rates for each of the pressure heads but the highest CVs.

Table 1. Average infiltration rates from the tension infiltrometer measurements on the 3 plots. The CV is in parentheses.

	infiltration rate (mm/h)		
Tension	Plot 1	Plot 2	Plot 3
10 cm	6.2	10.5	4.9
	(0.05)	(0.23)	(0.76)
5 cm	16.2	26.9	12.6
	(0.01)	(0.22)	(0.23)
3 cm	29.9	41.4	19.2
	(0.20)	(0.26)	(0.28)

The relationship between the applied tension and the measured infiltration rates for the three plots is presented in Figure 1. Fitted power functions are used to illustrate the relationships among the plots. The infiltration rates on plot 2 were consistently higher than the other 2 plots. The rates from the 10 and 5 cm tensions for plots 1 and 3 are similar; however, the fitted curve for plot 3 is flatter and there is an increased difference as the tension decreases.

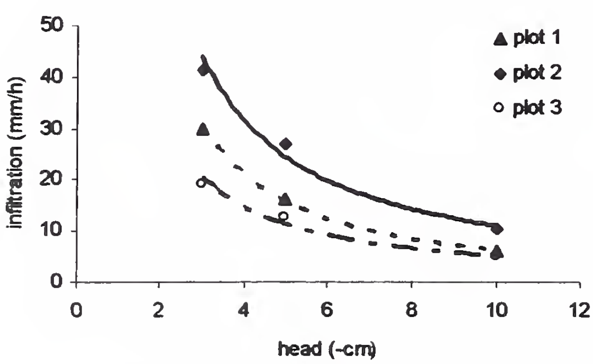

Figure 1. Tension - infiltration curves from the point measurements.

The steady-state infiltration rates from the rainfall simulator experiments were determined for each rainfall intensity applied for both the dry and wet rainfall simulator runs. The steady state infiltration rates from the rainfall simulator runs increased with increasing rainfall intensity indicating the spatial variability of the infiltration capacity across the plot (Hawkins 1982, Paige et al. 2002). The rates were higher for the dry runs than for the wet runs as expected due to the differences in antecedent soil moisture, and there was a difference among plots in the range of infiltration rates (Table 2). An infiltration rate equal to the rainfall intensity means that there was no observed runoff and that the infiltration capacity is greater than the applied intensity for that antecedent moisture condition. The fact that the infiltration rate was still increasing at the higher application rates indicates that even at 177.8 mm/h there are portions of the plots that are not contributing to the measured runoff and have an infiltration capacity greater than 177.8 mm/h.

Table 2. Steady state infiltration rates as a function of rainfall intensity calculated from the rainfall simulator experiments for the dry and wet runs.

	Rainfall intensity (mm/h)	Dry run Infiltration (mm/h)	Wet run infiltration (mm/h)
Plot 1	177.8	93.8	57.0
	127.0	65.3	42.9
	76.2	42.2	35.8
	50.8	38.9	33.3
Plot 2	177.8	121.7	101.3
	127.0	98.9	83.3
	76.2	72.9	68.2
	50.8	50.8	50.8
Plot 3	177.8	80.6	61.9
	127.0	67.2	47.2
	76.2	58.7	42.2
	50.8	50.8	38.9

Hawkins (1982) suggested a relationship between the infiltration rate, $f_s(i)$ (mm/hr), and application rate, i (mm/hr), assuming an exponential distribution of infiltration capacity over an area as

$$f_s(i) = u_f\left(1 - e^{-\frac{i}{u_f}}\right) \quad (1)$$

where u_f (mm/hr) is the average aerial infiltration rate when the entire area is contributing to runoff. This relationship is illustrated in Figure 2 using the results from the wet rainfall simulator runs. The infiltration rates from plot 2 are consistently higher and the intensity - infiltration curve is increasing even at the high intensities.

The curves from plots 1 and 3 are very similar and appear to level out at an intensity of about 180 mm/h. This indicates a plot average infiltration capacity of approximately 50 mm/h.

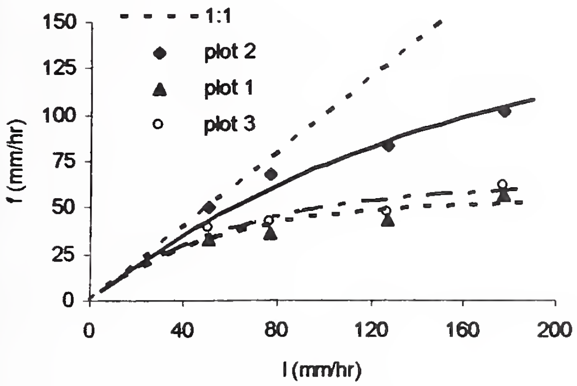

Figure 2. Intensity - infiltration curves from the wet rainfall simulator runs.

Tension infiltrometers measure infiltration at the point scale, in this case a 314 cm^2 area, using a constant pressure head. The infiltration rates determined using the rainfall simulator are averaged over a larger area (12 m^2- in this case) and the pressure head at the soil surface is spatially varied.

The point infiltration rates measured with the tension infiltrometer are, in general, much lower than the plot average infiltration rates calculated from the rainfall simulator runs. For plot 1, the average infiltration rate at 3 cm of tension (29.9 mm/h) is just slightly lower than the 33.3 mm/h infiltration rate for the wet run on plot 1 at 50.8 mm/h intensity. The same relationship held true for plot 2 as well, but not for plot 3. There does appear to be a common trend in both measurement results. The measured infiltration rates for both methods are higher for plot 2 than plot 1 or 3 and the resulting rates from plots 1 and 3 are very similar. This is evident in Figures 1 and 2.

Hydraulic conductivity

The range of average hydraulic conductivity values calculated from the tension infiltrometer measurements was similar to the infiltration rates as one would expect. The values range from 4.1 mm/hr on plot 1 at 10 cm of tension to 33.7 mm/hr on plot 2 at 3 cm of tension (Table 3). The CVs ranged from 0.07 to 0.46, a smaller range than for the infiltration rates (Table 1); however, they still indicate a significant amount of variability among the measurements.

Table 3. Average hydraulic conductivity from the tension infiltrometer measurements on the 3 plots. The CV is in parentheses.

	hydraulic conductivity (mm/h)		
Tension	Plot 1	Plot 2	Plot 3
10 cm	4.1	6.4	4.2
	(0.15)	(0.23)	(0.46)
5 cm	12.4	21.1	8.6
	(0.07)	(0.12)	(0.26)
3 cm	28.0	33.7	15.6
	(0.31)	(0.27)	(0.40)

The hydraulic conductivity values calculated from the dry runs of the rainfall simulator experiments were in general, much higher than those from the tension infiltrometer (Table 4). The single plane K_e values are very high, 26.7 to 52 mm/h, and the multiple plane K_e values range from 12 mm/h to greater than 178 mm/h. As with the results from the tension infiltrometer, plot 2 had the highest K_e values. The single plane values were similar for plot 1 and 3 for the single plane; however, the values for the multiple plane configurations are very different.

Table 4. Hydraulic conductivity values determined from the dry runs of the rainfall simulator experiments. The representative areas for each plane are in parentheses.

	hydraulic conductivity (mm/h)		
	Plot 1	Plot 2	Plot 3
Single plane	26.7	52.0	28.0
Multiple plane			
Bare soil area	26.7 (9.0%)	33.2 (13.3%)	12.0 (58.3%)
Cover area	22.9 (82.25%)	58.8 (77.3%)	102.0 (41.7%)
Shrub area	NC* (8.75%)	NC (9.4%)	N/A**

* NC means that the plane has a K_e value greater than the applied rainfall intensity and is therefore not contributing to runoff.
** Plot 3 had no shrubs.

The K_e values from the rainfall simulator experiments were not calculated directly from measured infiltration rates but indirectly by matching the measured runoff volume using the hydrologic simulation model (Paige et al. 2002). This is especially important to note when evaluating the single plane K_e values. These values represent the average conductivity rates for these plots for a large range of rainfall intensities (50 to 177 mm/h) that were applied during each simulator run. Using these plot average parameters in the simulation model, the runoff volume was matched however the runoff hydrograph was overestimated for the peak flow at the highest intensity and underestimated at the low intensities (Paige et al. 2002).

The K_e values for the multiple plane configurations show the same relationship among the plots. The values for plot 2 are consistently higher. The multiple plane approach resulted in large range in K_e values for each plot and in general a much better fit of the observed runoff hydrographs for both the dry and wet simulations (Paige et al. 2000). From the calculated infiltration rates (Table 2), it was known that there were portions of each plot that were not contributing to the observed runoff. Therefore, it was assumed that the shrub portion of each plot had an infiltration capacity greater than the highest applied intensity (Paige et al. 2002).

There is overlap in the K_e values from the tension infiltrometer and the derived values for the multiple plane configurations; the 5 and 3 cm values ranged from 9 to 34 mm/h while the bare soil and cover area values ranged from 12 to 102 mm/h. The 10 cm results were much lower than any of the parameters determined from the rainfall simulator experiments, indicating that the infiltration rate of the soil during rainfall is greater than the measured infiltration rate at this tension. In general, there is no clear relationship between the results from two methods and their application range at this site.

The K_e values from the tension infiltrometer and rainfall simulator measurements were both determined from steady state infiltration rates on the same plots; however, they were determined from different methods, measuring different processes at different scales at a range of tensions and intensities. The tension infiltrometer directly measured the infiltration rate of the soil under different pressure heads over a 314 cm^2 area, while the rainfall simulator indirectly measured the infiltration rate of the soil and vegetation components of the plot over a 12 m^2 area.

Both methods have advantages and limitations. Point measurements using a tension infiltrometer can be used to quantify the variability of infiltration within an area, however, they do not account for the runon-runoff processes that can occur during rainfall infiltration. The rainfall simulator results are plot averages and do not necessarily reflect the variability of infiltration capacities that can occur within the plot. However, by using a range of rainfall intensities, one is able to define the range of infiltration rates for that plot. Results from several plots (3 to 6) across a hillslope should be able to characterize the ranges in infiltration and runoff from a large range of rainfall intensities.

In an earlier study, Goodrich et al. (1996) presented good agreement between the tension infiltrometer and rainfall simulator results from Lucky Hills. The results, however, were from a single intensity (60 mm/h) rainfall simulator run and a single tension infiltrometer measurement at 5cm of tension. In this study, the methods were applied at a range of application rates or tensions. Though there is a correspondence between the infiltration rates at 3 cm tension and an intensity of 50.8 mm/h, the relationship between the measurement tension and rainfall intensity is still unclear. A modeler does not know apriori to use an infiltration parameter from 7 cm or 4 cm of tension to parameterize a simulation model.

Conclusions

The results from both measurement methods illustrated the variability of infiltration rates within the rainfall simulator plots, as well as the differences in infiltration rates among the plots. However, it evident from the results that the two methods are measuring different processes and that the merits of one method over another would be application dependent.

To measure the infiltration rate of the soil and quantify its spatial variability across an area, point measurements using a tension infiltrometer could be used. The measurements are simple and easy to make and do not require a lot of resources. However, the relationship among these measurements and the infiltration and runoff processes at the plot scale and larger is still unknown.

Plot scale measurements using a variable intensity rainfall simulator are expensive, time consuming, and require more personnel than the tension infiltrometer measurements. However, significant information can be obtained from these measurements in terms of the infiltration, runoff, and erosion processes that occur at the plot and hillslope scale. Land use managers are interested in sustaining the long-term productivity of the soil and vegetation resources; this includes minimizing runoff and soil loss, and increasing infiltration and biomass. The productivity of a site is often evaluated at the hillslope scale.

Acknowledgments

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Watershed-Scale Sensing of Subsurface Flow Pathways at the OPE3 Site

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Abstract

The Optimizing Production inputs for Economic and Environmental Enhancement (OPE3) research program focuses on developing strategies for meeting economic crop production goals while mitigating excess chemical loss to neighboring ecosystems. The OPE3 site has four hydrologically-bounded watersheds, about 4 ha each, which feed a wooded wetland and first order stream. Among the OPE3 watershed-scale research projects seeking to meet these goals are investigations focused on methods to quantify spatial variations of water and nutrients via characterization of subsurface flow pathways and analysis of crop response. Subsurface topography of the first continuous clay lens was determined by combining ground-penetrating radar (GPR) data with surface digital elevation maps to identify the spatial location of subsurface convergent flow pathways. Remote sensing provided information on crop nitrogen status and foliage density via spectral vegetation indices (SVI) from an airborne imaging system. Both the spectral and spatial information domains of imagery are being used to map LAI and leaf chlorophyll at high spatial resolutions (1 to 4 m pixels). These analyses provide links for mapping the impact of soil water dynamics over several growing seasons. The maturation and fusion of these technologies will permit an assessment of watershed strategies influencing water and chemical flows and their impacts on surrounding ecosystems while simultaneously assessing the effectiveness of management strategies on crop production.

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Keywords: subsurface water flow, yield patterns, remote sensing, preferential flow

Introduction

Plant growth and chemical behavior are strongly influenced by surface and subsurface soil water dynamics. Classical quantification of field-scale subsurface water movement and chemical transport was dependent upon knowledge of the spatial distribution and autocorrelation of soil hydraulic properties and subsurface soil layering structures. Unfortunately, traditional methods to assess the spatial nature of soil hydraulic properties are of limited benefit because only a fraction of the subsurface is sampled. As a result, it is impossible to ascertain the spatial behavior of soil hydraulic properties using point data because the sampling density of soil core and well log data is below the inherent spatial variability of soil hydraulic properties.

Abrupt changes of soil texture or density across the boundary of two adjacent soil layers creates a discontinuity of pores. A mismatch of both pore-entry value and soil hydraulic conductivity across this boundary can trigger funnel flow (Kung 1990, Kung 1993, Ju and Kung 1993). Under this latter condition, uniform matrix flow processes could congregate and become a preferential flow process, especially when the subsurface restricting layer is inclined. By evaluating changes of soil dielectric properties Kung and Lu (1993) and Casper and Kung (1996) showed that GPR can detect the size, inclination, and spatial pattern of subsurface layers. Recently, Gish et al. (2002) showed that subsurface convergent flow pathways could be identified with GIS analysis of ground-penetrating radar data and digital elevation maps. A knowledge of the GPR-identified flow pathways was utilized to characterize corn growth pattern (Walthall et al. 2001) and to understand surface-subsurface nitrogen transport (Daughtry et al.

2001). In addition, Angier et al. (2001) found that the GPR-identified flow pathways corresponded with the up-welling zones in the adjacent riparian ecosystem. These results may make it possible to accurately quantify subsurface water and chemical fluxes exiting agricultural land and entering neighboring ecosystems.

The objective of this paper is to present a brief overview of the OPE3 program and to show the relevance of subsurface flow pathways on crop yield during water-limited growing seasons.

Materials and Methods

Site description

The research site is a 25 ha agricultural production field located at the USDA, Beltsville Agricultural Research Center, Beltsville, Maryland (39° 01' 00" N, 76° 52' 00" W). The site is part of the Optimizing Production inputs for Economic and Environmental Enhancement program (OPE3) containing four small bounded watersheds, approximately 4 ha each, with earthen berms which feed a wooded riparian wetland and first-order stream (Fig. 1). The watershed has a sandy loam surface and is relatively flat with 73% of the site having slopes < 2% and 1% of the site having slopes > 3%. The watershed drains into a riparian wetland forest which contains a first-order stream.

For the first year of this study the same management treatment was applied to all four watersheds to assess watershed similarities. The variability of yields between watersheds was surprisingly similar (Table 1). The corn grain yield of each watershed ranges from < 900 kg/ha to over 7,900 kg/ha. Furthermore, the yield distributions for each watershed were normally distributed with a mean yield of about 3,750 kg/ha. Thus, the watersheds appear to behave similarly.

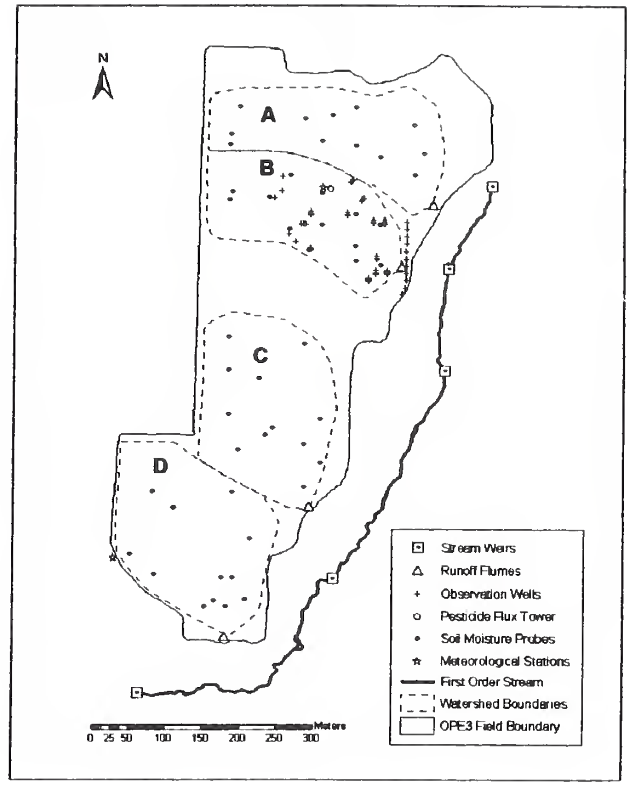

Figure 1. OPE3 schematic showing bounded waterdheds, first-order stream, and instrumentation.

Table 1. Corn Grain Yield Comparison Among Watersheds, 1998.

Watershed	**Mean** (kg/ha)	**Std. Dev.** (kg/ha)	**Min.** (kg/ha)	**Max.** (kg/ha)
A	4,076	1,442	878	7,964
B	3,700	1,568	627	8,466
C	3,449	1,693	377	8,654
D	3,763	1,630	876	8,027

Data collection activities

OPE3 is a small watershed program that is intensively instrumented, and where all data is geolocated with a differential or kinematic global positioning system. Some of the data collection activities include:

- Over 40 km of ground penetrating radar (GPR) data have been collected and analyzed.

- Every two years, soil cores are extracted to determine the spatial correlations and distributions of organic matter, pH, and sand, silt, and clay percentages.

- Electromagnetic induction (EM-31 and EM-38) has been collected for two of the watersheds.
- 36,000 volumetric water content measurements at 48 locations are collected daily.
- Micro-meteorological stations with eddy covariance systems monitor climatic conditions.
- Multiple pesticide vapor flux towers are operational after application.
- Water and chemical (N, P, and pesticides) runoff fluxes are collected from each watershed.
- Watershed B is instrumented with 52 groundwater observation wells.
- Corn grain yields are measured using a grain a yield monitor.
- Aircraft and satellite remote sensing imagery are collected. Ground and tower-based reflectances are collected as needed.
- Plant growth and development are measured periodically during the growing season.
- 180 observation wells in the riparian wetland are monitored for anions and pesticide.
- Stream flows and chemical fluxes in the stream are measured at five stations within the riparian wetland.
- Wetland soil cores were extracted and analyzed for grain size, bulk density, carbon content, hydraulic conductivity, water content, and denitrification potential.
- Dissolved gas is measured in groundwater samples throughout the wetland, for evidence of denitrification and methanogenesis. Dissolved oxygen, dinitrogen, nitrous oxide, and methane are measured.
- Sap flow rates for estimating evapotranspiration are measured on ten trees within the riparian wetland.

The site is also the center of the NASA BARC-EOS and EO-1 Core validation site.

Subsurface flow pathways

Soil moisture is a critical factor governing plant growth and chemical behavior and fate. Consequently, methods for determining the existence and location of subsurface flow pathways is important to optimizing production field management. At OPE3 over 40 km of digital GPR data were used to identify the location of potential subsurface convergent flow pathways. GPR data were acquired for the 25 ha site at two scales of observation: 1) parallel North-South transects, 25 m apart over the entire watershed; and 2) North-South transects, 2 m apart over forty-four, 25 m x 25 m plots within the watershed. The spatial autocorrelation of subsurface reflections potentially restricting water movement was determined using geostatistical software. Omnidirectional semivariograms were produced from point data derived from digitized traces of the depth to the first continuous restricting layer. Semivariogram models provided kriging parameters for subsequent spatial interpolation. The depth of these subsurface reflections was calculated for 10 m x 10 m cells over the entire watershed and subtracted from the DEM data. This calculated a subsurface topographic surface that restricts water flow. The Arc/Info GIS FLOWDIRECTION and FLOWACCUMULATION routines were applied to the subsurface topography to determine potential flow pathways (Gish et al 2002).

Soil moisture monitoring

Two hundred fifty-six soil moisture sensors were distributed between 48 locations to independently monitor the spatial and temporal changes of soil water content throughout the top 1.8 m of the site. Each sensor was calibrated before installation, and programmed to record volumetric water content every 10 minutes (Gish et al. 2002).

Airborne imagery

The Airborne Imaging Spectroradiometer for Applications (AISA) is used to collect visible-near infrared hyperspectral imagery of the site. Color infrared film is also used when very high spatial resolution is required.

Patterns of high crop density during water limited growing seasons have been observed in imagery for areas corresponding to the presence of GPR-identified subsurface flow pathways. Together with correspondingly high yield patterns from the yield monitor data, these crop density patterns are believed to corroborate the location of subsurface flow pathways. Imagery from dry and/or drought years appears especially useful for identifying areas of the field that appear to be continuously irrigated from subsurface sources.

Image-based procedures for chlorophyll density and leaf area index (LAI) mapping at high (1-4 m) spatial resolution are being developed for the hyperspectral data. Procedures to identify the subtle changes of canopy reflectance associated with leaf chlorophyll concentrations are also being developed as a tool for managing the spatial variability of crop N. Remotely sensed leaf chlorophyll concentrations were consistent with measured values over a wide range of soil reflectance and LAI values (Daughtry et al. 2000).

One LAI mapping procedure under investigation seeks to exploit the spatial information content by incorporating semivariograms of spectral vegetation index (SVI) maps with surface-based samples. Another procedure uses a hybrid approach that links SVI maps with radiative transfer model inversions to produce LAI maps. The LAI maps will serve as quantitative measures of crop variability.

The chlorophyll and LAI maps are being investigated as potential surrogate indicators of subsurface flow pathway locations. Image-based procedures may be applicable to other production fields in the absence of logistically difficult and expensive GPR data.

Crop foliage imagery is also being used to infer information about the underlying soil water holding properties. The spatial information of the remotely sensed LAI maps, coupled with weather data and a physical model are being used to investigate procedures for the inversion of soil water holding capacity.

Discussion

A protocol for determining the spatial location of subsurface flow pathways based primarily on GPR and DEM data was developed and successfully tested at the OPE3 experimental research site. Although there is good visual agreement between remote sensing activities and the GPR-identified flow pathways, there is considerable amount of uncertainty due to the manner in which the GPR data were collected. Apparently, uncertainty in the areas where no GPR data were collected increases as the flow network moves downslope.

Confirmation of subsurface water movement through preferential flow pathways can be difficult to asses due to: 1) the spatial and temporal dynamics of water movement; 2) the relatively small volumes of soil used in preferential transport; and 3) the small volumes of soil being evaluated by the soil moisture sensors (i.e. 10 cm diameter). Nonetheless, representative subsurface soil water contents patterns occurring near (<5 m) or far away (>5 m) from the GPR-identified flow pathways are shown Figures 2 and 3. Soil moisture measurements, acquired every 10 minutes, provided a motion picture view of the soil water dynamics at the two locations during the growing seasons.

The soil moisture probe located near a GPR-identified flow pathway and where the GPR data indicated a clay lens at 1.54 m, is shown in Figures 2a and 2b. At this location, a layer of coarse sand and gravel is above the clay lens. In 1998, (d = 126) volumetric water contents changed abruptly at 1.5 m from 0.08 to 0.32 $cm^3\ cm^{-3}$ in less than an 24 h – while volumetric water contents at 1.2 m changed slowly even though both depths had a similar soil textures. Additionally, moisture contents at 1.5 m were more than twice the moisture content at 1.2 m. Thus, the soil water dynamics near a GPR-identified flow pathway fully support the hypothesis of horizontal water flow along the clay lens interface – funnel flow. During the 1999 growing season a large plume of water appears (d = 260) – a consequence of Hurricane Floyd. Although water contents at 1.2 m and 1.5 m rise simultaneously, they remain higher at 1.5 m, indicating a larger plume and potential horizontal water flow. When fluxes are monitored, abrupt changes in soil water content are common with preferential flow processes (Kung et al. 2000).

Since preferential flow typically occupies less than 1% of the available pore space, most of the watershed should experience water movement predominantly as a function of hydraulic conductivities and matrix potential gradients, and as such one would generally expect gradual changes in soil moisture with time. In Figure 3a and 3b, the soil moisture probe is located > 5 m away from a GPR-identified flow pathway and where the clay lens is situated 1.6 m below the soil surface. Several rain events occurred in August of 1999 which increased the subsurface water contents by only a few percent. However, little increase in subsurface moisture was noted during Hurricane Floyd (d = 260), suggesting that subsurface water bypassed this site where the probe was located. Other than the soil moisture changes at d = 242 in 1999, all subsurface moisture observations changed gradually. In summary, no preferential flow was observed in the soil moisture probes located > 5 m from a GPR-identified flow pathway for identical weather patterns and similar soil conditions.

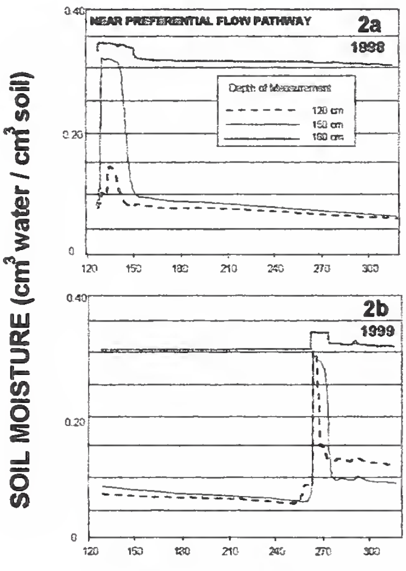

Figure 2. Real-time soil moisture observations near the GPR-identified flow pathway.

Relationship between yield and flow pathways

Corn grain yields during a mild drought are also influenced by the GPR-identified flow pathways. Corn grain yields decrease with increasing distance from the subsurface flow pathways (Fig. 4).

Since corn grain yields are influenced by the subsurface flow pathways a moisture response index, MRI, is being developed and tested to identify those areas of the field and the degree to which they are influenced by subsurface soil moisture. The MRI uses yield monitor data from wet and dry years to generate an index that may help refine the location and shape of the GPR-identified flow pathways.

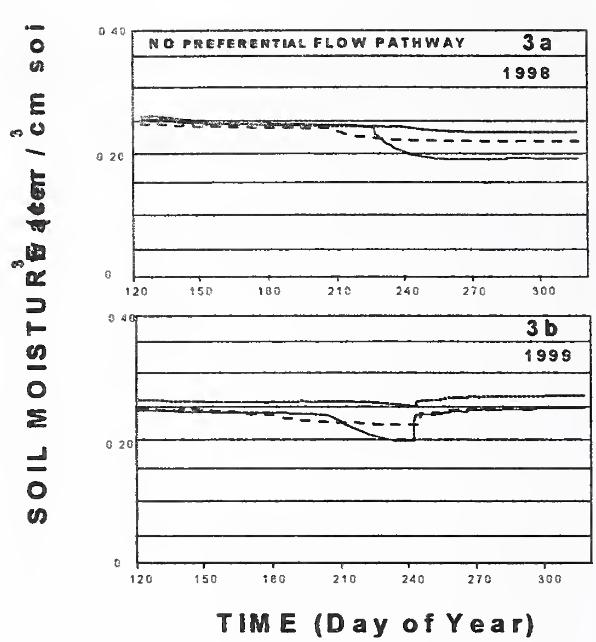

Figure 3. Real-time soil moisture observations >5 m from a GPR-identified flow pathway.

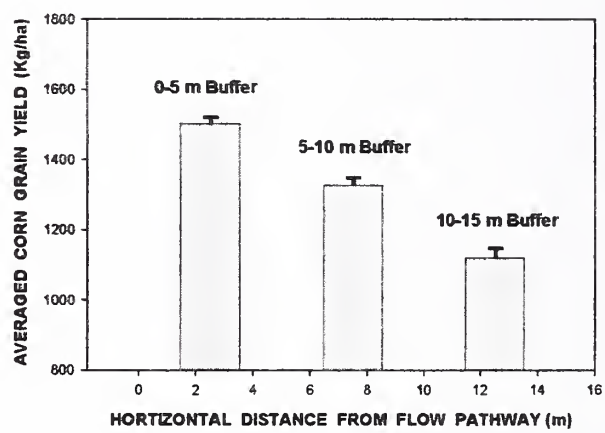

Figure 4. Impact of distance from subsurface flow pathways on corn grain yield during a mild drought. Data presented is for watershed B. Error bars indicate standard error of the means.

Conclusions

The OPE3 program demonstrates that georeferenced GPR data sets on a sandy soil have great potential to locate soil layers which control subsurface water flow, crop yield patterns, and perhaps field-scale chemical behavior and fate. Real-time soil moisture data supported the existence of funnel flow processes, which indirectly confirmed the existence of restricting layers. These techniques may have the capacity to monitor and evaluate subsurface water pathways which are necessary to determine agrichemical fluxes beyond the root zone. Additionally, soil moisture data coupled with GPR-identified flow pathways suggest that: 1) a coupling of GPR data with real-time soil moisture monitoring may be an effective tool for evaluating and monitoring subsurface flow processes; 2) the spatial location of the soil moisture monitoring system is critical to monitoring water movement; and 3) real-time monitoring of water movement is critical if preferential flow pathways are to be accurately monitored.

Acknowledgments

Manuscript reviews and help with the preparation came from Dr. Ali Sadeghi and Andrew Russ.

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The Challenges of Measuring Rainfall: Observations Made at the Goodwin Creek Research Watershed

Lisa Sieck, Matthias Steiner, Stephen Burges, James Smith, Carlos Alonso

Abstract

A major storm passing over the 21.4 km^2 Agricultural Research Service (ARS) Goodwin Creek experimental research watershed in northern Mississippi on April 23-24, 2001 is used as a case study to highlight uncertainties associated with using hydrological and hydrometeorological data from various remote-sensing and point sources at greatly differing space-time resolution and coverage. Instrumentation of the research watershed includes approximately 45 rain gauges of various designs (above ground and buried), a raindrop spectrometer, and stacked anemometers to observe the wind profile at the climate station in the center of the catchment. A local-scale mobile Doppler radar was also deployed to record very high-resolution precipitation observations (50 m by 1 degree in space, tens of seconds in time) in both the vertical and horizontal directions over the catchment. These data, along with regional-scale lower-resolution observations from the Memphis WSR-88D (KNQA) radar (1 km by 1 degree in space, several minutes in time), are utilized to analyze the storm. The difficulties of obtaining accurate measurements and of merging observations made by point and remote sources are discussed. The basin response to the storm is illustrated with runoff measurements at the basin outlet.

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Keywords: rainfall estimation, uncertainties, radar, rain gauge, drop size distribution

Introduction

Monitoring the water budget on a catchment requires an accurate representation of rainfall and its variability. Such representations can be used for flood and erosion mitigation. Precipitation is highly variable in space and time, and this variability affects our capability to build a complete picture of rainfall reaching the surface from either in-situ or remote sensing measurements. The lack of continuous observations in space and time requires a merging of information obtained from various point (e.g., rain gauges) and remote sources (e.g., radar) at differing resolution, coverage, and accuracy.

A central issue of the merger of point and remote rainfall information is how much of the observed variance between radar-based estimates and rain gauge measurements can be attributed to sensor resolution differences. For example, both sensors may yield precise measurements at their respective resolution yet the two observations are likely not identical. Detailed observations of rainfall in space and time were thus carried out over the small 21.4 km^2 ARS Goodwin Creek watershed (Figure 1) in northern Mississippi (Alonso 1996, Steiner et al. 1999) to address this question. The instrumentation at this site includes approximately 45 rain gauges of varying design, a Joss-Waldvogel (1967) raindrop spectrometer (often called disdrometer), four anemometers mounted at different heights above the ground to observe the wind profile, and a Surface Radiation (SURFRAD) network station (Hicks et al. 1996) to determine the energy budget located in the center of the catchment (latitude 34° 15' 16" N, longitude 89° 52' 26" W) (Figure 2). A mobile "Doppler-on-Wheels" (DOW) radar system (Figure 3) was used to provide high-resolution storm

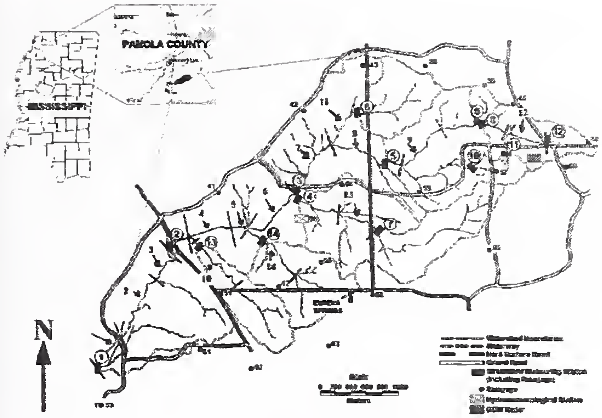

Figure 1. Geographical location and instrumental setup of the ARS Goodwin Creek research watershed in northern Mississippi.

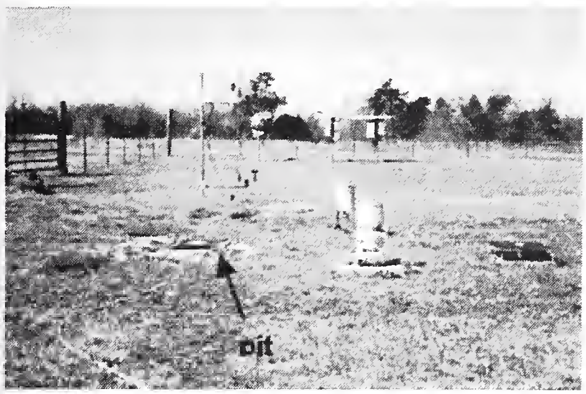

Figure 2. Climatological station (station 50) in the catchment center includes above-ground and buried/pit rain gauges, a disdrometer, wind profile measurements, and a SURFRAD station.

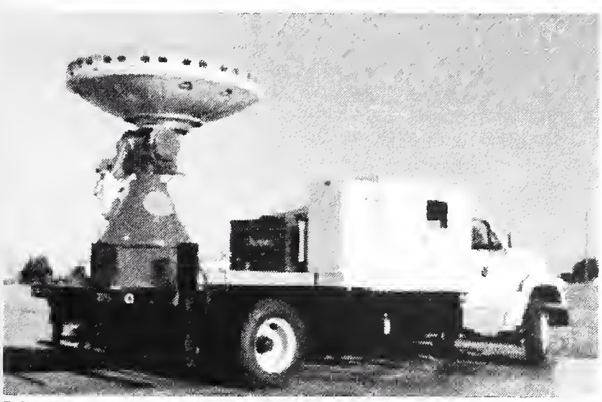

Figure 3. Doppler-on-Wheels (DOW) radar of the University of Oklahoma (deployment site marked by rectangle at upper end of catchment in Figure 1).

intensity and motion observations for several storms passing over the Goodwin Creek area. The DOW operates at a 3 cm wavelength (X band), providing reflectivity and radial velocity data at 50 m by 1 degree resolution in space and updates within tens of seconds in time (Wurman et al. 1997). The Goodwin Creek watershed is also under coverage from four Weather Surveillance Radar – 1988 Doppler (WSR-88D) radars (Heiss et al. 1990). These radars operate at a 10-cm wavelength (S band) and collect data at a resolution of 1 km by 1 degree in space and several minutes in time. The closest of these WSR-88D (i.e., KNQA) is located near Memphis, Tennessee, approximately 120 km to the north of the Goodwin Creek catchment.

These data are used to characterize and discuss the difficulties of obtaining accurate measurements of rainfall reaching the surface by means of rain gauges and radar. In particular, we evaluate measurement issues associated with the rain gauge catch (e.g., calibration and wind effects) and radar rainfall estimation (e.g., calibration, signal attenuation, and reflectivity to rain rate conversion). Analyses of the 23-24 April 2001 storm illuminate the high variability of rainfall in space and time and limitations of using short-wavelength (X band) radar for hydrologic applications.

Storm Analyses

The storm that crossed Goodwin Creek on April 23-24, 2001 was part of a major storm system that extended from southern Texas to Canada (Figure 4). It was well organized, with an intense line of convection (squall line) followed by some widespread (stratiform) rainfall. The storm passage over Goodwin Creek is reflected in the rainfall trace shown in Figure 5 indicating that there was an initial rainfall shower before the most intense part of the storm. The catchment's response to this rainfall event is shown in the bottom panel of Figure 5. The radar and rain gauge analyses are discussed in following sections.

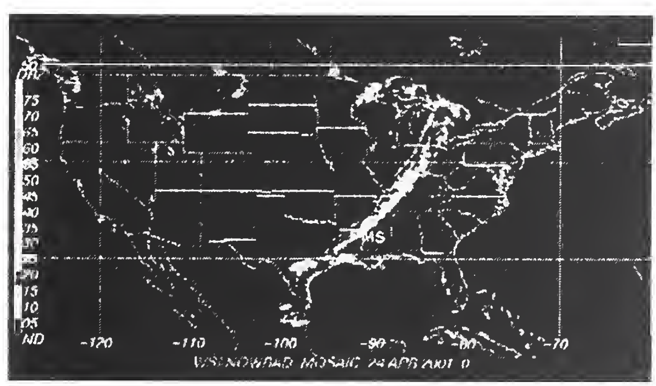

Figure 4. Weather Services International radar reflectivity mosaic of storm on 23-24 April 2001 at 0000 UTC as it passes over Northern Mississippi.

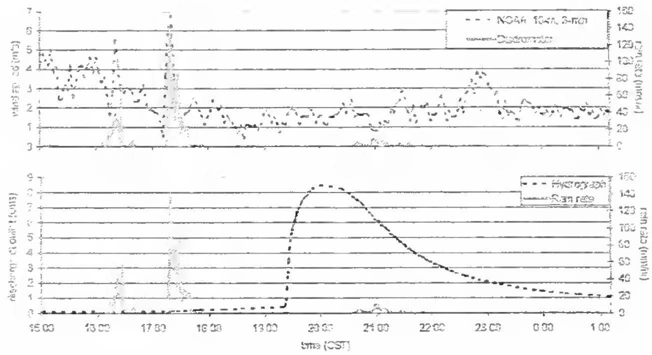

Figure 5. The top panel shows the 10-m wind speed (dashed) plotted with rain rate from disdrometer (solid) in the center of catchment (station 50). The discharge at the outlet of Goodwin Creek (station 1) is shown in the bottom panel.

Radar analysis

Figures 6 through 9 reveal key aspects of the storm as seen by the local DOW and remote Memphis KNQA radars. Figure 6 shows a horizontal snapshot of the storm on April 23, 2001, 2201 UTC. The left and right panels show the reflectivity observations made by the DOW deployed at the eastern end of the watershed (see Figure 1) and the KNQA radars, respectively. The KNQA reflectivity is shown only for the area covered by the DOW. The KNQA data have been adjusted in time to minimize the root-mean-square difference between DOW and KNQA observations.

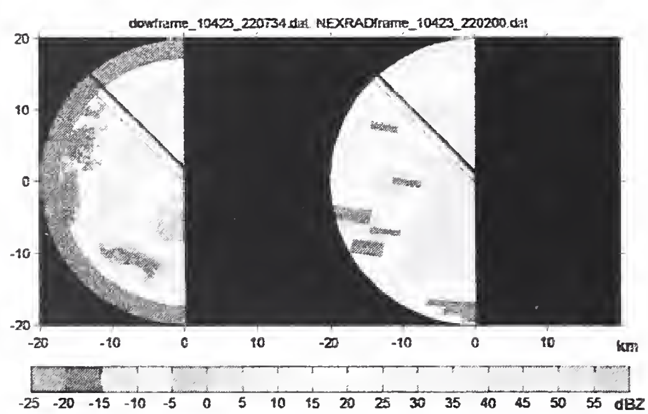

Figure 6. Horizontal cross-section of storm reflectivity as seen by the DOW at 2207 UTC (left) and the KNQA at 2202 UTC (right) radar on April 23, 2001. The dark line denotes the 315° azimuth.

The DOW provides an order of magnitude increase in spatial resolution over the KNQA radar. At this finer resolution, significant small-scale structures within the convective line can be seen that are not resolved by the KNQA. This is especially true for the vertical structure shown in Figure 7.

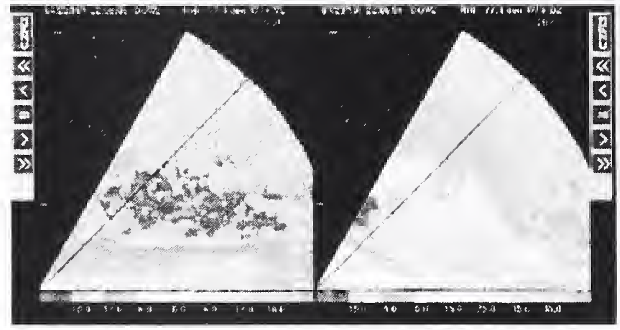

Figure 7. Vertical cross-section of radar radial Doppler velocity (left panel) and reflectivity (right panel) as observed by the DOW radar on April 23, 2001 at 2208 UTC. Range rings are shown at 5 km intervals.

Figure 8 shows a snapshot of the storm on April 23, 2001 at 2319 UTC as seen by the DOW and the KNQA radars during the passage of the most intense part of the storm when rain rates reached 150 mm/h. The vertical cross-section (Figure 9) illustrates how the air is lifted up along the frontal boundary (left panel) and precipitation formed, yet this cross section also demonstrates the severe limitation of radar reflectivity observations made at shorter wavelengths – a complete loss of signal in radial direction behind the intense convective cell (right panel). This attenuation effect can also be seen in the horizontal depiction of the storm by comparing the DOW and KNQA reflectivities in Figure 8.

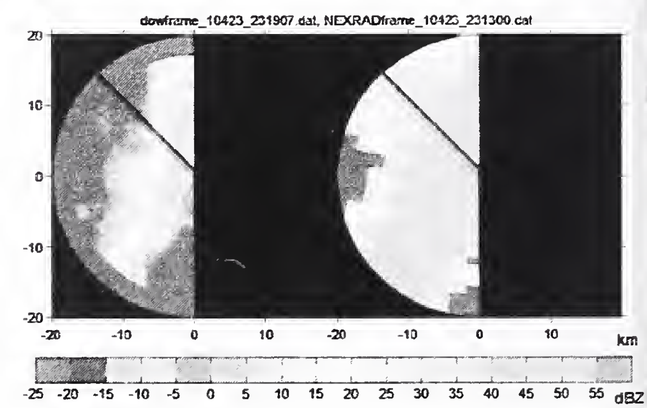

Figure 8. Horizontal cross-section of storm reflectivity as seen by the DOW at 2319 UTC (left) and the KNQA at 2313 UTC (right) radar on 23 April 2001. The dark line denotes the 315° azimuth.

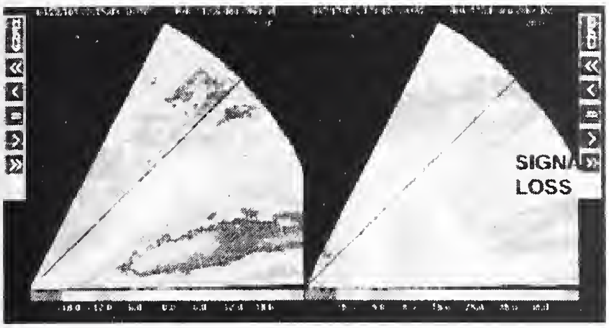

Figure 9. Vertical cross-section of radar radial Doppler velocity (left panel) and reflectivity (right panel) as observed by the DOW radar on April 23, 2001 at 2315 UTC.

The earlier snapshots (Figures 6 and 7) do not reveal an attenuation problem even though the rain rates were also high (approximately 80 mm/h). The difference in attenuation between the two time periods may be explained by the fact that the second and more intense rainfall period was associated with a significant amount of lightning, indicating that this part of the storm included high-density ice particles such as graupel or small hail that cause a different behavior of the radar signal.

Figure 10 shows a time series of reflectivity calculated based on the raindrop spectra at the center of the Goodwin Creek catchment compared to the closest reflectivity pixels observed by both the DOW and KNQA radars. The radar-based intensities nicely trace the observed rainfall at the surface considering the significant differences in sampling volume and sampling frequency. The attenuation problem of the DOW observations can be clearly seen for the passage of the most intense part of the storm (after 2300 UTC). Signal attenuation is less evident during the first rainfall burst (2200 – 2230 UTC).

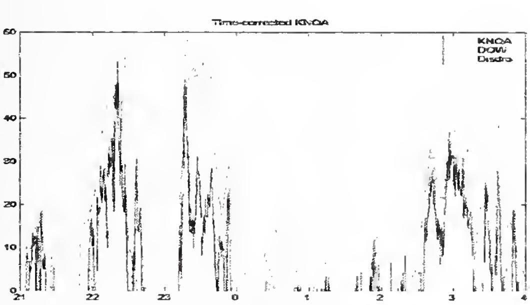

Figure 10. Reflectivity based on disdrometer observations at station 50 and closest pixel of DOW and KNQA radar.

The loss of DOW signal can be estimated with respect to the KNQA observations by comparing DOW and KNQA reflectivities along a common path, and (safely) assuming that the KNQA (S-band) observations are not attenuated. Attempts to quantify the loss of DOW signal in this manner, however, are complicated by the spatial and temporal resolution differences between the KNQA and the DOW – there are approximately 400 DOW pixels that correspond to a single KNQA pixel at any given location and there are more than 20 DOW sweeps in time for each KNQA radar sweep. Figure 11 shows such a comparison along the DOW azimuth 315° (see Figure 8). Note the rapid decrease of DOW signal with distance from the radar, as well as the significant variability with time. DOW signals within 2 km of the radar (vertical line, Figure 11) were not considered for this analysis because of a questionable close-range correction.

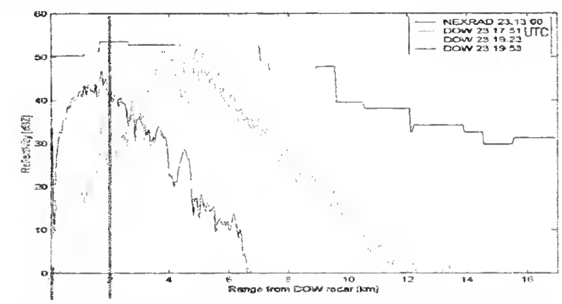

Figure 11. Reflectivity vs. distance from the DOW for both the time corrected KNQA and DOW radar.

Iterative attenuation corrections, constrained by the KNQA and raindrop spectra-derived reflectivities, have not been successful for the most intense storm front, possibly because of the added complexity of high-density ice particles contained in the DOW radar sampling volume. Moreover, this correction procedure depends on the radar calibration. In order to achieve a reasonable calibration, the KNQA and DOW radar reflectivities have been compared to the raindrop spectra based values for weak to moderate rainfall, where the X-band attenuation for the DOW observations should be small.

Point rainfall analysis

The ultimate test of successful data quality control and correction is to compare the radar-estimated rainfall amounts to rain gauge-based surface measurements. The Goodwin Creek rain gauge network consists of several different types of instruments, including Belfort weighing gauges (BEL), Texas Instruments tipping bucket gauges (TXI), USDA Agricultural Research Service tipping bucket gauges (ARS), Australian Hydrologic Service tipping bucket gauges (TB3), and simple buried/pit collectors (COL) that have their rim at ground level (see Figure 2). At least one of each type of gauge was operated at the climatological station in the center of the catchment during this storm. Also, one tipping bucket gauge was mounted above ground (sARS) while another by the same manufacturer was buried (bARS).

The collected water in the BEL gauges was measured after the storm to compare with the amount recorded on the chart. Moreover, water collection devices have been installed for some of the ARS and TXI tipping bucket gauges and the TB3, providing an independent measure of rainfall for these gauges. Detailed calibration curves have been established for most gauges to correct for rain rate dependent effects. Pit collectors (COL) were buried next to several of the gauges to obtain a best estimate of the "true" rainfall reaching the surface.

The rainfall information collected for the 23-24 April 2001 storm passing over the Goodwin Creek watershed is compiled in Table 1. The instruments worked properly, with the exception of a few rain gauges (5, 6, and 62) and one disdrometer (DIS2). Although the calibration for two of the Belfort weighing rain gauges (50 and 57) was somewhat questionable, this gauge type was the most reliable. This is due to its sturdy design and built-in capability for redundant measurements (i.e., chart recording of weighed rain amount, plus collection of total water). The tipping bucket gauges were more prone to malfunction. In addition, the calibration of the ARS gauges varied by as much as ± 10%. The TXI gauges were more stable but one still varied by + 10%. The water from TXI gauges 41, 43, 46 and 65 was collected to check against the cumulative quantity indicated by the gauge tips. Volume collection checks were also made for ARS gauges at stations 1 and 50 (surface and buried). The collection of the water flowing through the ARS and TXI gauges was difficult and, indeed, this check system did not work reliably at gauge 43 for this storm. However, the agreement of collected water amounts with the tip-based rain totals was encouraging. Even for a well-maintained network, rain gauges are prone to malfunctioning, demonstrating the need for multiple gauges at a "point" location to enable cross-checking of values to reveal data inconsistencies.

Table 1. Rainfall accumulations (mm) measured by gauges across the catchment using manufacturer (Mcal) or individual (Ical) gauge calibrations and estimated by the KNQA and DOW radars at the gauge locations. Collected values are shown as well (Collect).

No	Type	MCal	Ical	Collect	KNQA	DOW
1	ARS	22.6	23.4	23.4	27	3.82
1	COL			24.2	27	3.82
2	ARS	23.6	23.1		24.1	4.59
4	ARS	25.4	26.8		24.5	5.45
5	ARS	0.76	0.79		30.8	4.37
6	ARS	3	3.2		31.3	4.86
7	ARS	24.9	27.4		26.2	2.92
8	ARS	30.7	34.2		27.9	3.67
11	ARS	25.1	27.9		30.2	2.70
12	ARS	24.4	22.2		30.2	
13	ARS	26.7	28.4		24.1	4.25
14	ARS	26.1	26.6		24.3	5.02
34	BEL	26.2	-	30.7	31.8	5.31
35	BEL	29.2	-	31	27.9	3.77
41	TXI	23.4	25.6	24.3	24.1	3.69
41	COL			26.2	24.1	3.69
42	TXI	25.4	27.3		31.7	4.88
43	TXI	26.9	28.4	21.6	34.1	4.07
43	COL			29.6	34.1	4.07
45	TXI	26.7	29.3		24.8	1.93
46	TXI	26.2	28.2	27.4	28	
46	COL			27.1	28	
50	sARS	26.2	25	25.1	24.5	4.97
50	bARS	26.9	30	30.5	24.5	4.97
50	TB3	25.2	24.2	26.1	24.5	4.97
50	COL1			26.6	24.5	4.97
50	COL2			27.1	24.5	4.97
50	DIS1	28.4	-		24.5	4.97
50	DIS2	-	-		24.5	4.97
50	BEL	23.9	-	26.7	24.5	4.97
51	BEL	25.1	-	25.1	20.3	3.95
51	COL			25.5	20.3	3.95
52	TXI	24.4	26.4		21.1	4.30
53	BEL	25.1	-	25.1	21.1	2.41
54	BEL	25.1	-	26.9	30.8	4.24
55	TXI	24.4	26.3		33.1	3.82
57	BEL	22.6	-	26.9	31.9	3.07
57	COL			27.1	31.9	3.07
61	BEL	21.6	-	23.1	22.6	3.57
62	TXI	2.5	2.7		22.4	2.53
63	TXI	23.9	25.5		22.2	3.86
64	BEL	24.9	-	25.9	22.9	4.34
64	COL			26.8	22.9	4.34
65	TXI	25.1	27.3	25.7	28.6	2.76
66	BEL	26.7	-	26.7	28	

Another potentially significant source of uncertainty for rain gauge measurements is wind effects on the catch. The wind may come in strong gusts, often associated with the most intense parts of storms (see Figure 5), which makes it difficult to assess quantitatively. For our purposes, we estimated the wind effect on the rain gauge catch by comparing the rainfall amounts of the above-ground to the buried gauges. The undercatch due to wind effect estimated this way ranged from 1%-9% depending on location. Whenever possible, it is recommended that rain gauges be buried to reduce the wind effect.

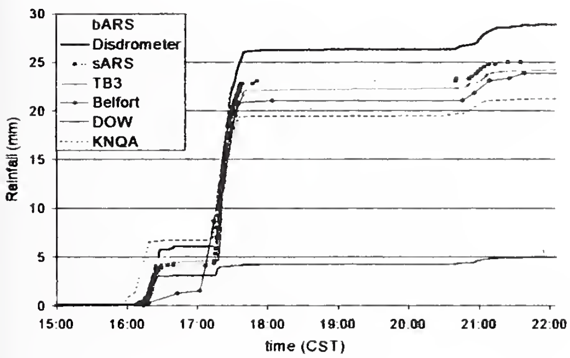

Figure 12. Accumulated rainfall in the center of the Goodwin Creek catchment.

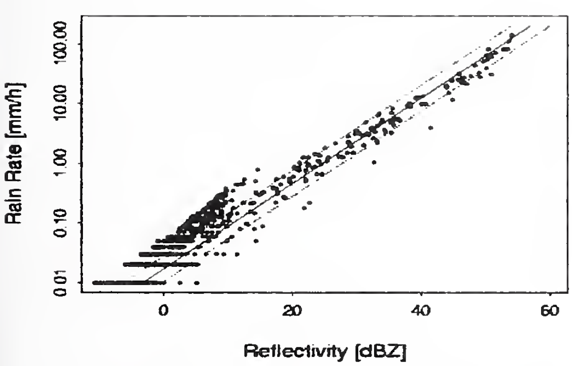

Figure 13. Variability of raindrop size distributions and reflectivity-rain rate relationship for the storm of April 23-24, 2001. Solid line shows a Z-R relationship ($Z = AR^b$) with multiplicative factor A = 300 and power factor b = 1.4, while dotted lines show relationships with A = 600 and A = 150, respectively.

The rainfall accumulations recorded in the center of the watershed are shown in Figure 12. There is significant variation in accumulated rainfall amounts among the various rain gauges, the disdrometer, and the two radars. The 20% variability of accumulated rainfall among the gauges and disdrometer reflects differences in collection mechanisms and wind effects. The radar rainfall estimates shown in Figure 12 and Table 1 are based on the relationship $Z = 300R^{1.4}$ which provides a good fit to the data (Figure 13) based on the raindrop spectra collected in the center of the watershed. While the KNQA radar provides reasonable rainfall amounts, Figure 12 demonstrates the effect of signal attenuation on accumulated rainfall estimates based on the DOW radar. The DOW rainfall estimates amount to less than 20% of the total rain that reached the surface during this storm.

Conclusions

Detailed observations of a major storm system that passed over the small well instrumented ARS Goodwin Creek research watershed in northern Mississippi were used to highlight the range of uncertainty encountered in measuring rainfall from in-situ to remote sensing perspectives. These uncertainties are related to the rain gauge measurements (e.g., calibration, wind effect), radar rainfall estimation (calibration, attenuation, Z-R conversion), and the merging of information from various sources (space and time differences in sampling and coverage). The most accurate spatial rainfall estimates are achieved by combining information from all available data sources.

It is crucial to use only reliable rain gauge information for storm analysis. Rain gauges, especially tipping-bucket gauges, are prone to malfunction. Redundancy is the key to obtaining high-quality rain gauge data. Clusters of at least three rain gauges within tens of meters (or less) are preferable over networks of individual evenly spaced gauges. Clustering allows for cross-checking of data to detect malfunctioning gauges. In addition, rain gauges should be buried if possible to minimize wind effects on the catch.

Mobile short-wavelength radar are increasingly being deployed for rainfall monitoring over urbanizing areas and small catchments. Our study demonstrates, however, that the problem of signal attenuation may seriously limit the quantitative use of such radar for rainfall estimation, especially for situations of intense rainfall that might cause flooding. A correction of radar signal attenuation proves difficult even when additional information is available to constrain an iterative correction procedure.

Analyses of several storms observed in a similar fashion over Goodwin Creek will provide guidance with regard to effectively merging information from various sources to yield the best rainfall measurements. This may include rain gauge and corrected short-wavelength radar observations.

Acknowledgments

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Estimation of Watershed Scale Soil Moisture from Point Measurements During SMEX02

Michael H. Cosh, Thomas J. Jackson, Rajat Bindlish, John H. Prueger

Abstract

Watershed scale soil moisture estimates are necessary to validate current remote sensing products, such as those from the Advanced Microwave Scanning Radiometer (AMSR). Unfortunately, remote sensing technology does not currently resolve the land surface at a scale that is easily observed with ground measurements. One approach to validation is to use existing soil moisture measurement networks and scale these point observations up to the resolution of remote sensing footprints. As part of the Soil Moisture Experiment 2002 (SMEX02), one such soil moisture gaging system, in the Walnut Creek Watershed, Iowa, provided robust estimates of the soil moisture average for the watershed. Twelve in-situ soil moisture probes were installed across the watershed. These probes recorded soil moisture at a depth of 5 cm from June 29th, 2002 to August 19th, 2002. The sampling sites were analyzed for temporal and spatial stability by several measures including mean relative difference and Spearman rank. Representative point measurements were scaled up to the watershed scale (~25 km) and shown to be accurate indicators with low variance and bias of the watershed scale soil moisture distribution. This work establishes the validity of this approach to provide watershed scale soil moisture estimates in this study region for the purposes of satellite validation. Also, the potential errors in this type of analysis are explored. This analysis is an important step in the implementation of large-scale soil moisture validation using existing networks such as the Soil Climate Analysis Network (SCAN) and several Agricultural Research Service watersheds as a basis for calibrating satellite soil moisture products.

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Keywords: soil moisture, instruments and techniques, networks, hydroclimatology

Introduction

Satellite soil moisture products are being developed from new sensors such as the Advanced Microwave Scanning Radiometers on the NASA Aqua and Japanese Midori-II platforms. These products will be the basis for long term global observations of the Earth surface. The calibration of algorithms and validation of these products are of vital importance at this stage in the development of the technology.

For surface soil moisture, two factors make satellite product validation difficult. The first is a mismatch in scale between satellite footprints (1-50 km) and a ground sample (~5 cm). The second is high spatial variability of soil moisture, which is influenced by various land surface and meteorological factors at different scales. Both factors necessitate a large number of distributed observations within a footprint to accurately estimate the average. The issues described above lead to the conclusion that a large number of ground based in-situ samples will be required to validate a single footprint. It would be difficult to provide such information for a large number of footprints. Two approaches have been used in the past. The first is short term intensive field campaigns such as SGP97, SGP99, and SMEX02. These provide reliable estimates but only for a specific subset of physical and climate conditions. Another approach has been to use data from existing in-situ networks. A problem with this approach is the density of the network. Most provide only a single point within a footprint.

Soil moisture scaling theory (Warrick et al. 1977, Russo and Bresler 1980) demonstrates that estimates of a moisture field can be obtained using point observations; however, this requires extensive surface sampling over long periods of time (Kachanoski and De Jong 1988, Vinnikov et al. 1999, Yoo 2002). Geostatistical analyses, such as kriging (Burgess and Webster 1980) and semivariogram analysis, also requires a dense sampling network to adequately portray the spatial character of the soil moisture field. Vachaud et al. (1985) first proposed a method of large scale soil moisture estimation by establishing temporal and spatial stability in a 2000 m^2 grass field in Grenoble, France. This technique investigates the idea that a soil moisture field maintains its spatial pattern over time. If the pattern is stable at long time scales, it is possible to use this pattern to an advantage. The mean of the field at a given time is compared to specific sampling sites within the field to identify locations with a small bias to the mean and a low variability in its relationship to the mean. Once a specific location in an area is demonstrated to accurately estimate the average soil water content for the region, it should be possible to use that point or a reduced number of points for future studies. Their study demonstrated that it is possible to conduct watershed scale soil moisture estimation simply and efficiently. Grayson and Western (1998) extended this research to several additional small watersheds with significant relief ranging in size from 0.1 km^2 to 27 km^2. These included the Tarrawarra catchment (Australia), Chickasha (Oklahoma), and Lockyersleigh (Australia). Kachanoski and De Jong (1988) argued that spatial scales must be considered in this type of analysis because of the correlation length scales with a soil moisture field.

These previous projects were conducted over scales (< 27 km^2) smaller than most satellite remote sensing technologies (100 - 2500 km^2). The scale of temporal stability must be established at larger scales (Kachanoski and De Jong 1988), if this approach is to be used in the validation of large scale remote sensing products. Also, there is a need to extend this research to a larger variety of surface types such as agricultural crops.

The study reported here estimates watershed scale (~100 km^2) soil moisture averages for the purpose of validating current remote sensing products by means of point to watershed scaling of in-situ soil moisture sensors. Using three methods of statistical exploration, namely mean relative difference analysis, Spearman rank coefficients and correlation analysis, the temporal and spatial stability of soil moisture for a region can be assessed. For a given season, representative locations can be identified for future regional estimation, greatly reducing the complexity and operational costs of watershed and regional scale monitoring. This work focuses on a temporary sensor network that was installed during the Soil Moisture Experiment 2002 (SMEX02). This network was in place for two months during the summer of 2002 and serves as a model for future watershed investigations.

In this investigation, we explore the potential of temporal stability theory as a solution to the problem of satellite based soil moisture validation. This may provide a means to effectively design sparse validation networks and may also provide a way to utilize existing in-situ low density networks in validation. In addition, this project will investigate the intricacies of using only a few in-situ points for large scale validation.

Study Region

The intensive study region of SMEX02 was the Walnut Creek watershed and the surrounding area, located south of Ames, Iowa, which is on the order of 100 km^2. An outline of the watershed is shown in Figure 1. Corn and soybean dominate the land cover, with approximately 50% and 40% respectively. The remaining 10% of the area's land cover is grains and urbanization. The intensive field campaign portion of SMEX02 took place from June 25th to July 12th, 2002. As part of that experiment, 12 Stevens-Vitel Hydra (www.stevenswater.com) probes were installed in 10 study fields near surface meteorological stations, which were located throughout the area as part of the experiment. These stations operated during the field campaign and continued until August 19th, 2002. This extended period of time allowed for a wider range of soil moisture patterns to be observed. This study will demonstrate how SMEX02 contributes to the field of temporal stability.

The soil moisture probes measured the dielectric constant of the soil, from this the volumetric soil moisture was computed from previously determined relationships (Campbell 1990). Each probe was installed at a depth of 5 cm, which is appropriate for comparing soil moisture

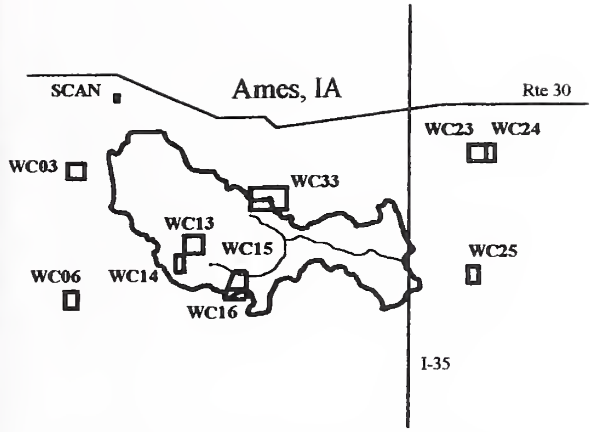

Figure 1. The Walnut Creek Watershed and the surrounding region near Ames, Iowa. The ten WC Fields and the NRCS SCAN station are outlined.

readings to L-band microwave remote sensing estimates (Jackson 1993). The land cover distribution of the sampled fields is as follows: Soybean-WC03, WC13, WC14, WC16, WC13; Corn-WC06, WC15, WC24, WC25, WC33; Grass-SCAN.

In addition to this temporary soil moisture sensor network, there is also a permanent soil moisture profiling station situated northwest of the watershed as part of the Natural Resources Conservation Service-Soil Climate Analysis Network (NRCS-SCAN) (Schaefer and Paetzold 2001). This SCAN site records a suite of meteorological and hydrological variables, including precipitation, soil temperature, and soil moisture. Though the location of this particular site is covered in grass in a low swale within the field which causes abnormally high soil moisture readings occasionally. The site was used in this study to evaluate its potential as a future tool for estimating the soil moisture in this region.

Methods

Current approaches to the estimation of watershed scale surface soil moisture requires a dense network of moisture probes located throughout the region to provide a large number of samples. The most efficient way of reducing this burden is to find a way to predict large scale moisture averages from only a few sensors located at 'representative' sites. These sites can be identified through temporal stability analysis. If temporal stability can be established in a watershed, a small number of soil moisture sensor sites can be used to accurately and precisely predict watershed averages. This is accomplished by determining those sites that maintain a consistent temporal relationship with the watershed average with little variability.

The primary method for determining the temporal stability of a soil moisture field is the mean relative difference plot. This plot represetns the ability of a particular soil moisture sensor location to estimate the average over the watershed. Building on Vachaud et al. (1985) and Grayson and Western (1998), this type of analysis was applied to the SMEX02 watershed network. The mean relative difference is defined as

$$\bar{\delta}_{\bullet,j} = \frac{1}{n}\sum_{i=1}^{n}\frac{S_{i,j}-\bar{S}_{i,\bullet}}{\bar{S}_{i,\bullet}} \tag{1}$$

where $S_{i,j}$ is the i^{th} sample of n samples at the j^{th} site within the study region. $\bar{S}_{i,\xi}$ is the computed average among all sites for a given date and time, i. This variable gives a direct measure of how a particular site compares to the average of a larger region, whether it is consistently greater or less than the mean and how variable is that relationship. The mean relative difference of each site is then plotted by rank with error bounds of one standard deviation of the relative differences to determine which site best estimates the mean of the watershed. There are two criteria for selecting the ideal site for watershed estimation. Proximity of a site's mean relative difference to zero indicates it can accurately estimate the watershed average and small standard deviations (narrow error bars) indicate low variance of that estimate. If a site has both of these characteristics, it can be concluded that it accurately and precisely predicts the average watershed soil moisture for long time periods.

It is also important to assess the spatial stability of the soil moisture field which can be accomplished with the Spearman rank coefficient. This coefficient measures the correlation of site rankings from one day to the next. It is defined by

$$r_i = 1 - \frac{6\times\sum_{i=1}^{n}\left(R_{i,j}-R_{i,j'}\right)^2}{n\left(n^2-1\right)} \tag{2}$$

where $R_{i,j}$ is the rank of the soil moisture, $S_{i,j}$, at location i on day j, with a total of n days. $R_{i,j'}$ is the rank of the same location i for day j'. A value for r_i near 1 indicates a stable soil moisture field, while r_i values near zero indicate a lack of stability. Therefore, an r_i of 1 is computed for pairs of days that maintain the same ranking among the soil moisture gaging sites.

When dealing with an in-situ network, it is necessary to address the temporal resolution. For these purposes, it is only necessary to consider soil moisture from one day to the next. Therefore in this analysis, the Spearman rank coefficient is calculated between each hour of each day (to account for any diurnal pattern in the signal) and then these are averaged together to obtain a single coefficient for each day.

Results

The first step in the analyses is an examination of the time series of surface soil moisture measurements for the Walnut Creek watershed, as shown in Figure 2. This plot shows the individual site and average soil moisture, as a function of time. One can readily observe the variation that exists among the individual points on any specific day. Applying Eq. (1) to the data set resulted in a mean relative difference plot, shown in Figure 3.

Several key results can be drawn from this plot. WC13, a soy field in the center of the watershed, had a mean relative difference close to zero and a small standard deviation, indicating a close correlation between the WC13 soil moisture at 5 cm and the expected average of surface soil moisture across the entire watershed region.

Patterns are visible in Figure 3 when the location of each site is considered. WC23, WC24, and WC25 are all located in the eastern portion of the study region and, from observations made during the experiment, had smaller precipitation amounts. This is determined from the negative mean relative differences for these sites. Negative mean relative differences indicate that the average at that particular site is less than the average across the whole region.

Also, there was a precipitation event on Day 185 which was very heterogeneous across the watershed; therefore, each site received a different amount of rainfall. This resulted in moisture patterns, which would be different from a large scale precipitation event, thereby nullifying any temporal stability. This issue proves to be a problem for watershed scale estimation for particular time periods. Precipitation events can be divided into two scales: Field scale and watershed scale. It is expected that larger events will dominate the moisture field of a watershed at long time scales, but for any small time period, there could be an influence of heterogeneous precipitation

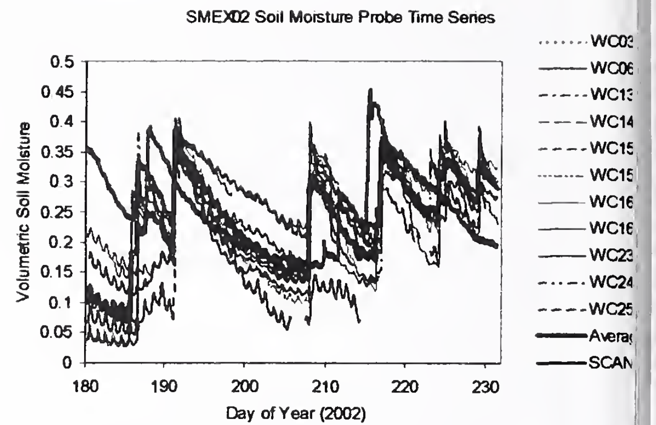

Figure 2. Time series of surface (5 cm) soil moisture for each soil moisture probe in and around the Walnut Creek watershed. The average for each time step is also plotted in bold.

occurring at the smaller field scale. Therefore, using singular point estimates to approximate watershed scale soil moisture should only be considered for long-term validation. Also, it would prove to be unwise to use a single 'random' point to estimate regional soil moisture in the short term. For instance, the SCAN site demonstrates a significant bias (nearly 20%) to the regional soil moisture average. However, there is still potential to use the SCAN site as a rough approximation if this bias can be taken into account.

It is also apparent that there was little or no deterministic relationship between mean relative difference and crop type. Soybean and corn fields are scattered across the mean relative difference plot, indicating that the location within the watershed may play a greater role in the selection of a representative site than does land cover
type.

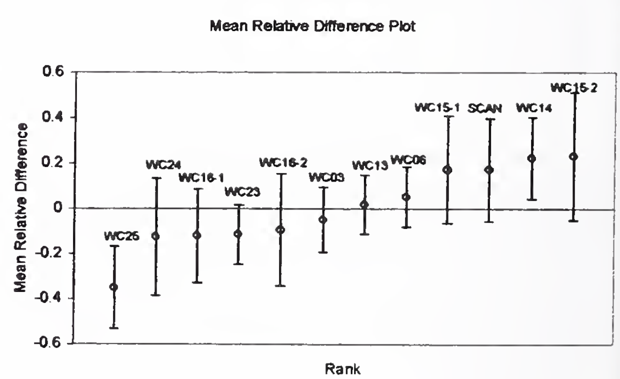

Figure 3. The mean relative difference plot for the SMEX02 soil moisture network. The bars are +/- one standard deviation.

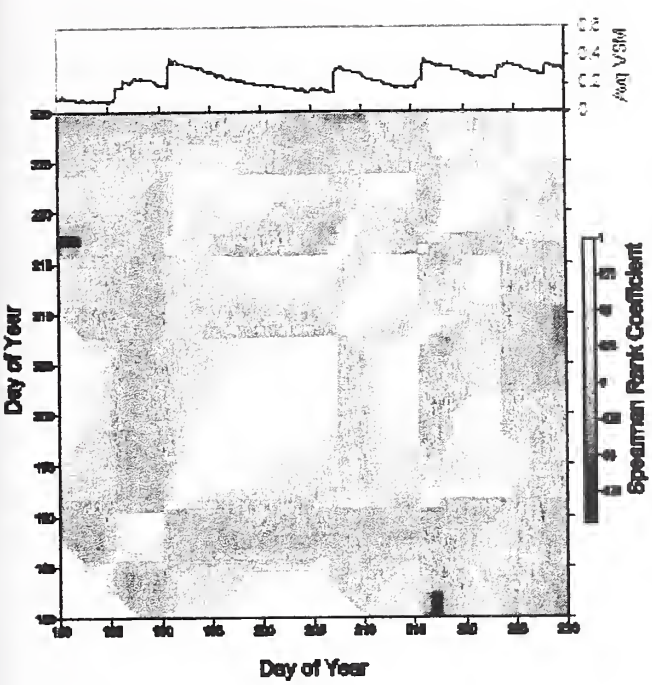

Figure 4. Spearman rank coefficient plot of volumetric soil moisture by day of year. Also included is a plot of the average soil moisture for the watershed for the same time period. Coefficients near 1 indicate strong rank correlation between the dates.

A Spearman rank analysis determined that for most of the study period there is a strong temporal stability across the region. Figure 4 shows a plot of these coefficients over time as well as a plot of the average soil moisture for the watershed. The plot is grayscale; therefore, the whiter the plot, the higher the Spearman rank coefficient. Dark pixels indicate low values and time instability. For several time periods, there is a distinct lack of stability, such as for the days proceeding days 185, 208, and 223. Each of these periods follows a heterogeneous precipitation event, as shown by the drastic changes from high to low Spearman rank coefficients. Conversely, on day 191, there was a larger than watershed scale rain event, which affected each of the sites uniformly. Following this event, an order is observed in the ranking of the magnitudes of the soil moisture at each gaging station, similar to that of the mean relative difference plot. Overall, the plot indicates that there is a persistent pattern to the watershed moisture condition such that for a given homogeneous precipitation event, there is a ranking among the surface soil moisture measurement sites. This temporal stability should prove useful for the prediction of watershed scale soil moisture with a sparse array of in-situ soil moisture measurements.

Site selection was examined in greater detail to try and identify characteristics that make particular sites representative of the watershed. Initial considerations would reveal that closeness to the center of the region of study is not a necessity, because both WC03 and WC06 have low mean relative differences and are the western most sites. However, if a site is close to the center, it is more than likely receiving the mean precipitation for the region for long periods of time. Land cover type did not appear to be a significant factor because there was no apparent link between soybean, corn, and mean relative difference rank. There is a complex set of variables which appear to affect mean relative difference.

Further investigation into the sensor at WC13 revealed that if only one site was available for estimating average watershed soil moisture, this sensor would be a credible choice. A random sampling of points between the sensor at WC13 and the watershed average had a strong correlation ($R^2 = 0.928$) and low root mean square error (rmse = 0.028). The bias was also quite small at 0.006 (m^3/m^3). WC13 was a typical row-crop soybean field with some topography, while WC14, for example, was a drilled or broadcast soybean field with similar topography with a similar precipitation history. The only apparent distinction between these fields was the method of planting, but there is a considerable deviation in their mean relative differences.

Conclusions

Watershed and regional estimates of surface soil moisture are necessary for a wide variety of hydrologic and climatologic studies; however, it is infeasible to gage a system adequately for true measures. Remote sensing provides an attractive alternative. However, these methods must be calibrated and validated. This work demonstrates that single point in-situ measurements can be used to estimate area average values accurately if spatial and temporal stability can be established in the region of interest. It has been shown that for the Walnut Creek watershed the soil moisture pattern during the summer of 2002 was both temporally and spatially stable for uniform precipitation events. A mean relative difference plot established that with accuracy and precision, a single site (WC13) could accurately and precisely estimate the watershed soil moisture average for long time periods. For time periods that are subject to heterogeneous rain patterns, this stability is reduced. Several points may be necessary to accurately characterize the soil moisture for specific time periods. Certainly, the use of one random in-situ point would be a risky proposition. For example, if the SCAN site was used as a representative point, there would be a significant amount of bias. Fortunately, experiments such as SMEX02 permit the SCAN to be calibrated to the watershed average for long term studies. It is demonstrated that short term field experiments may be an appropriate method for establishing temporal stability and calibrating in-situ field sensors.

For the purpose of validation of remote sensing of surface soil moisture products, the temporal scales are greater than the short episodes of heterogeneous precipitation often experienced in field experiments. Indeed, the time scales of validation span many seasons and a watershed's soil moisture distribution at this time scale is, on average, a result of large-scale weather systems. It can be concluded that for the purposes of validation, temporal stability is a valuable tool for accurate and precise estimation of mean soil moisture.

Acknowledgments

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Monitoring Rangeland Watersheds With Very-Large Scale Aerial Imagery

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Abstract

Motion-blur free, very-large scale (1:200), aerial-photographic samples of grazing allotments on high-elevation Wyoming (USA) rangelands were systematically acquired (n = 172) using an ultra-light type, fixed wing airplane and a modified Hulcher 70 mm camera with Kodak Aerocolor HS SO-846 film. Cover measurements from the digitized aerial samples were not different from cover measurements made on the ground using point-sampling methods.

Key Words: bare ground, motion blur, sampling adequacy

Introduction

Rangeland watershed management has depended more on judgement than science for monitoring the condition or health of vast landscapes. The result is a crisis of confidence in traditional monitoring methods and data, and an understanding that we need objective monitoring (NRC 1994, Donahue 1999). The challenge is to develop economical methods that will detect important vegetation change within acceptable error rates (Brady et al. 1995). Some have promoted a suite of key indicators for assessing rangeland health (USDI 1973, Pellant et al. 2000, Rowe et al. 2002). We recognize the complex, multivariate nature of rangeland ecosystems; we also recognize that information-collection costs require that we define key indicators -- those that lend themselves to detection of ecologically important change with minimal expense. Bare ground is usually listed as a key indicator (USDI 1973, Abel and Stocking 1987, WRAC 1997, Pellant et al. 2000). Here we report progress on our effort to use very-large scale aerial (VLSA) imagery as a means for inexpensive acquisition of statistically adequate, unbiased, high-resolution (high detail), samples (images) from which to accurately measure bare ground. The main technical constraint in acquiring VLSA imagery from a moving, low-altitude platform has been motion blur in the imagery (Hinckley and Walker 1993).

Platforms for acquiring VLSA imagery have included camera stands (Bennett, Judd and Adams 2000), poles, balloons, dirigibles, kites, radio and computer-controlled unmanned aircraft, ultralight aircraft, and helicopters (Tueller et al. 1988, Hinckley and Walker 1993, Hansen and Ostler 2002, Aerosonde 2002, and personal communication). Helicopters and long-range unmanned aircraft (Aerosonde) are high cost. The other platforms are impractical for extensive monitoring (100 to 200 km^2). Therefore we tested an ultralight-type, 3-axis, fixed-wing airplane as an inexpensive platform for obtaining motion-blur free, VLSA imagery over extensive areas of high-elevation rangelands in Wyoming's Red Desert.

Materials and Methods

Nadir aerial images (1:200 scale calculated as negative length over ground distance) over two public-land grazing allotments in south-central Wyoming were made with a modified Hulcher Model 123, 70-mm camera equipped with a 500-mm lens (Charles Hulcher Co., Hampton, VA, USA) and mounted in a Rans

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S12XL, 2-seat airplane. The airplane was flown at 72 km/hr ground speed (straight and level flight), 100 m above 1520-m-elevation rangelands and we used 1/4000 second shutter speed on the camera. Altitude above ground level (AGL) was continuously monitored and displayed to the pilot with a laser altimeter and the camera was automatically triggered for systematic, intermittent, aerial sampling (Booth 1974, Abel and Stocking 1987) by a Track'Air aerial survey system using pre-programmed coordinates (Track'Air, Hengelo, The Netherlands). Our Track'Air system was specifically adapted to our application. At take off the system defaults to ferry mode and directs the pilot to the target area. When the plane is within 300 m of a target-area flight line the system "locks on" to that flight line and directs the pilot to the first target while providing constantly updated information on ground speed, and time and distance to the first target. When on target, the system triggers the camera, records the geographic positioning system (GPS) coordinates of the actual trigger location, and advises the pilot that the camera was triggered. The system then directs the pilot to the next target. This is likely the only aerial survey system in the world designed to operate with an airplane crew of 1 (pilot). Using this system the pilot's hands need never leave his controls, nor his eyes leave the forward view -- critical safety considerations when flying slow and close to the ground.

To program the Track'Air system, a digitized raster graphic of the target area was downloaded and Didger II (Golden Software, Golden, CO, USA) used to extract GPS coordinates in a 0.8 km grid over the two grazing allotments. The coordinates were then used to create a TrackAir flight plan for the mission. (The system also accommodates irregular targets as may occur in sampling riparian or other critical areas.)

Twenty, 1-m^2-plots were located on the ground using GPS coordinates where the Hulcher was triggered and where ground access was not problematic. Images of these plots were acquired using an Olympus E20, 5.0 megapixel, color digital camera mounted 2 m AGL on a portable camera stand having a m^2 base. An infrared remote was used to trigger the shutter of the fully automatic camera. Images were saved as uncompressed Tif files at maximum resolution (1 pixel = 0.5 mm ground area, scale = 1:110 calculated as CCD length over ground distance). Additionally, cover and bare ground were measured on these plots using standard point-sampling methods (100 points per m^2). The base of the camera frame was the reference used for the point-sampling data collection.

Film from the Hulcher camera was pushed 1 f stop in development, and scanned at 1 pixel per 25μm of negative. Bare ground was measured from the Hulcher and Olympus images using manual methods (digital grid overlay using 100 points) and Vegmeasurement software (Louhaichi and Johnson 2001). The digital-grid-overlay method is simply using software to overlay a grid on the image, then recording the type of ground cover underneath each intersection of the grid. Vegmeasurement software was developed at Oregon State University and uses an algorithmic manipulation of color hues to separate image characteristics like bare ground and plant cover. Data from these measurement methods were compared with each other and with measurements from on-the-ground point-sampling.

Results

We obtained motion-blur-free, 1:200 scale images by flying at 72 km/hr ground speed, 100 meters AGL and using a 500 mm lens with Kodak SO-846 film and 1/4000 second shutter speed (Fig.1). Light during the monitoring effort ranged between 8 and 10 thousand lx.

Figure 1. Grayscale rendition of an aerial sample (1:200) from study area with a m^2-enlarged portion demonstrating the resolution possible with this photograph.

Safe, systematic aerial sampling --consistent with sampling needs of watersheds and other large land areas -- was immeasurably facilitated by our custom-configured Track'Air aerial survey system and by precise measurements of airplane altitude AGL from our laser altimeter. We found no difference in bare ground measurements between aerial and ground methods, implying that for the plant communities monitored in this test, bare ground measurements made from the Hulcher imagery were as accurate as measurements made on the ground (Fig. 2). There was a significant difference between measurements made with the digital grid (100 points) and Vegmeasurement. We judge the Vegmeasurement data to be more accurate since the software uses approximately 2 million image pixels as data points.

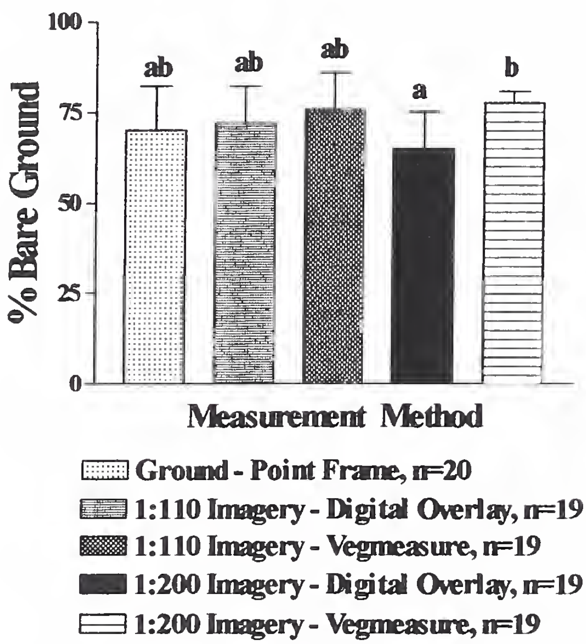

Figure 2. Bare ground measurements for the study area derived from five measurement methods. Measurements from 1:200 images were different from each other (P = 0.0013) but neither one was different from point sampling nor 1:110 image-derived measurements.

Ellison and Croft (1944) observed, "There are two levels of observation in range inspection: one extensive and the other intensive. From observations at the extensive level the inspector can get an idea of only general over-all features of the range, i.e., the extent and character of vegetal types, topographic features, ... *Intensive observations on small areas are necessary to secure the detailed facts from which the only valid conclusions of range condition can be made.*" (emphasis added). Remote sensing is the only way to obtain accurate information over extensive areas at reasonable cost (West 1999), but until now the measurement of details from images acquired from a continuously moving platform has largely been limited by motion blur to scales of about 1:600 or larger. We conclude that our methods merit further research as a means for monitoring extensive areas by obtaining a statistically adequate number of aerial samples from which to make detailed measurements of bare ground.

Note

Disclaimer: Mention of trade names is for information only and does not imply an endorsement by USDA or USDI.

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Hyperspectral Remote Sensing of Water Quality Parameters for Large Rivers in the Ohio River Basin

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Abstract

Optical indicators of water quality have the potential of enhancing the abilities of resource managers to monitor water bodies in a timely and cost-effective manner. However, the degree to which optical indicators are useful may depend on their applicability to data collected from multiple water bodies. In 1999, a Compact Airborne Spectrographic Imager (CASI) was flown over the relatively shallow Great Miami River (GMR), in Southwest Ohio, collecting hyperspectral bands of data. Concurrently, water quality samples and hand-held spectrometer data were collected directly from the river. Using correlations between the ground-truth data and combinations of spectral bands from the remotely sensed data, spectral indices were developed which could be used to estimate chlorophyll *a*, turbidity and phosphorus. In 2001, a similar study was conducted in which a CASI was flown over a portion of the Ohio River while ground-truth data were collected. These data were analyzed and tested against the spectral indices developed during the 1999 study. The GMR's spectral index for chlorophyll *a* was applicable to the Ohio River data. However, slightly refined spectral indices for turbidity and phosphorus were required in this new environmental setting.

This study demonstrates the ubiquitous application of the chlorophyll *a* spectral index while revealing the limited reliability of the turbidity and phosphorous spectral indices. Although differences between the dynamics of the two rivers may have made these spectral indices incompatible, with further refinement they may yet prove to be useful tools that can be modified for use in other rivers to detect potential water quality problems.

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Keywords: remote sensing, water quality, chlorophyll *a*, nutrients

Introduction

Eutrophication diminishes water quality by promoting the excessive growth of algae, and increasing suspended organic material. When degraded, unpleasant odors and tastes can result from the excessive amounts of algae. Furthermore, microorganisms associated with eutrophication may pose health risks to consumers. It is important for water resource managers to find the most efficient way to diagnose the condition of drinking water sources. This may be especially demanding when assessing water quality damage in large rivers when field measurements may be time consuming, costly, and limited logistically. Increases in water quality parameters such as chlorophyll *a*, turbidity, total suspended solids (TSS), and nutrients are symptomatic of eutrophic conditions. Concentrations of these parameters can provide insight on the extent of eutrophication and the potential impact on aquatic biota and overall water quality. It would be advantageous to resource manager to be able to detect eutrophic conditions using multiple sites in a river without relying on field measurements.

Environmental researchers have been making efforts to monitor, simulate and control eutrophication for more than two decades. Various mathematical models have been developed and applied to rivers, lakes and estuaries (Lung 1986, Thomann and Mueller 1987, Kuo and Wu 1991, Kuo et al. 1994). All water quality models simulate increases in eutrophication based on the initial condition of the water body and therefore demand comprehensive water quality sampling programs. However, the conventional measurement of water quality requires *in situ* sampling and expensive and time-consuming laboratory work. Due to these limitations, the sampling effort often does not represent the condition of an entire water body. Therefore, the difficulty of overall and successive water quality sampling becomes a barrier to water quality monitoring and forecasting.

Remote sensing could overcome these constraints by providing an alternative means of water quality monitoring over a greater range of temporal and spatial scales (Shafique et al. 2001). Remote sensing is the science of measuring the properties of objects by measuring the amount of radiation they absorb, emit, or reflect at various wavelengths along the electromagnetic spectrum. Optical water quality research has a broad scope for developing environmental indicators that are useful in assessing, quantifying and monitoring instream water quality. Measurable parameters for optical water quality includes the attenuation coefficient (K_D) of photosynthetic active radiation (PAR), turbidity, concentrations of algal chlorophyll, suspended sediment, and dissolved organic matter. More fundamentally, the absorption and scattering of light by components of a stream's water column provide basic information from which relationships with other water quality indicators (such as water clarity from Secchi disk readings) can be derived (Jupp et al. 1994a, Dekker 1997). Although a fairly new method, the development of spectral indices can be a useful and easy tool for the diagnosis of eutrophic conditions by water resource managers.

Remote sensing techniques for monitoring coastal and inland waters have been under development since the early 1980's. The tools used to develop these techniques have ranged from an empirically-based method for producing qualitative water quality maps to semi-empirical techniques and analytical methods for producing quantitative water quality maps (Dekker 1997). Several investigators (e.g., Dekker 1993, Gitelson, et al. 1993, Jupp et al. 1994a, Jupp et al. 1994b) have developed empirical regression formulas for the prediction of lake water quality parameters from spectrometer data by employing spectral ratios, typically reflectance ratios, as the independent variables. The predicted water quality parameters have included chlorophyll *a* concentrations, suspended matter concentrations and turbidity.

This investigation also focuses on the prediction of chlorophyll *a* concentrations, turbidity and total phosphorus concentrations by applying spectral indices developed from spectral data collected by spectro-radiometers as independent variables. However, this study is unique in that it has developed a regression formula for lotic systems, specifically using large rivers as a model. Spectral indices are transformations of reflectance values at specific wavelengths that minimally correspond to a field tested concentration of the parameter of interest and minimize the effects of other optically active constituents. The method used made correlations using simultaneous collected remote data, field spectrometer data, and field collected water quality data to demonstrate the feasibility of remote sensing techniques for water quality monitoring in large rivers. This paper also discusses the application of the optical properties of water for multiple river systems. This includes the determination of the optical properties of some water quality parameters and the development of spectral indices using hyperspectral airborne data for the shallower Great Miami River (GMR), and the transfer of these spectral indices to the larger and deeper Ohio River.

Materials and Methodology

The investigation utilized compact airborne spectrographic imager (CASI) data from approximately 60 river miles of the GMR (Shafique et al. 2001), and 80 river miles of the Ohio River. Based on the analyses of preliminary field spectrometer data, 19 appropriate spectral bands with 5-nm spectral resolution were selected and programmed into the CASI unit. Data were collected during four field efforts, carried out under similar conditions, during 1999 and 2001. While the hyperspectral data were being collected by the airborne CASI, *in situ* water samples were collected and a field spectrometer was used to collect spectral data directly from the river.

The field, laboratory and remotely sensed data were analyzed in a systematic manner. First, the spectral library database was developed and used to establish the variability and/or stability of absorption and scattering coefficients in the GMR (Shafique et al. 2001) and the Ohio River. Single spectral bands, ratios of spectral bands, and combinations of multiple bands were then used to develop linear regression equations. The semi-empirical models were developed in Excel (Microsoft Software, v. 2000) spreadsheets, and the imagery was analyzed using the ENVI (3.6) image processing software. First, in order to represent a homogeneous unit in the imagery, the water area was masked. The unsupervised image classification technique, K Mean, was used to cluster imagery into spectrally similar categories. This classification technique was used over another method, supervised classification, because the identifications made by the latter are made on the basis of human sight, which is limited to visible wavelength range (Vincent 1997). Scatter plots were created between spectrally classified image and ground-truthing data, based on their linear trends; simple linear regressions were used to determine the relationships between single and combinations of bands and water quality parameters. Based on the image pixels of the locations from which ground-truth data were collected, equations were developed for particular water quality parameters. Then, the entire image was converted into a water quality map using the predictive equations. Some of the ground-truth data were not used to develop the equations, but were instead used to validate the predictive quality of the equations. Using this semi-empirical approach, separate equations were developed for each water quality parameter. This approach was the primary means used to analyze the images collected for this study.

The analytical approach of spectral image analysis used the spectral library that was developed from the 1999 GMR study and the 2001 Ohio River study (Shafique et al. 2001). The reflectance/radiative transfer model used in that analytical approach quantifies and simulates the individual contributions of the water constituents to the reflectance measured by the remote sensors. The development of a reflectance/radiative transfer model depends on how well the specific absorption and scattering coefficients are determined for the various constituents. Once stable specific coefficients were established, the reflectance/radiative transfer model could be used to mathematically convert airborne imagery into water quality maps with limited use of ground-truth data. The use of the ground-truth data could be limited because a physical understanding of the interactions between the various water constituents and water reflectance was incorporated into the radiative transfer equations. The success of the analytical approach depends on the successful optical characterization of the water body and potential contributing sources such as industrial wastewater discharges. Once such a characterization is made, available optical water quality toolkits can be adapted to a particular study region. One of these prototype toolkits has been developed by Dekker (1997).

Models

All bands were tested for relationships with water quality parameter until it was found which bands and parameters correlated with the highest certainty. Scatter plots showed that linear models using the ratio of wavelengths 705/675 nm and the logarithmic ratio of wavelengths 554/675 nm can describe chlorophyll *a* and total phosphorus, respectively. Logarithmic transformation is useful in cases, such as this, where it is necessary to stress the difference between scores in a manner that is proportional to their ratio rather than in terms of their absolute difference. The band readings that represented the difference of 740 nm from 710 nm correlated best to turbidity. The r-values and R^2 for each of these are above 0.7 and 0.5 respectively, therefore, indicating the ability to provide good linear models for these water quality parameters (Figure 1). Based on the linear relationship with water quality parameters, the spectral indices were then transferred to the following mathematical models to calculate the concentrations of the respective water quality parameters.

$$\text{Chlorophyll } a = 48.849 * (705/675 \text{ nm}) - 34.876 \quad (1)$$

$$\text{TP} = 0.1081 * \log(554/675 \text{ nm}) - 0.0371 \quad (2)$$

$$\text{Turbidity} = 186.59 * (710 - 740 \text{ nm}) + 8.5516 \quad (3)$$

Results and Discussion

The results of this research can be divided into three general sections. The first section describes the preliminary data that were collected using the field spectrometer and the results of the water laboratory analyses. This data was used to explore the feasibility of using hyperspectral data for the identification and discrimination of in-stream features via the GMR and Ohio River. The second section describes the development of spectral indices using water constituents. It includes the A preliminary radiance/reflectance transfer model is presented that can be used to interpret remote sensing imagery in the same environment where the specific absorption and scattering coefficients are known. The third section addresses the correlations made between the remotely sensed imagery and the consequent estimations made for water quality parameters using the field spectrometer data. Through atmospheric and water column correction, the instrument calibrates the remotely-sensed data to the conditions that were present at the water's surface at the time of CASI data acquisition. Both semi-empirical and analytical models were applied to mathematically convert the hyperspectral imagery into water quality maps.

Relationships found using water quality and field spectrometer data

The correlation between various water quality parameters was calculated using a Pearson's correlation test. The most significant relationship observed was between the concentration of chlorophyll *a* plus pheophytin and the concentration of dissolved oxygen (DO). As is often observed in eutrophic systems, DO is negatively correlated ($r = -0.81$) with chlorophyll *a* plus pheophytin. Overall, dissolved oxygen had a negative correlation with other measured parameters.

There was a weak relationship between chlorophyll *a* concentration and water depth, and there was no significant correlation between reflectance at any wavelength and water depth. The water samples for turbidity data and the water samples for the chlorophyll *a* analysis were collected in close spatial and temporal proximity. Although it is assumed that the composition of the water did not vary significantly, it is recognized that an unaccounted error may be introduced when trying to establish relationships for these parameters.

Development of spectral indices

Spectral indices are simple arithmetic expressions of a combination of spectral bands that help reduce or eliminate some differences in viewing geometry and atmospheric conditions between measurements. Due to the large number of bands measured with the field spectrometer data, a sub-set of bands at 5 nm intervals was selected to develop the spectral indices. The bands that carry the most information about water quality parameters were selected by qualitatively analyzing field spectral plots of actual measurements. Generally, bands that show peaks and troughs (i.e., where the reflectance spectral curve changes slope) were selected. Correlation values were calculated between ground-truth spectral data, water quality data from laboratory analyses, and the spectral indices developed from exploratory single bands, ratios of bands, differences between bands, and/or combinations of differences and ratios. The values from the indices with the highest correlation values with the ground-truth and laboratory data were used to produce scatter plots and calculate the R^2 values.

Once the bands were selected, three types of indices were developed. These were the difference, ratio, and combination of ratio and difference indices (Figure 1) originally developed from the GMR (Shafique et al. 2001). Table 1 shows spectral indices for the GMR and Ohio, River. It is noteworthy that the same index for chlorophyll *a* (i.e., 705/675 nm) worked equally well for both rivers. However, the other two indices (i.e. turbidity and total phosphorus) required slight modifications in the spectral band selections. Likely the modification was needed due to differences of suspended sediments and their effects on reflectance in the two rivers.

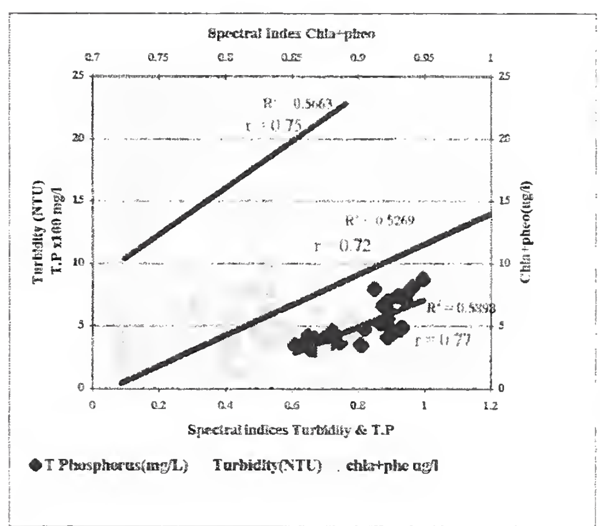

Figure 10. Correlation between water quality parameters and spectral indices. Total phosphorus values were multiplied by 100 to fit in the scale.

Table 4. Correlation coefficients for water quality parameters (WQP) and spectral indices (SI) for the two rivers.

WQP	SI	GMR	Ohio River
Turbidity	675-700	0.79	0.20
Turbidity	710-740	-0.33	0.75
Chlorophyll-*a*	705/675	0.71	0.72
T.P	(554/740)-(620-740)	0.66	0.34
T.P	log(554/675)	0.29	0.77

Correlations between water quality parameters and spectral data

Generally, the correlations between the ground-truth data and spectral parameters were stronger with the ratio indices and weaker with the individual bands. The difference and ratio indices seem to identify with a particular group of ground-truth data. For example, the difference index performed better for parameters such as turbidity, TSS and secchi depth, while the ratio index correlated better with chlorophyll *a* and pheophytin parameters. These differences are likely attributed to the reduction of turbulent effects in the water body when the ratio index was used, while the difference index reduces some of the atmospheric effects.

Spectral bands with wavelengths at 670, 675, 700, 705 and 740 nm appear to dominate the spectral indices. Of these, difference indices involving 675, 700 and 740-nm wavelengths provide information about turbidity while the ratio and combination indices using 672, 675, 700 and 705-nm wavelengths provide information about algal parameters. Taking into account all of the chlorophyll parameters, the concentration of chlorophyll *a* plus pheophytin correlated best with the same spectral parameters for the GMR and Ohio River. This can be explained by chlorophyll *a* and pheophytin tending to reflect and absorb light energy similarly. Logarithmic ratios of spectral bands in the green and red to near infrared region of the spectrum (i.e. 554 and 675-nm wavelengths) showed good correlation for TP.

Using the relationships between chlorophyll *a* and a ratio index (i.e., 705/675-nm wavelengths) from the CASI image, the ratio's 16 (K-means) spectral classes, converted into chlorophyll *a* concentration levels, ranged from 1.64 µg/l to 38.5 µg/l (Figure 2). The broad chlorophyll *a* levels are indicated on the atmospherically and radiometrically corrected CASI images acquired in September and November 2001 (Figure 2). The overlay of these classes on a true color image revealed plumes of higher chlorophyll *a* concentrations at some of the confluences of tributaries with the Ohio River. For example, the relatively high concentration of chlorophyll *a* in the Licking River can be seen at the confluences with the Ohio River. Similarly, classes of turbidity and TP were overlaid on the CASI image (Figure 2).

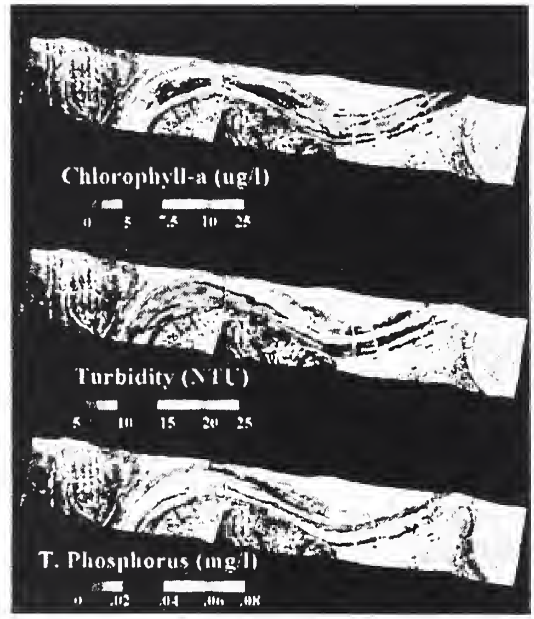

Figure 2. Water quality parameters map of Ohio River at the confluence of Licking River near Cincinnati, OH. Higher chlorophyll *a* and turbidity levels are found at the confluence. R^2 values of scatter plots of observed and estimated parameters are above 0.9 indicating high accuracy.

Conclusions

This study demonstrates that the hyperspectral remote sensing technique can be a useful tool for monitoring the distributions of chlorophyll *a* concentrations in large rivers. In this study, the wavelengths of 675 nm and 705 nm from the CASI data were found to be the most suitable wavelengths for predicting chlorophyll *a* concentrations. Correlation analysis between remotely sensed data and chlorophyll *a* data has indicated the possibility of mapping chlorophyll *a* concentrations accurately. The strong correlations of reflectance ratios corresponding to these wavelengths with field spectrometer data were used in the development of equations and constants for the estimated chlorophyll *a* concentrations in the Markland Pool.

The results show that it is also feasible to estimate the relative chlorophyll *a* levels of large rivers when ground-truth data is not routinely available. This is essential for operational applications in large rivers where the total number of *in situ* water quality observations only cover a small fraction of the river for a limited time. Moreover, the results indicate that chlorophyll *a* has a unique spectral signature and it is possible to estimate chlorophyll *a* concentrations for any inland water body with the chlorophyll *a* spectral index.

The methods developed and analyzed in this paper used CASI, but it is predicted that the same information can be revealed using hyperspectral data acquired by a satellite such as the Hyperion satellite. Although the spatial resolution of the data collected by the Hyperion satellite is only 30 m, in contrast to the 2-m resolution of the CASI data, the use of the satellite may be more cost effective and as reliable because the Hyperion spectrometer includes channels with the same wavelength bands employed here for chlorophyll *a* retrieval. However, it is still necessary to compare the results of data collected from *in situ*, CASI, and Hyperion studies to investigate the reliability of the data. Future research will provide information about whether satellite data can be substituted for field-collected data to determine water quality parameters such as chlorophyll *a*, nutrients, and turbidity. In the future, Hyperion remote sensing data may prove to be the preferable method for the detection of eutrophic water quality indicators over large areas of water.

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A GIS-based Management Tool to Quantify Riparian Vegetation Groundwater Use

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Abstract

Rapid population growth in semiarid regions of the southwestern United States is increasing the demand for water. In many cases, groundwater is mined from valley aquifers to meet this demand, which results in declining water levels in the aquifers. Riparian corridors are vulnerable to these declines since near-surface groundwater supports baseflow in the rivers and the abundant vegetation/habitat found therein. This is the case for the San Pedro River Basin in southeastern Arizona and northern Mexico. In such basins, effective management of water resources requires accurate measurements of water fluxes, including the evapotranspiration from the vegetation in the riparian corridor. This paper describes a management tool to help estimate groundwater demand from riparian vegetation along the San Pedro. The tool combines calibrated, process-based ecosystem models of riparian water use with a vegetation map to provide watershed-scale estimates of riparian vegetation groundwater use. This model is GIS-based to provide a user-friendly application that allows the user to change the vegetation cover in order to evaluate the effects of vegetation change (e.g., prescribed or accidental burns, rehabilitation of abandoned agricultural fields, shrub removal, etc.) on the groundwater demand.

Keywords: evapotranspiration, riparian vegetation water use, consumptive use

Introduction

Humans living in dryland regions increasingly rely on regional aquifers as a source of fresh water due to the limited availability of surface water sources and population increases. Without this groundwater resource, the further development and perhaps even the sustainability of these communities would not be possible. Similarly, the vegetation and enhanced biological productivity of oasis-like riparian areas in these regions are dependent upon the same groundwater source. Riparian regions are now recognized as biological "hotspots" and are extremely important for providing habitat for wildlife.

Groundwater pumping affects the dynamic balance between groundwater inputs (recharge) and outputs (discharge) within a watershed. The result is declining water levels until either recharge is increased (e.g., effluent injection) and/or discharge is reduced (e.g., decreasing stream flows, reduced groundwater use by vegetation) to balance the pumping demand. Since both the long-term sustainability of human habitation and riparian health are dependent upon the consequences of groundwater pumping, resource managers and scientists are making a significant effort to improve understanding of the water balance of these regional groundwater systems. An improved description and quantification of the key recharge and discharge processes will greatly support management decisions that will lead to sustainable human communities and the continued health of riparian ecosystems.

The Upper San Pedro River Basin in southeastern Arizona and northern Sonora, Mexico is an ideal area in which to investigate these poorly understood processes of regional aquifer water balance. Unlike many riparian systems that have been disrupted due to the lowering of the groundwater table by pumping, the basin has a lengthy reach of intact perennial flow, which sustains abundant riparian corridor vegetation. In 1988, the U.S. Congress recognized the importance and rarity of this ecosystem by establishing the San Pedro Riparian National Conservation Area (SPRNCA), which protects and enhances

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approximately 70 km of the river and its associated ecosystem. From previous observation and modeling studies, three dominant components of the basin's natural groundwater system have emerged. These three components--mountain front recharge, surface water discharge, and water uptake by riparian vegetation--are estimated to be of similar magnitude (Vionett and Maddock 1992, Corell et al. 1996).

It is widely believed that the presence of large-scale groundwater pumping in the nearby urban areas of Sierra Vista and Fort Huachuca has created a cone of depression which has, or will soon, diminish the baseflows in the river (e.g., Steinitz et al. 2003). The disruption of riparian corridor ecology due to groundwater depletion has been well documented throughout this region (Stromberg 1993, Grantham, 1996). Numerous groundwater modeling and conceptual studies have been performed for various sub-basins of the San Pedro. All of them include the "Sierra Vista sub-basin," the area of principal concern due to the larger amount of pumping therein (Freethey 1982, Vionnett and Maddock 1992, Corell et al. 1996, Steinitz et al. 2003).

In the San Pedro, an important component in the basin's groundwater budget is the amount of groundwater used by riparian vegetation. This flux was traditionally estimated by using groundwater models, where the riparian water use was the residual discharge that resulted from the model after it was calibrated against known inputs, groundwater levels, and discharges (e.g., Corell et al. 1996). Considerable improvements in these estimates have been made in recent studies with the use of actual measurements of riparian vegetation functioning/evapotranspiration (Goodrich et al. 2000, Scott et al. 2000, Schaeffer et al. 2000, Snyder and Williams 2000). Goodrich et al. (2000) combined these measurements with a vegetation map to derive observation-based estimates of riparian vegetation water use for different river reaches within the SPRNCA.

In this paper, we describe a prototype GIS-based tool designed to help management agencies determine the total riparian vegetation groundwater use in the San Pedro Basin and how the groundwater use will likely change with different management strategies. We also analyze how the incorporation of a new vegetation map and water use measurements will change the most the recent estimates of riparian vegetation water use made by Goodrich et al. (2000). One of the limitations of current estimates of vegetation groundwater use is that the amounts are fixed to a particular vegetation state. Our tool allows the user to change the vegetation cover within the riparian corridor. This flexibility allows us to understand how vegetation change due to natural (e.g., succession, wildfires) or human-induced (e.g., prescribed fires) causes might alter the vegetation water use. Additionally, the tool incorporates new, longer-term measurements of mesquite and cottonwood groundwater use that have been made over the last few years. These new estimates help us to better understand the variability of riparian water use and important factors that affect it.

Overview of GIS-Based Tool and Its Component Parts

The GIS-based tool is an accounting model that merges a vegetation map with component vegetation groundwater use models. This tool and its elements are described in the following subsections.

GIS-based tool

The GIS-based tool has a user-friendly interface that allows for easy manipulation of a vegetation map and projection of the seasonal demand of groundwater-using vegetation. The tool calculates the total amounts of different types of phreatophytic vegetation from a vegetation map of the riparian corridor of the Upper San Pedro River and, then, multiplies these amounts by the appropriate seasonal groundwater demand per unit area of vegetation to calculate the total groundwater use. ArcView GIS (ESRI, Redlands, CA) supplies the structure on which the tool is built, and easy to use menus with complete instructions are included. If desired, the user may select any area of a map or any type of vegetation to change. Out of the many different types of land cover in the San Pedro riparian corridor, we have identified the following as significant groundwater-using components: mesquite, cottonwood/willow, sacaton grass, and open water categories.

To modify the vegetation map, the user either supplies a polygon map of the area to be revised (i.e. a prescribed burn), or is prompted to draw a polygon of the area to be revised directly on the vegetation map. Upon starting the tool, the user is presented with a screen showing three choices of vegetation manipulation:

1) all vegetation within a user-defined polygon is changed to a new type of vegetation (e.g., sacaton);
2) one vegetation type within a user-defined polygon is changed to a new type of vegetation (e.g., change saltcedar to cottonwood);
3) simulate a burn, all vegetation within a user-supplied polygon map is changed to a new type of vegetation (e.g., change a prescribed burn area to bare soil).

To perform the vegetation manipulation the user first chooses which one of the above three types of vegetation change to perform. If option number 1 is selected, a new screen appears asking the user to select the grid to modify, the new vegetation type, and the name of the new map to create. If the user has chosen option number 2, the new screen also requests the type of vegetation to change from. If the user has selected the "simulate a burn" option, the user must specify the burn map, the new vegetation type, and name the new map to create. This option may also be used to analyze other types of vegetation manipulation where a polygon map of the area to be modified is available.

If either of the first two options is selected, the user is prompted to draw a polygon using the mouse of the area of interest. After the polygon is drawn, the tool performs the vegetation revisions, creates the new map, and calculates the new groundwater use values for the entire riparian corridor. When the last option is selected, the draw polygon step is skipped, the tool immediately calculates the change in groundwater use based on the user supplied polygon map, and presents the results. Using this option, the progression of vegetation re-growth after a prescribed burn or wildfire is shown. The results from all options are presented as a plot against the values calculated from the original, unaltered map. In all cases, the original vegetation map is not changed; a new map is created each time. The newly created maps may then be used for subsequent analyses.

Vegetation map

Goodrich et al. (2000) made the most recent estimates of riparian groundwater use along the San Pedro using estimates of vegetation area that were made from a 1997 *pixel-based* vegetation classification (hereafter referred to as VEG97). In the map, each 3 x 3 m pixel is classified as a particular vegetation cover. From aerial photography made in 2000 and field data collected in 2001, the U.S. Army Corp of Engineers produced a new *polygon-based*, GIS vegetation cover map (VEG00), where continuous stands of vegetation alliances were delineated and given various attributes like vegetation alliance, polygon area, total area of vegetation cover, area of dominant vegetation cover, etc. It includes 33 different vegetation communities, open water, and urban lands.

The conversion from a pixel- to a polygon-based coverage made the task of computing total vegetation areas for the relevant land cover types more difficult. For the new map, VEG00, both the polygon area and the percent area that is covered by the vegetation of interest were needed to estimate the total area of groundwater-using vegetation. The basic classification in VEG00 has five ranges for the vegetation percent cover. They are: 1 – 10, 11 – 25, 26 – 60, 61 – 80, 81 – 100 %. This range is quite course for calculating the total area covered by a specific vegetation type and induced uncertainty in the new estimates of vegetation groundwater use. To reduce this uncertainty, the map provides the vegetation percent cover estimated to the nearest 5 % for the mesquite or cottonwood polygons classified as a woodland or forest, defined as those patches dominated by mesquite or cottonwood/willow with greater than 60 % cover.

Unfortunately, there were still many polygons not classified as woodland or forest that contain vegetation that uses groundwater (e.g., mesquite patches with less than 60 % cover, sacaton grasslands, etc.). We incorporated this uncertainty into the GIS-tool by providing the user with a choice to calculate the minimum, median, and maximum amount of each functional vegetation group. Then, total vegetation area was calculated by summing up, over all polygons of a certain plant functional group, the product of the polygon area and the minimum, median, and maximum percent cover, or, if the more accurate percent cover was available, then this was used instead.

Evapotranspiration

We used a combination of micrometeorological and eco-physiological measurements to make evapotranspiration (ET) measurements of plant functional groups. Because sacaton and mesquite ecosystems along the San Pedro occupy more extensive and broad areas, we used long-term eddy covariance measurements to get the total ecosystem ET fluxes in these cover types. Scott et al. (2000) made measurements of mesquite and sacaton ET using Bowen ratio techniques. We used sap flow techniques to measure cottonwood transpiration in order to further test the measurements and model of cottonwood water use made previously (Goodrich et al. 2000, Schaeffer et al. 2000). The multiple years of growing season ET observations indicate that groundwater use is quite variable annually. The GIS-tool accounts for this variability by displaying a range for the total amount of groundwater used that has been shown in the observations.

We have made mesquite ET measurements since 2000 at a mature, dense mesquite woodland, while the measurements of cottonwood, sacaton, open water and seep willow water use began in 2003. In this paper, we concentrate on using the mesquite measurements of ET to estimate mesquite groundwater use.

Scott et al. (in review) report in detail on the mesquite ET measurements for the 2001 and 2002 growing seasons. In order to estimate a yearly groundwater use from these ET measurements, we employed a water balance computation for the entire growing season:

$$Q_t = ET - (P - \Delta S) \qquad (1)$$

where Q_t is groundwater use, ET is evapotranspiration, P is precipitation, and ΔS is the change of soil moisture in the top 1 m of soil. At the site, runoff was negligible and there were only small changes in soil moisture deeper than 1 m. Thus, Q_t is the ET in excess of precipitation and soil moisture storage. We assumed that this excess soil moisture is derived from groundwater. Scott et al. (2003) and Scott et al. (in review) showed that the mesquites at the site used groundwater. Lastly, we computed the amount of groundwater used on a per unit *mesquite* area, $Q_{mesquite}$, (rather than per unit *ecosystem* area) by dividing Q_t by the percent cover of mesquite found at the site.

The GIS-tool requires daily estimates of groundwater use rather than ET. For the cottonwood and willows, we will use the sap flow measurements to calibrate a model to estimate the transpiration. We assume that this transpiration is derived mainly from groundwater as shown by Snyder et al. (2000). For the measurements of mesquite and sacaton ET, we plan to employ a simple understory ET model to compute the amount of ET derived from precipitation. Subtracting this from the eddy covariance ET measurements, the mesquite tree or sacaton grass transpiration component will be calculated. Until the results of on-going studies of mesquite or sacaton water sources are known, we will use the simplifying assumption that the tree/grass water source is groundwater. The details and results of this work will be reported in future publications.

Results

We proceed here with a comparison of the vegetation maps and a summary of the mesquite water use estimates. These two issues will greatly influence the water use amounts that the tool will compute.

The change from the grid-based vegetation map, VEG97, to the polygon-based GIS coverage, VEG00, results in dramatic changes in computed vegetation area. As an example of this shift, Table 1 presents the total amount of area covered by each of four groundwater-using groups for the riparian area within Sierra Vista Sub-basin (defined as the San Pedro reach between the Palominas and Tombstone USGS gages.) The range given for the VEG00 map represents the minimum and maximum amounts. Recall that many of the vegetation polygons have an assigned range instead of an exact percent cover. For the reach in Table 1, all the cottonwood and open water polygons have an exact area given to them; hence, there is no range given for these functional groups. This is not the case for the sacaton and mesquite amounts.

Table 1. Sierra-Vista Sub-Basin Riparian Vegetation Areas (ha).

	Vegetation Map	
Vegetation Type	**Veg97**	**Veg00**
Mesquite	1166	721 - 967
Cottonwood/Willow	526	300
Sacaton	382	363 - 513
Open Water	5	42

The GIS-tool accounts for the uncertainty in the vegetation amounts by computing a range of water use for each plant functional type. The range in water use is computed by using the minimum, median and maximum vegetation areas and multiplying each by the appropriate water use amounts. Nonetheless, the change in amount of vegetation between maps will clearly result in a large change in the water use calculations. The magnitude of this change will far outweigh the changes due to the refinement of plant groundwater use amounts. While there have been some vegetation cover changes, mainly due to fires, from 1997 to 2000, it is unlikely that all this change is natural. A further check in the accuracy of the maps is warranted.

Goodrich et al. (2000) identified mesquite water use as the most uncertain and, likely, the most significant component of the total vegetation groundwater use. The three reasons for this uncertainty were: 1) mesquites cover the largest area within the SPRNCA, 2) previous measurements were made from a relatively immature mesquite site probably not representative of denser, more mature woodlands, 3) mesquites can use both precipitation and groundwater as a water source.

Table 2 lists the components of the 2001 and 2002 mesquite water balance and compares them to measurements made in 1997 (Scott et al. 2000). The aerial cover of mesquite at these sites was 0.5 and 0.7 for 1997 and 2001-2002, respectively. While the 1997 measurements were at a site that was considerably less dense, these differences are not sufficient to explain the much greater groundwater use in 2001-2002. The 2001-2002 site, was composed of much larger and more mature trees. The trees at the 1997 site, being less developed, were arguably less adept at tapping the deep groundwater source. (The water-table depth at both sites was ~ 9 m).

Table 2. Mesquite Growing Season Water Balance (May 1 – Nov 30). Units are in millimeters. See Methods Section for term definitions.

	1997	**2001**	**2002**
ET	330	694	638
$P - \Delta S$	173	206	244
Q_t	157	488	394
$Q_{mesquite}$	314	697	563

The new 2001 and 2002 mesquite measurements also show that the mesquite groundwater use varied considerably between the years. In 2002, much drier and hotter conditions prevailed in the first two months of the growing season prior to the onset of the summer rains. The trees showed considerably more stress (Scott et al. in review). It is possible that this stress caused some loss of conductivity in the stems and led to a decreased tree water use throughout the rest of the season. Measurements at this site continue and hopefully will allow us to better quantify and explain this seasonal variability. In the meantime, groundwater use by mesquites in the GIS-tool will reflect the mean seasonal behavior and the variability will be represented by uncertainty estimates in the final groundwater use calculations.

Conclusions

In the San Pedro Basin, the amount of groundwater used by phreatophytic plants is a substantial, yet difficult to estimate, component of the water budget. A combination of improved vegetation maps and understanding of plant groundwater use now makes it possible to better quantify this use in the San Pedro Basin. An easy-to-use GIS-tool will make it possible to communicate these results to management agencies and the public more readily, and it will allow them to better understand how natural and human-induced change will alter groundwater use in the future.

We consider this GIS-tool as a prototype since it is designed to be applied only in the San Pedro Basin and, thus, assumes a certain climate and riparian vegetation functioning for the basin. Future work will entail the development of a more general and flexible tool that can be applied elsewhere. This will be done by allowing the user to specify their own vegetation map, climate data, and vegetation water use models.

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Watershed Modeling I

Estimating Channel Morphologic Properties from a High Resolution DEM

Scott N. Miller

Abstract

Channel morphology plays a critical role in the understanding and interpretation of the hydrologic and geomorphic characteristics of an area. Traditional techniques for determining channel width, depth, and cross-section area are time consuming and may not be truly representative of the spatial variability within a watershed. A method for extracting channel morphologic properties from a high resolution digital elevation model (DEM) is presented. Interferometric synthetic aperture radar data was used to build a high resolution digital elevation model for the USDA-ARS Walnut Gulch Experimental Watershed. While a fully automated technique proved elusive, a quasi-automated system using a geographic information system and expert opinion was successful in estimating channel shape properties. This integrated technique was not as effective in estimating channel depth, but estimated values of channel width were highly correlated with field observations.

Introduction

This paper presents the preliminary results of an investigation into the automated extraction of channel morphologic properties from high resolution digital elevation models. A quasi-automated approach was used to extract cross-section profiles on an ephemeral stream system at numerous sites that had been previously manually surveyed. A geographic information system was used to extract terrain information from the terrain model, which was then subjected to manual interpretation to determine the bank locations, after which the average channel morphologic properties of width, depth, and cross-section area were determined.

Results indicate a high correlation between observed and estimated width, with slightly poorer results for channel depth and area. These findings will be used as the basis for the development of a fully automated system in which channel properties may be extracted from high resolution terrain models.

The accurate estimation of channel dimensions is a critical element in many hydrologic and geomorphic investigations. Process-based hydrologic models that simulate the various components of processes controlling runoff, such as transmission losses, may require that the channel width and depth be known in order to accurately simulate runoff (Smith et al. 1995). For example, Table 1 illustrates the impact of channel width on runoff and sediment yield using the Kinematic Runoff and Erosion Model (KINEROS) (Smith et al 1995).

Detachment and transport of sediment within a stream channel is a function of the energy associated with water moving through a channel reach, and modifications to the estimated channel width may alter the prediction as to whether a given reach will be aggrading or degrading for a given stream flow. Likewise, geomorphic studies that seek to determine the short- and long-term fluxes in sediment and potential for bank failure of alteration in channel planform, require inputs related to channel width and depth.

The hydraulic relationships between channel morphology and runoff were first explored by Leopold and Maddock (1953) in which exponential equations were developed between channel morphologic properties and runoff characteristics. Channel geometry has been used for indirectly estimating streamflow. Because of variability in recording and the possibility for measurement errors in determining channel depth, Hedman and Osterkamp (1982) focused on the relationship between streamflow and channel width and reported relationships between streamflow and channel width for areas of similar climates within the Western United States. By taking into account the shear stress distribution within the channel,

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Table 1. Impact of systematically changing channel width on hydrologic simulations using KINEROS.

	Multiplier of channel width						
	0.7	0.8	0.9	1	1.1	1.2	1.3
Runoff	12.76	8.41	4.07	0.00	-3.85	-7.31	-10.52
Sediment yield	3.88	2.60	1.30	0.00	-1.30	-2.38	-3.37
Peak Runoff	11.90	8.13	4.12	0.00	-4.12	-8.01	-11.97
Transmission Losses	-10.82	-7.13	-3.46	0.00	3.19	6.06	8.70

Osterkamp et al. (1983) further explored the relationships governing streamflow and channel geometry and derived the exponents for the width, depth, and velocity factors as used in hydraulic geometry. Other researchers have found the variables of width and depth to display a large variance, while cross-sectional area displays a strong relationship to flow distance (Miller et al. 1996).

In support of such investigations, a field campaign is typically undertaken in which channel cross-sections are surveyed using either hand-held means or more sophisticated surveying equipment. Such techniques have been shown to be relatively robust, although a degree of subjectivity and error are embedded in the estimation of bank height and location of break points along a given profile. Ephemeral streams are relatively easy to survey when dry, but the nature of runoff in such areas often leads to uneven or indistinct channel forms. Perennial streams may provide a more stable and uniform profile, but are challenging to survey as the surveyor must navigate through inundated areas.

A significant drawback to field investigations of channel morphology is the amount of time necessary to carry out such a campaign. For example, Miller et al. (1996) manually surveyed over 300 channel sections using primarily a line-level and rod approach, with an average of 2.4 profiles per section, in support of a joint watershed / channel morphology project. This field work consumed approximately 40 field days and often required more than one scientist. Using a total station requires a minimum of two people and can consume more time due to increased set-up and mobility constraints. In sum, field-based channel morphology investigations are both time-consuming and costly in terms of manpower and equipment.

Significant advances in hydrologic model development have been made with respect to developing linkages between geospatial data and the parameterization of geomorphic and hydrologic models using geographic information systems, or GIS (Arnold et al. 1994, Miller et al. 2002). In general, these GIS-based linkages rely heavily upon users for the estimation of channel morphologic and hydraulic properties; several do not attempt to utilize channel morphology in the simulation of runoff and avoid the difficulties associated with acquiring intensive field-based data.

An alternative approach to utilizing GIS for the estimation of channel morphology is presented here. A high resolution DEM was acquired using interferometric synthetic aperture radar (IFSAR) at a 2.5 m resolution. Channel profiles were extracted from the DEM at the same locations surveyed by Miller et al. (1996) and compared to the field observations. A methodology for extracting the channel profiles for estimation of channel width, depth, and cross-section area is presented. Results show a high correlation between the observed data and those derived from the DEM.

The objective of this project was to develop a quasi-automated approach to determining channel morphology from a high-resolution DEM using a GIS. Following Miller et al. (1996) it was hypothesized that the estimation of channel width would be in relatively close agreement with the observed values, while the estimation of depth would be less reliable. This approach is intended to provide a basis for a fully automated GIS-based technique for generating spatially distributed estimates of channel width and depth for support of hydrologic and geomorphic modeling.

Description of the Study Area

The USDA - Agricultural Research Service Southwest Watershed Research Center administers the Walnut Gulch Experimental Watershed. Located in southeastern Arizona, Walnut Gulch encompasses the city of Tombstone and contributes runoff to the San Pedro River. Approximately 148 km^2 in size, the watershed is located on the pediment between the Dragoon mountains and the San Pedro River. Climate in this region is semi-arid, with the majority of the rainfall occurring during summer monsoon rainfall, primarily as a result of high-intensity localized convective events (Renard et al. 1993).

Walnut Gulch is within the transition zone between Chihuahuan and Sonoran deserts. Vegetation is a mixture of grasslands (in the upper, eastern portion of the watershed) and shrub-steppe (dominant in the lower, western section). Soils in the watershed are primarily sandy loams, and the stream beds are a mixture of sands and gravels with high infiltration capacities.

The terrain is primarily composed of rolling topography with a dendritic stream network. In some areas relatively shallow depth to bedrock and small-scale faulting exert geologic control over the channel pattern and morphology. A majority of the watershed overlies deep alluvial outwash from the Dragoon mountains. However, igneous and exposed sedimentary rocks form the Tombstone Hills and form much of the southern boundary of the watershed.

Methods

A quasi-automated methodology was employed in the estimation of channel morphologic properties from a high resolution IFSAR DEM. The IFSAR model was built using data collected from a mission flown in the year 2000. Radar backscatter was acquired using low-flying aircraft and a terrain model was generated with a 2.5m resolution. Since no averaging or smoothing was performed on the raw DEM occasional errors in elevation are present. At a larger scale, the cumulative errors cause problems for continuity in the terrain surface and make hydrologic modeling difficult, but at the cross-section scale these data are appropriate, since spurious elevation may be readily identified and excluded from the analysis.

Cross-section profile locations from the field campaign of Miller et al. (1996) were input into a GIS. These cross-sections were located by Miller et al. (1996) using a 0.5 m resolution ortho-rectified aerial photography. In the current effort these section locations were transformed into linear (arc) features using ArcGIS. The aerial photographs were geo-rectified and input as grid features into the GIS and served as background imagery to ensure the correct placement of the profiles.

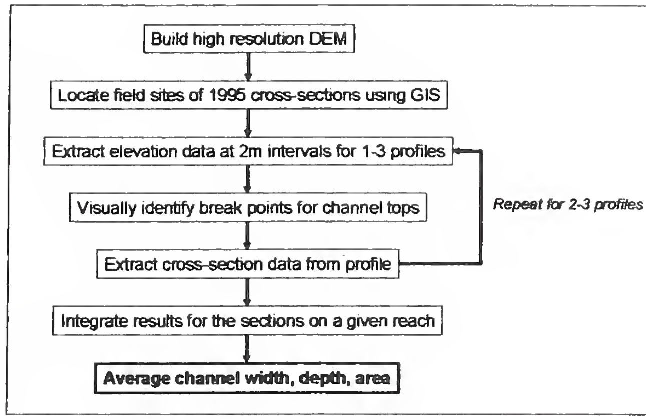

Figure 1. Generalized methodology for extracting channel dimensions using a combined GIS and graphical technique.

In order to capture the variability in channel morphology found within a given stream segment, multiple cross-section profiles were measured at each site. Channel morphologic features were extracted for each profile, and the results averaged to create a composite, or representative, cross-section. Figure 1 shows the generalized approach used to create the composite cross-sections.

Once the individual profiles were built as line features in the GIS they were intersected with the 2.5 m IFSAR DEM. Elevation points at 2 m intervals along the profiles were determined from the DEM. In this way, a three-dimensional representation of the profiles was built and these data were exported into a spreadsheet. A long-term goal of this ongoing research is to automate the determination of the channel banks for the purpose of isolating the exact channel width and depth. However, in this effort an interactive approach was taken to demonstrate the feasibility of using DEM data for channel morphologic investigations. Data exported from the GIS were imported into a spreadsheet and used to create cross-section profile in graphical format (Figure 2).

Field investigations typically rely on indicators such as slope breaks, changes in bed or bank materials, a shift in vegetative type, debris lines, and bank staining may be used to determine bankfull depth (Osterkamp et al. 1983, Gordon et al. 1992). Evidence indicative of a constructive, rather than destructive process is preferable in the determination of bank height; in the southwestern United States channel processes are governed by rapid and violent runoff events, and many of the channels on Walnut Gulch are actively degrading and therefore not in equilibrium.

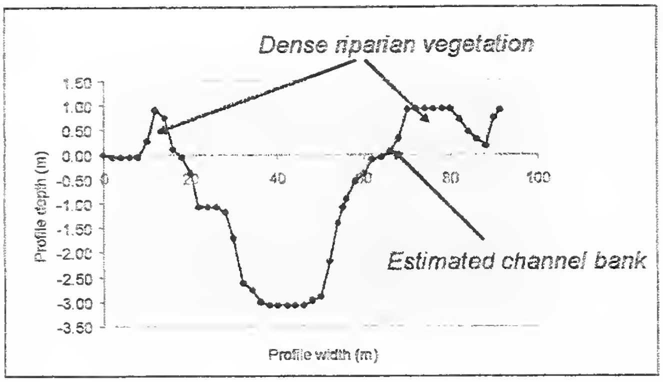

Figure 2. Example of channel profile extracted from DEM. In this example the channel bank is obscured by dense riparian vegetation that obscures the true elevation of the land surface.

A significant drawback to using a GIS is the inability to estimate whether a channel is stable or to accurately depict the correct location of a channel bank. In systems such as Walnut Gulch, where repeated incision has developed stream segments with multiple overbank deposits, bank height is often a subjective interpretation. Using a GIS increases the subjectivity in estimating bank location and introduces error. In the case of high frequency IFSAR such as was used in this experiment, dense vegetation often reflected the radar pulse and obscured the true ground elevation. Figure 2 illustrates this problem; in this case a relatively thick vegetative cover at the channel banks is present as an anomaly in the DEM surface. Because of these restrictions, an interactive technique was developed where the user manually interpreted the location of the channel banks.

Once the approximate location of the channel banks was determined, channel width was derived by simply locating the paired location of the opposite bank and determining the distance. Channel depth for a given profile was calculated by weighting the difference in elevation between each two-meter segment and the datum represented by the bankfull elevation. Cross-section area was calculated as a product of width and depth. Each stream section has between one and three profiles whose results were averaged to produce the composite estimated morphology values.

Results and Discussion

Composite cross-section results were compared to those collected by Miller et al. (1996). It should be noted that the techniques for creating composite values for width, depth, and area were identical for these two studies. In most cases the individual profile locations were located within a tolerance of +/- 5 m. However, there were sites where errors in the IFSAR DEM made it impossible to generate a valid cross-section profile. In these cases, the profile was moved slightly up- or downstream several meters. Comparisons between the channel properties show a high degree of correlation among all three measured values (Figure 3).

As hypothesized, channel width was most highly correlated with the field observations. IFSAR channel widths have a Pearson's correlation value to the field observations of 0.87, and the coefficient of determination produced from simple linear regression is 0.74. Channel width is also highly correlated (Pearson's correlation of 0.72), but the r^2 value resulting from regression analysis is 0.52. Given that cross-section area is a product of channel width and depth, the comparative results between the IFSAR and field data were in between the channel width and are observations (Pearson's value of 0.80, r^2 of 0.63).

A two-sample t-test was used to determine if the sample populations between the IFSAR and field observations were identical. Results for channel width revealed that the samples were the same, while those of channel depth and area were not. A closer inspection of Figure 3 underscores some relevant differences in the populations. Overall, IFSAR values for channel width underpredicted the observed data. This underprediction is apparent in the regression relationship detailed in Figure 3a, where the slope of the regression is 0.82 with an offset of 9.1. In contrast, the predicted IFSAR depths fall almost entirely below the 1:1 line, indicating a clear tendency for overprediction. In this case, the slope of the regression is similar to that of width (0.79), but the offset is -0.03 (Figure 3b). In both cases the regression slopes indicate a tendency to increase their predictions relative to the observed channel morphology at high values.

While the regression relationship for width was significantly better than for depth, the absolute error of the measurements was considerably greater. The average absolute error for channel width was 10.1m with a standard deviation of 7.8. The average absolute error for

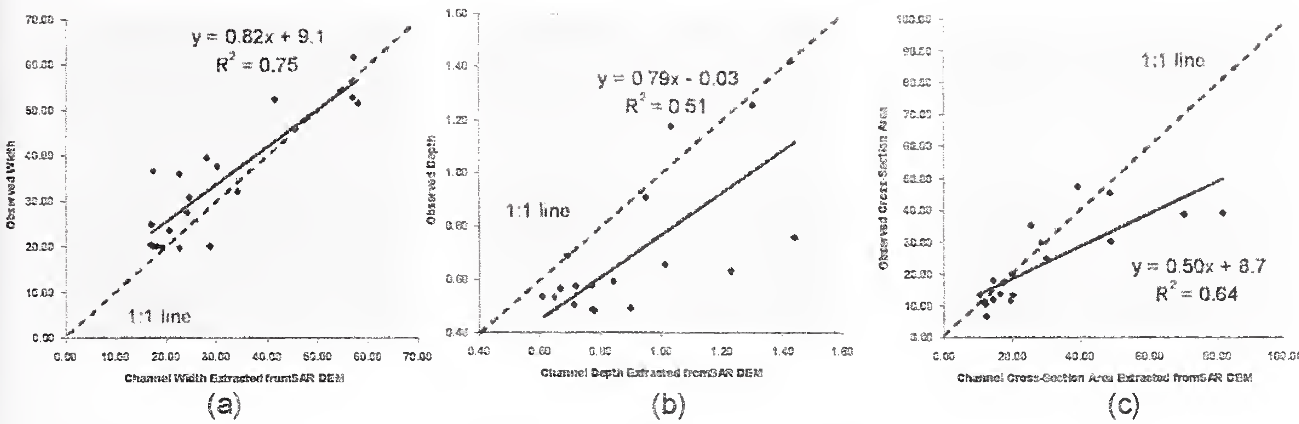

Figure 3. Comparisons of channel cross-section width (a), depth (b), and area (c) estimated from high resolution IFSAR DEM and field observations.

channel depth, on the other hand, was 0.18 with a standard deviation of 0.18. The channels of Walnut Gulch have a high width:depth ratio (53 in the case of these 20 samples) and are relatively rectangular. Thus, errors in channel width approximation do not greatly affect the estimate for channel depth. The average absolute percent errors for width and depth were relatively similar (0.29 and 0.26, respectively) with similar standard deviations (20.0 and 20.7, respectively).

Several difficulties were encountered in the extraction of cross-section data from the IFSAR DEM. The ability to discriminate small features is a function of the vertical and horizontal resolution of the terrain data, and in this project, the 2.5 m resolution of the data was an impediment to extracting fine landscape features. As noted earlier, the presence of small shrubs and dense riparian vegetation along the channel banks introduced error into the determination of the channel banks and possibly contributed to the errors in estimation. In some cases spurious elevation data were present in the DEM. These errors resolved themselves as anomalous sinks or spikes in the surface model. In such cases, the cross-section profile was slightly moved, but these anomalies are a serious impediment to the development of fully automated systems. The appropriate method for removing such spurious data is to average the raw 2.5 m data to produce a hydrologically correct 10 m DEM. This process obviously degrades the ability of the examiner to discriminate fine features in the landscape and would lead to greater errors in the estimation of channel morphology.

A time lag of approximately five years separated the field observations of Miller et al. (1996) and the IFSAR mission. Conceivably, some of the cross-sections investigated in this project could have undergone substantial change in their morphology. However, these changes are more likely to be reflected in the estimated channel depth. Miller et al. (1997) demonstrated that the effective return flow that contributes to channel forming processes on Walnut Gulch is 10 years. Thus, the likelihood of a channel-forming event having occurred on the field sites is relatively low. A review of the intervening years reveals an absence of large runoff events down the main stem of the watershed. Thus, while the time lag is unfortunate with respect to absolutely defining the differences between the two methods, it is unlikely that the basic channel morphologies observed by Miller et al. (1996) were significantly affected by runoff prior to the initiation of this investigation.

Whether in the field or using a GIS, extracting channel cross-section data is a somewhat subjective exercise, especially in ephemeral streams such as Walnut Gulch. The classic thalweg-bank-floodplain complex found on perennial streams is often absent in a sandy wash. Actively degrading channels may form no definitive bed or bank features to guide the observer in determining the appropriate bankfull depth. Field observations in such environments are relatively challenging, and the researcher must often utilize secondary information such as flood debris, soil properties, or vegetative characteristics. None of these tools is available to the researcher attempting to design a cross-section based on profiles extracted from a DEM. The user must rely on the presence of obvious landscape features; where none exist the prospect of determining an appropriate width or depth is futile. However, in the majority of cases, the channel form was clearly apparent in the graphical representation of the channel profiles. Thus, this approach shows promise for the future development of fully automated GIS-based techniques.

Conclusions

A quasi-automated approach for extracting channel morphologic properties from high resolution digital elevation models was developed. Results indicate that the extracted channel properties of width, depth, and cross-section area were highly correlated to field observations. While the absolute errors in channel width were greater than for depth, the percent errors for these measures were approximately the same. Statistical analyses showed that the populations of the field observations and IFSAR DEM-based estimates for channel width were the same.

While several obstacles remain in the development of a fully automated GIS-based routine for the estimation of channel morphology, these results are encouraging. Current research efforts are focused on developing an ArcMap tool that would represent a more streamlined approach and reduce user interaction. However, due to the high degree of subjectivity and the reliance on expert opinion in locating bankfull depth, it is expected that some degree of interaction will be required.

Field observations are generally preferable to secondary data extracted from terrain models. However, the high cost of pursuing detailed field work necessary for process-based hydrologic and geomorphic models makes a large-scale effort both difficult and time-consuming. It is anticipated that better terrain models with high vertical and horizontal resolution, such as LIDAR, will allow for a more detailed approach. The improvement in resolution and accuracy will potentially allow for the creation of a logical rule-based system for determining the channel banks based on some minimal user input. Process-based models are sensitive to the estimation of channel morphology, and the automated parameterization of channel properties would be of significant benefit to these lines of research.

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Multi-Scale Evaluation of Watershed Health in the Delaware River Basin and CEMRI

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Abstract

The Delaware River Basin is in the coastal Mid-Atlantic region of the United States, covers 12,700 square miles of primarily forested land, and is home to 7.2 million people. Major watershed issues in the Delaware River Basin (DRB) are urbanization and forest fragmentation, varied abiotic and biotic stressors, diminished condition of forest soils and plants, invasive exotic species, and relationships between forest terrestrial and aquatic processes. In 1998 the U.S. Forest Service (USFS), the U.S. Geological Survey (USGS), and the National Park Service (NPS) formed a Collaborative Environmental Monitoring and Research Initiative (CEMRI). Initial efforts involved combining monitoring and research efforts of participating Federal programs to evaluate health and sustainability of forest and fresh water aquatic systems in the DRB. In 1999-2002, CEMRI focused on urbanization and forest fragmentation, carbon stocks and fluxes, nitrogen saturation and calcium depletion, vulnerability to exotic insects, and improving knowledge of associations between terrestrial and aquatic processes. Models were developed or modified to associate process-level information (from two long-term watershed monitoring sites representing three physiographic provinces) with landscape-scale information (from satellite, aerial, and ground monitoring systems). Eventual goals are development of models of responses of forest and aquatic processes to perturbations, estimation of future forest condition, and identification of threats to forest health and sustainability. This paper discusses development of CEMRI, utilization of the Intensive Site Monitoring component of the Forest Health Monitoring program, terrestrial and aquatic issues to be addressed by this collaborative effort, and proposed methods to evaluate issues across multiple spatial scales.

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Keywords: watershed health, multi-scale assessments, carbon cycling, forest monitoring, calcium depletion

Introduction

Resource management agencies often seek a holistic approach to management of ecosystems, but few agencies have the resources to support multi-component ecosystem-level research. Much of the information collected by any one agency is often fragmentary and incompatible, because of the lack of resources to design and collect multi-resource data (terrestrial, aquatic, atmospheric, etc.) data at multiple spatial scales.

One solution is to develop a concept for "virtual" integration of the capabilities of diverse agencies to address environmental problems in a holistic manner. By supplementing and/or adjusting existing monitoring and research strategies, collaborating programs could continue to meet specific agency missions while also contributing to multi-scale, multi-resource inventory and monitoring systems. To test this approach, several federal agencies and academia developed a strategy to evaluate the condition of forests and associated waterways in the Delaware River Basin (Figure 1).

Databases derived from an ecologically and spatially broad integrated monitoring system, and incorporated into an analytical modeling framework, should allow for more accurate parameterization of scientific and resource management models to improve predictive capability. The ultimate goal is to improve the ability to monitor ecosystem status and change across a range of temporal and spatial scales, and thus provide earlier and more accurate detection and prediction of environmental change.

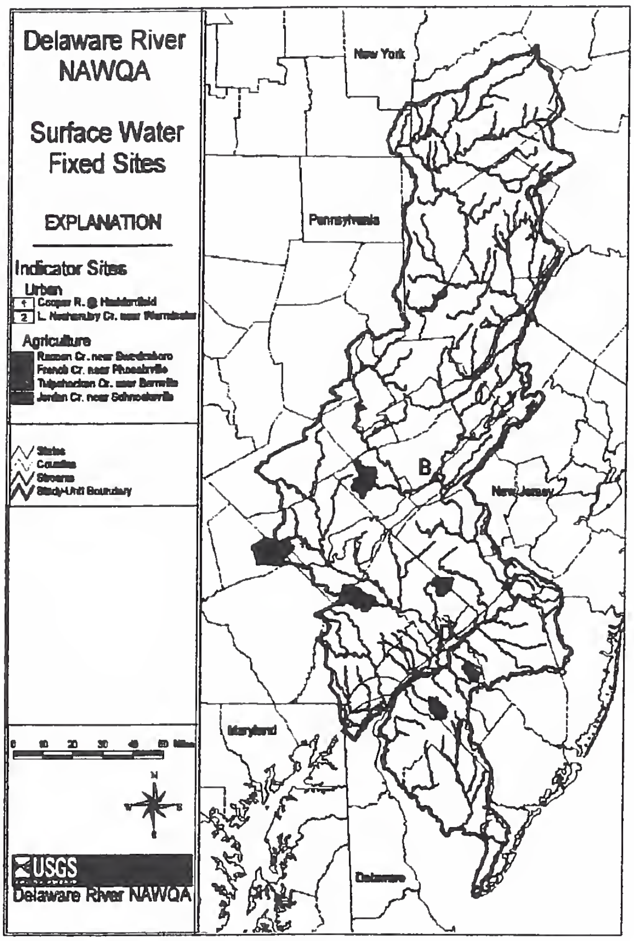

Figure 1. The Delaware River Basin in the Mid-Atlantic region of the U.S. is ecologically diverse. The USGS NAWQA program monitors water quantity and quality in forest and agriculture land types. Source: Gilliom et al. 1995.

The Delaware River Basin (DRB) in the eastern United States was chosen as CEMRI's first pilot region because:

- The river basin hosts several existing agencies and programs accepting the challenge of designing an integrated monitoring system to address specific issues, using their existing monitoring systems.
- The DRB is a relatively simple version of a river basin-to-estuary landscape delineation, with a single large river entering the estuary, as opposed to more complicated systems such as the Chesapeake Bay drainage.
- The DRB is a logical conceptual unit for integrating environmental information on a regional scale, because the aquatic systems are integrators of many environmental issues.
- The DRB contains several Intensive Monitoring and Research (IMR) sites (Neversink, DEWA, French Creek), representing two of five major physiographic regions, with long-term research and monitoring of ecological components and processes.

The DRB encompasses more than 12,700 mi^2 and includes parts of Pennsylvania (6,465 mi^2), New Jersey (2,969 mi^2), New York (2,363 mi^2), Delaware (968 mi^2), and Maryland (8 mi^2) (Figure 1). About 7.2 million people live within the Basin, and an additional 7 million people in New York City and northern New Jersey rely on surface water diverted from the Basin for their water supply.

Methods

The Untied States Department of Agriculture (USDA) Forest Service (FS) and the United States Geologic Survey (USGS), Water Resources Division initially conceived the approach of evaluating several important forest health issues in the Delaware Basin using a multi-spatial scale, multi-resource approach, primarily the integration of forest and aquatic resources. These strategies for multi-agency and resource collaboration became known as the Collaborative Environmental Monitoring and Research Initiative (CEMRI), and the initial pilot test was conducted in the Delaware River Basin.

Shortly thereafter, this initial CEMRI group was joined by the National Park Service (Delaware Water Gap National Recreation Area (DEWA), National Air and Space Administration (NASA), and the United States Environmental Protection Agency (EPA). The Delaware CEMRI pilot integrated environmental data collection across a range of temporal and spatial scales administered by diverse agencies, including process-level research sites (Neversink Watershed, Delaware Water Gap NRA, French Creek), regional ground monitoring systems using long-term, fixed-area plots, and wall-to-wall remote sensing programs (aerial photography and satellite pixels).

The CEMRI approach is an effective environmental monitoring strategy based on a collaborative structure for independent monitoring and research programs to combine complementary activities to address specific environmental questions at multiple spatial, temporal, and ecological scales. In general, most current environmental monitoring programs address issues of one resource type and at one spatial scale, yet information needs often require data from multiple resources that span three broad categories of spatial and temporal monitoring (Figure 2):

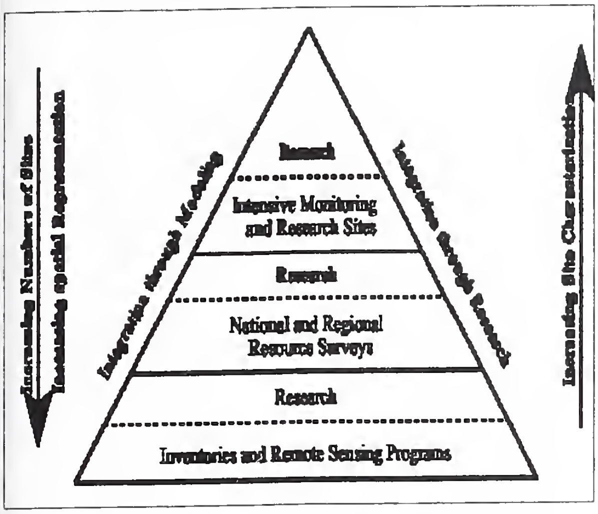

Figure 2. Conceptual framework for conducting evaluation of forest and watershed issues that span multiple spatial scales. Source: Murdoch and Jenkins 2003.

- **Tier 1** – temporally and spatially continuous landscape-scale monitoring and analysis (such as satellite remote sensing, aerial photography, etc.). Primary objectives are development of spatially-continuous coverage of information, such as land-use and change, forest species distributions, forest fragmentation, etc.

- **Tier 2** – frequent multi-point ground monitoring (plots and surveys) at large regional scales. Purpose is to document the status, change, and extent of unusual or disturbed conditions based on ground-collected information, link process-level Tier 3 information to landscape-scale Tier characterizations, and to provide ground verification of remotely sensed parameters in Tier 1.
- **Tier 3** – frequent monitoring and intensive research on components, processes, and determining cause-and-effect relationships at a limited number of intensively monitored sites within relatively small areas. The purpose is to determine key forest processes, understand how these processes inter-relate, how stressors affect these processes, and how these processes relate to macro-indicators that can be monitored at large special scales. In CEMRI these process-level sites are referred to as Intensive Monitoring and Research (IMR) areas.

One measure of the success of the CEMRI pilot will be the comparison of a pre-integration assessment of existing monitoring and interpretation capability with new information obtained in the post-integration assessments. The pre-integration assessment (Murdoch and Jenkins, 2003) addressed the five issues listed above, identified data gaps in existing monitoring programs, and provided a benchmark against which the benefits of the CEMRI approach in the Delaware Basin could be evaluated.

Physiographic Provinces

Ecologically the DRB is very diverse, with the mostly flat Coastal Plain in the south, with soils underlain by unconsolidated sediments (Figure 1). Further north rolling lowlands and a series of broad uplands in the Piedmont are found, with soils underlain by metamorphic rock. North of the Piedmont Province, the New England and the Valley and Ridge Provinces consist of rock layers that have been deformed into a series of steep ridges and parallel folds that trend more northeast-southwest.

The Appalachian Plateau occupies the upper one-third of the basin, and is characterized by rugged hills with intricately dissected plateaus and broad ridges. Bedrock in the Appalachian plateau consists of inter-bedded sandstone, shale, and conglomerate. Topography in the Basin ranges from sea level in the south to more than 4,000 feet elevation in the north.

During the last major glacial advance, the Appalachian Plateau and parts of the Valley and Ridge and the New England Provinces were glaciated. North of the line of glaciation, valleys typically are underlain by thick layers of stratified drift and till. The primary focus of the CEMRI pilot is in the forested landscape of the Appalachian Plateau section of the Delaware Basin. An assessment of urbanization, forest fragmentation, and water quality included both the Piedmont and Appalachian Plateau study areas.

Average annual precipitation ranges from 42 inches in southern New Jersey to about 50 inches in the Catskill Mountains of southern New York; annual snowfall ranges from 13 inches in southern New Jersey to about 80 inches in the Catskill Mountains (Jenner and Lins 1991). Generally, precipitation is evenly distributed throughout the year. Annual average temperatures range from 56°F in southern New Jersey to 45°F in southern New York.

Analysis of 1992 satellite-derived Thematic Mapper (TM) land-use data estimated that about 60% of the DRB is forested land, 24% is agricultural, 9% is urban and residential, and 7% is surface water bodies and miscellaneous land uses. Eighty percent of the population of the study unit lives in the Piedmont and Coastal Plain Provinces in the southern portion of the DRB, which cover only about 40% of the total area.

Results

Explicit collaboration among the participating agencies was essential for collection of data that was both complementary and comparable, focused on the environmental issues identified below, and included data gaps identified in the pre-integration assessment.

Numerous meetings of the lead agencies led to the identification of issues that required data from multiple spatial scales and multiple resource groups. The regionally-integrated monitoring in the CEMRI pilot targeted five specific environmental issues that required data from multiple resource groups collected at three spatial scales:

- Measuring and monitoring forest carbon stocks and fluxes;
- Identification and monitoring of forests vulnerable to non-native invasive pest species;
- Monitoring recovery from calcium depletion and nitrogen (N) saturation in forests of the Appalachian Plateau;
- Measuring and monitoring forest fragmentation and associated ecosystem changes; and
- Integrating the effect of terrestrial ecosystem health and land use on the hydrology, habitat, and water quality of the Delaware River and Estuary.

Collection and synthesis of data were accomplished by augmenting existing monitoring programs already established in the DRB, including local, process-level studies, regional ground monitoring using fixed-area plots and surveys, and remote sensing with aerial photography and satellites.

For example, the FIA and FHM monitoring programs collaborated on the development of a national standardized monitoring system of fixed-area plots (1/16th acre plots composed of 4 subplots) (http://www.fia.fs.fed.us/). These plots are the basis for the collection of a broad suite of ecologically-based indicators that address many of the Criteria and Indicators of the Montreal Process (Anonymous 1995).

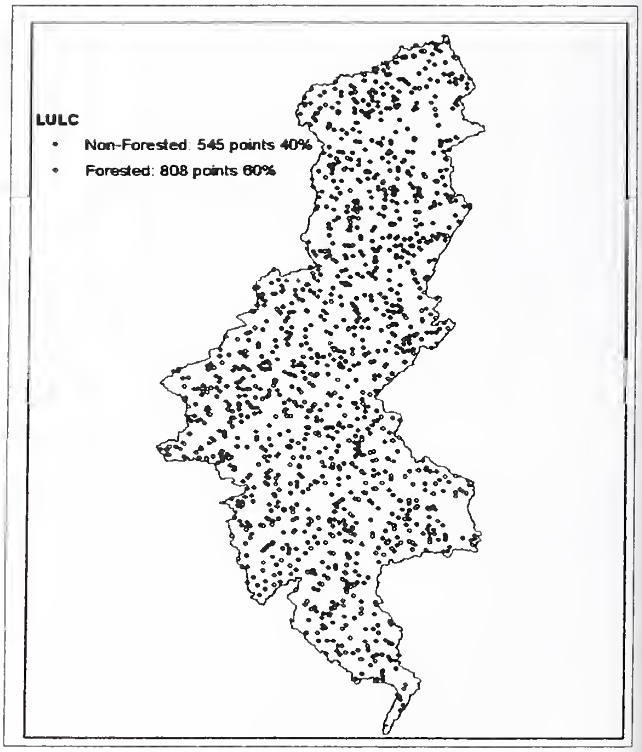

Figure 3. Distribution of FIA fixed-area plots within Delaware River Basin. Phase 2 plots are 1 plot per 6,000 acres. Phase 3 plots are 1 plot per 96,000 acres. Source: Murdoch and Jenkins 2003.

Within the Delaware Basin, these plots are the primary data source for carbon sequestration, forest composition, structure, and condition, and biological diversity. Information on trees is the primary focus of the Phase 2 (P2) sampling framework (1 plot per 6,000 acres), and information on forest health comes from the Phase 3 (P3) plots (1 plot per 96,000 acres) (Figure 3). P3 plots are P2 plots where data on soils, dead wood, air pollution indicators, fuel loading, understory diversity, etc. (Stolte et al. 2002) are collected with P2 tree data.

Intensive monitoring and research areas

Three watersheds in the Delaware River Basin were selected as Intensive Monitoring and Research area (IMRs) for the process-level studies in forested landscapes. Each IMR site was selected for process-level studies because it contained existing monitoring infrastructure or programs. Additionally, the three IMRs represent a range of climatic and forest conditions found in the Delaware Basin (Figure 4).

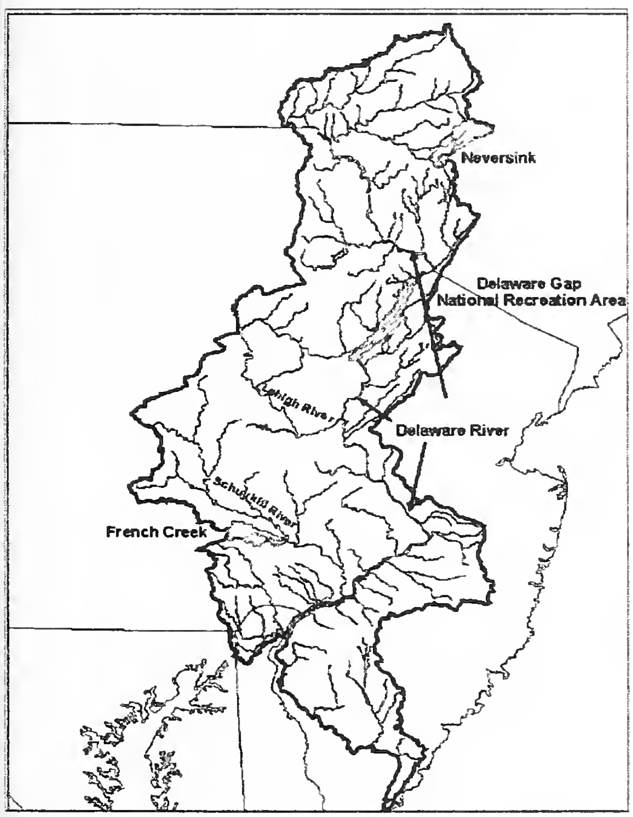

Figure 4. Site of intensive monitoring and research areas (IMRs) in the Delaware River Basin. Source: Murdoch and Jenkins 2003.

The IMR sites selected for the CEMRI pilot were:

1. The Neversink watershed has been monitored by the USGS District Research Program since 1982. It lies within the northern, more forested Appalachian Plateau province, and contains a set of nested discharge and water quality monitoring stations in basins ranging in size from 130 to 1 km^2.
2. The Delaware Water Gap National Recreation Area (DEWA) also lies within the Appalachian Plateau province. Discharge and water quality monitoring stations have been operated during the past 4 years on Flat Brook in DEWA as part of NAWQA (Gilliom et al. 1995). The National Park Service (NPS) has established special forest plots at DEWA for studying the effects of the introduced insect hemlock woolly adelgid (*Adelges tsugae*).
3. The French Creek watershed is located in the mid-basin Piedmont province and contains a partially-forested landscape that is being rapidly suburbanized. Discharge and water quality monitoring stations have been operated during the past four years on the main channel of French Creek.

At the Neversink and Delaware Water Gap IMR sites, the FIA and FHM programs have enhanced the P2 sampling frame to provide a finer-resolution fixed-area plot sampling of the watersheds where process-level monitoring was occurring. These plots are called Phase 4 (P4) plots. Also within each IMR, other P4 plots are intentionally placed directly within process-level research and monitoring studies. These latter plots are called Phase 5 (P5) plots (Figure 5).

Since the same data on trees, soils, down wood, etc. is collected at P2, P3, P4, and P5 spatial scales, all within the same plot framework, the relationship between processes and indicators collected at the IMR sites (P4 and P5) can be used to interpret the condition of the forests at the larger spatial scales, since they are all systematically linked in a common sampling frame with the same fixed-area plots and indicator data (Dunn and Stolte 2003).

For example, a system for integrating process-level information on C cycling rates with the regionally-extensive forest monitoring network and the NAWQA surface water monitoring program is being developed at the three IMRs. Variables such as fine foliar litterfall, coarse woody debris production, foliar chemistry, and soil C and N stocks will be measured at the P4 and P5 plots (Tier 3), and relationships between these variables and indicators typically measured at P2 and P3 plots (Tier 2), and also measured at P4 and P5 plots, such as forest type and basal area, will be developed and applied to develop regional estimates of C dynamics.

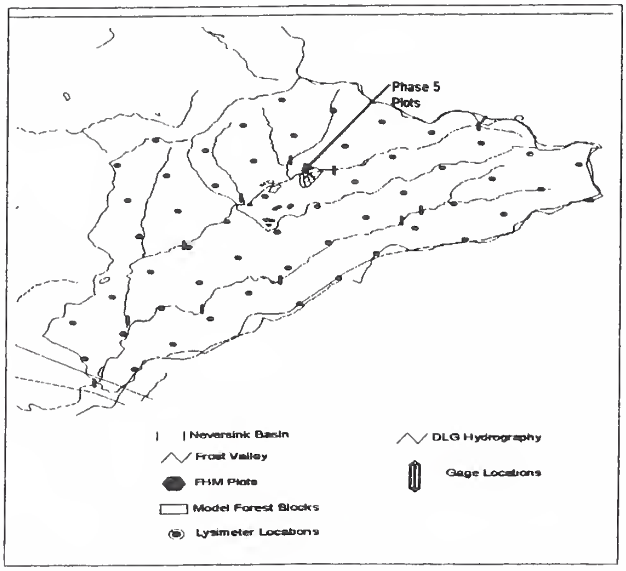

Figure 5. Locations of P4 and P5 plots within the Neversink watershed. IMR area. The watershed is highly instrumented to measure input/output budgets for C and N relationships between forests and aquatic systems. Source: Murdoch and Jenkins 2003.

Also, since the P2 and P3 (Tier 2) sampling is national in scope, the enhanced Tier 2 data within the DRB can also be compared to forest conditions in similar ecological strata outside the Delaware Basin. The CEMRI program will integrate data across spatial scales (Tiers 1, 2, and 3) using two primary models on forested landscape data in the Delaware River Basin – the SPARROW model developed by the USGS and the PnET model developed at the University of New Hampshire.

The SPARROW (Spatially Referenced Regressions of contaminants on Watershed attributes) will be used as a tool for cross-linking data from different programs collecting environmental data within the Basin. The model has the ability to link process data from small-scale watershed studies with monitoring data from large-scale river sites.

The SPARROW will use an empirical model to evaluate the CEMRI data collected on a watershed-wide basis in forested landscapes, and will link that modeled export to estimates developed by the USGS NAWQA program for urban and agricultural landscapes – in this way, the model will create overall estimates of watershed export of chemical constituents within the DRB. The model output will then be compared to measured export values computed at USGS monitoring stations nested within the DRB (Murdoch and Jenkins 2003).

Thus smaller-scale stream segment data will be independently generated by the development of the National Hydrologic Dataset (NHD) by the USGS for the DRB as part of the collaborative effort. The NHD is a 1:24,000 river-reach dataset with full GIS capability. The NHD for the Delaware River Basin will be compiled and referenced to a 30 m digital elevation model (DEM), and to a 10 m DEM in selected portions of the watershed.
Ecosystem process modeling for the DRB can be derived from a combination of the CEMRI databases and existing work on climate change scenarios and N deposition on forested regions of the mid-Atlantic and Chesapeake River Basin (Hom et al. 1998, Pan et al. 2000) using the PnET family of models.

The PnET model is a process-based ecosystem model that uses spatially referenced information on vegetation, climate and soil to make estimates of important variables of forest ecosystems such as carbon storage, net primary production (NPP), water yield, and N leaching loss. It includes a series of compatible sub-models. The model was well validated for NPP and water yield predictions at locations within the northeastern U.S. (Ollinger et al. 1998). In addition to general input information, the model also requires data on N deposition, ozone, and atmospheric CO_2 in order to examine the impact of changing atmospheric chemistry on forest ecosystems.

Conclusions

The CEMRI approach of virtual integration of existing monitoring programs to address key environmental issues at multiple spatial scales is a practical solution that requires determination of participating agencies, clearly identified issues to evaluate, and modifications of existing monitoring systems and analytical models. The combined research and monitoring systems created produces a desirable template to attract other monitoring and research programs to this data-rich arena. Other groups have joined the CEMRI pilot, and additional assessment issues now include the cooperation of groups that focus on the study of urban and suburban land dynamics. The CEMRI group is continuing to pursue collaboration with monitoring and research groups working with agricultural landscapes, and groups working with the estuary environment of the Delaware Basin.

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Process Representation in Watershed-scale Hydrologic Models: An Evaluation in an Experimental Watershed

T.L. Veith, M.S. Srinivasan, W.J. Gburek

Abstract

Hydrologic response varies within a watershed as a function of topography, soil, and land cover. Spatial and temporal data from experimental watersheds may provide information on where, when, how, and why the response varies. This study examined the hydrologic response of an agricultural watershed, FD-36, in the Appalachian Valley and Ridge physiographic region. FD-36 is characterized by shallow, fragipan soils in near-stream areas and deep, well-drained soils in upland areas. Three computer simulation models – Areal Nonpoint Source Watershed Environmental Response Simulation (ANSWERS-2000), Soil and Water Assessment Tool (AVSWAT2000), and Soil Moisture Distribution and Routing (SMDR) – were used to simulate the surface hydrologic processes in FD-36. The three models vary in their temporal and spatial process representations. AVSWAT2000 and SMDR are daily time-step models while ANSWERS-2000 runs at a one-minute time step. Spatially, ANSWERS-2000 and SMDR divide the watershed into grid cells; AVSWAT2000 uses hydrologic response units (HRUs). Of the three models, temporal output from AVSWAT2000 matched measured stream flow most closely ($r^2 = 0.67$). ANSWERS-2000 and AVSWAT2000 both reacted to variations in land cover and soils, whereas SMDR did not. ANSWERS-2000 and AVSWAT2000 indicated the majority of high runoff depths from croplands on near-stream, fragipan soils. Overall, AVSWAT2000 was determined the most favorable for depicting hydrological processes in FD-36, although spatial representation of runoff processes in this model may need further refinement.

Keywords: variable-source-area, storm flow response, land management, simulation models

Introduction

Variations in spatial and temporal efficiencies of watershed-scale rainfall-to-runoff conversion have led to stream flow generation concepts such as variable-source-area (Hewlett 1961) and partial-source-area (Dunne and Black 1970). Often, less than 10% of a watershed directly participates in storm flow generation (Freeze 1974). Even in these hydrologically active areas, rainfall-to-runoff conversion rates vary with the types of runoff generation processes: infiltration excess or saturation excess. Spatial and temporal variations in the hydrologic behavior of a watershed directly impact nutrient transport from land to water. Engman (1974) argued that management of nonpoint source pollution at the watershed scale could be confined to controlling losses from hydrologically active areas.

The cited studies motivate a need for accurately modeling spatial as well as temporal hydrologic responses within a watershed before modeling pollutant losses within that watershed. Simulation models have been very useful in studying spatial and temporal hydrological processes at watershed scales (e.g., Beven and Kirkby 1979). The objective of this study was to determine the impact of model representation of spatial and temporal processes on the characterization of runoff generation for a case watershed.

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Watershed Description

The study watershed, FD-36, is a 39.5-ha headwater subwatershed of the USGS-gauged Mahantango watershed in east-central Pennsylvania. FD-36 is a study watershed of the Pasture Systems and Watershed Management Research Unit, USDA-ARS. Hydrology and nutrient transport studies have been conducted in FD-36 since 1996. Previous fieldwork has provided 5-m grid detail on topography and soil classification. Multi-year data were also available on the dynamics of weather, land management, and stream flow. Climate is typically temperate and humid. This watershed has a mixed land use: 50% soybean/wheat/corn, 30% forest, 19% pasture, 1% urban (Figure 1).

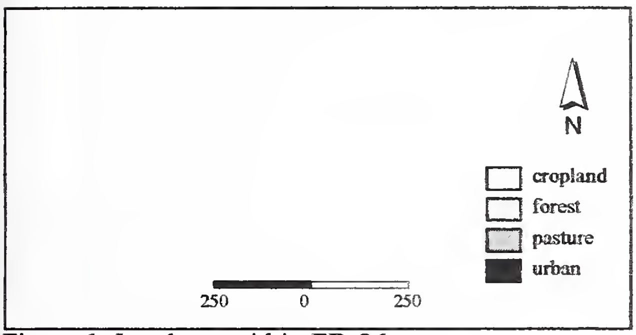


Figure 1. Land use within FD-36.

FD-36 is characterized by shallow, fragipan soils in near-stream areas and deep, well-drained soils in the uplands (Figure 2). Fragipan soil layers impede percolation, facilitating lateral flow. Field studies by Zollweg (1996) and Srinivasan et al. (2002) in an adjacent, non-fragipan watershed established that near-stream areas are hydrologically active during storm events. A landscape-scale study in FD-36 demonstrated the dominance of fragipan soils in runoff generation (Needelman 2002). FD-36, with fragipan soils in near-stream areas, appears to be a good candidate for model comparisons of hydrologically active areas.

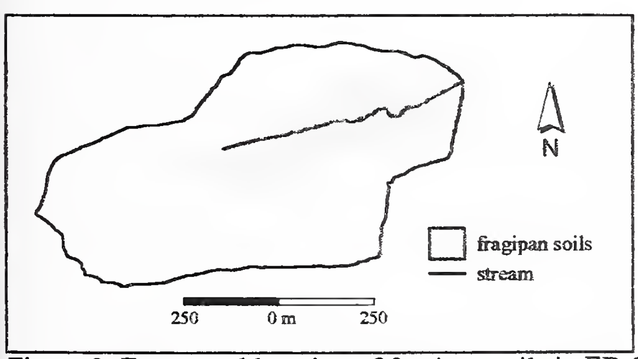


Figure 2. Extent and location of fragipan soils in FD-36.

Simulation Models

Three continuous, watershed-scale, simulation models were selected: ANSWERS-2000 (Bouraoui and Dillaha 1996), AVSWAT2000 (Arnold et al. 1998), and SMDR (Soil and Water Laboratory 2002). All three models are designed for use in un-gauged, agricultural watersheds. ANSWERS-2000 and SMDR are physically-based and not suited to calibration. AVSWAT2000 can be calibrated when data are available, as was the case in this study. Additionally, each model represents spatial and temporal processes differently (Table 1).

ANSWERS-2000 and SMDR use more detailed spatial resolution than does AVSWAT2000 (Table 1). Also, ANSWERS-2000 employs a much smaller time step for both precipitation and model processing than do the other two models. These increases in temporal and spatial resolution have the potential to improve depictions of hydrologic response.

ANSWERS-2000 calculates runoff using both infiltration and saturation excess mechanisms (Table 1). AVSWAT2000 uses the Curve Number approach to calculate runoff based on soil moisture and land cover. SMDR calculates surface runoff as saturation excess.

The rigor of surface flow routing within the watershed declines from ANSWERS-2000 to AVSWAT2000 to SMDR (Table 1). However, both AVSWAT2000 and SMDR route subsurface flow laterally, while ANSWERS-2000 contains only a limited groundwater recharge component and no stream base flow component. Absence of a subsurface flow component may be a disadvantage in FD-36 where lateral flow across fragipan soils is thought to be a key factor.

Results and Discussion

The models were run for a four-year period, 1997-2000. During this period the average annual precipitation was 1021 mm, resulting in an average measured runoff depth of 393 mm. For the purpose of discussion, 1999 was considered typical of the four-year simulation period. In 1999, 1021 mm of rainfall and 297 mm of stream flow were recorded.

Table 1. Comparison of spatial and temporal processes within simulation models.

		ANSWERS-2000	AVSWAT2000	SMDR (ver. 2002)
1.	Watershed representation	5-m grid	Hydrologic response units (HRUs): unique combinations of soils and land use	5-m grid
2.	Simulation interval	60 seconds	Daily	Daily
3.	Precipitation interval	Breakpoint	Daily	Daily
4.	Rainfall-runoff conversion	Green-Ampt infiltration equation	Curve Number (adjusted for soil moisture)	Infiltration capacity from soils data
5.	Surface flow routing	Cell to cell	HRU to stream	Cell to watershed outlet

Temporal output

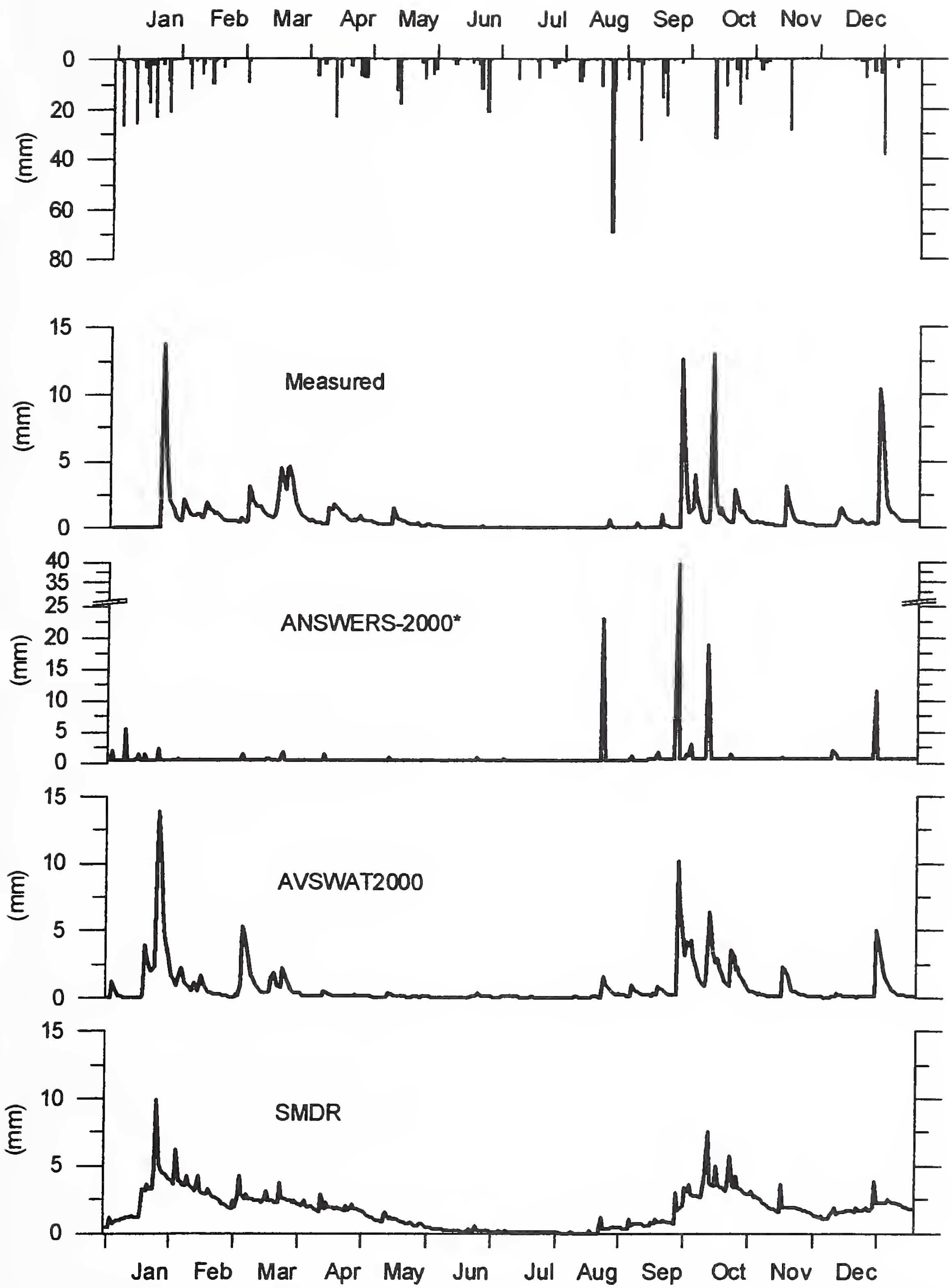
Figure 3 shows observed and simulated stream flow hydrographs during 1999. Of the three models considered, AVSWAT2000 stream flow values agreed most strongly with the observed flows ($r^2 = 0.67$, Nash-Sutcliffe = 0.66). In ANSWERS-2000, the effect of a storm event on stream flow does not last beyond the storm day. Steep falling limbs of ANSWERS-2000 hydrographs are indicative of quick conversion, routing, and cell outflow of surface runoff during storm events. Considering only storm events and adjusting for concurrent measured base flow, ANSWERS-2000 matched weakly with observed values ($r^2 = 0.27$, Nash-Sutcliffe = -1.26). SMDR-simulated stream flows also correlated weakly with observed values ($r^2 = 0.33$, Nash-Sutcliffe = 0.03). Absence of an infiltration excess runoff component in SMDR was noticeable during large storm events. While SMDR did not match observed peak flows during storm events, it produced larger than observed flows on days following storm events. This resulted in a mismatch of base flows and base flow recession curves as compared to measured data. Observed and SMDR-simulated storm flow volumes were comparable but flow timings were not.

Below-freezing temperatures and significant snowfall were observed during the early part of 1999. AVSWAT2000 best matched the observed stream flow responses during this period. ANSWERS-2000 treats all forms of precipitation as rain. Immediate conversion of snowfall to stream flow and absence of snow pack in ANSWERS-2000 affected soil moisture conditions and storm flow simulations during winter months and the warm period immediately following. The slow hydrograph recession of SMDR-simulated stream flow during winter supports conclusions by Srinivasan et al. (2003) that snowmelt routines in SMDR need further refinement.

Dry weather conditions from April to August resulted in very low stream flows. All three models simulated these low flow conditions. A large storm event (69 mm) ended this dry spell but resulted in less than 1% conversion of rainfall to runoff. Both AVSWAT2000 and SMDR produced low flows similar to the observed for this event; ANSWERS-2000 converted more than 50% of the rainfall to runoff. A similar situation occurred for a September event. These high over-predictions of storm flow by ANSWERS-2000 could be due to poor tracking of soil moisture conditions during dry periods or shortness of simulated storm duration.

Spatial output

Figure 4 presents annual runoff generated by each cell (ANSWERS-2000 and SMDR) or HRU (AVSWAT2000) in FD-36 during 1999. ANSWERS-2000 routes surface runoff from cell to cell. This interaction between cells enables infiltration or accumulation of runoff, depending on downstream conditions. For example, runoff levels remained low over the pasture in the center of the watershed (Figure 4), indicating runoff entering from surrounding cropland infiltrated into the pasture. In contrast, runoff through the forest on the southern side of the watershed accumulated as it followed the flow path north through forest and cropland to the stream. ANSWERS-2000 generally simulated high runoff depths where croplands occur on fragipan soils. However, these areas are also near-stream areas. It is unclear, using the output formats currently available, whether ANSWERS-2000 treated the near-stream areas as hydrologically more active than other areas, as simply transporting upland accumulation, or as some combination. Clarification of this issue is advisable



* Note: ANSWERS-2000 does not simulate base flow. For comparison purposes, base flow has been added here.

Figure 3. Observed and simulated stream flow depths from the FD-36 watershed during 1999.

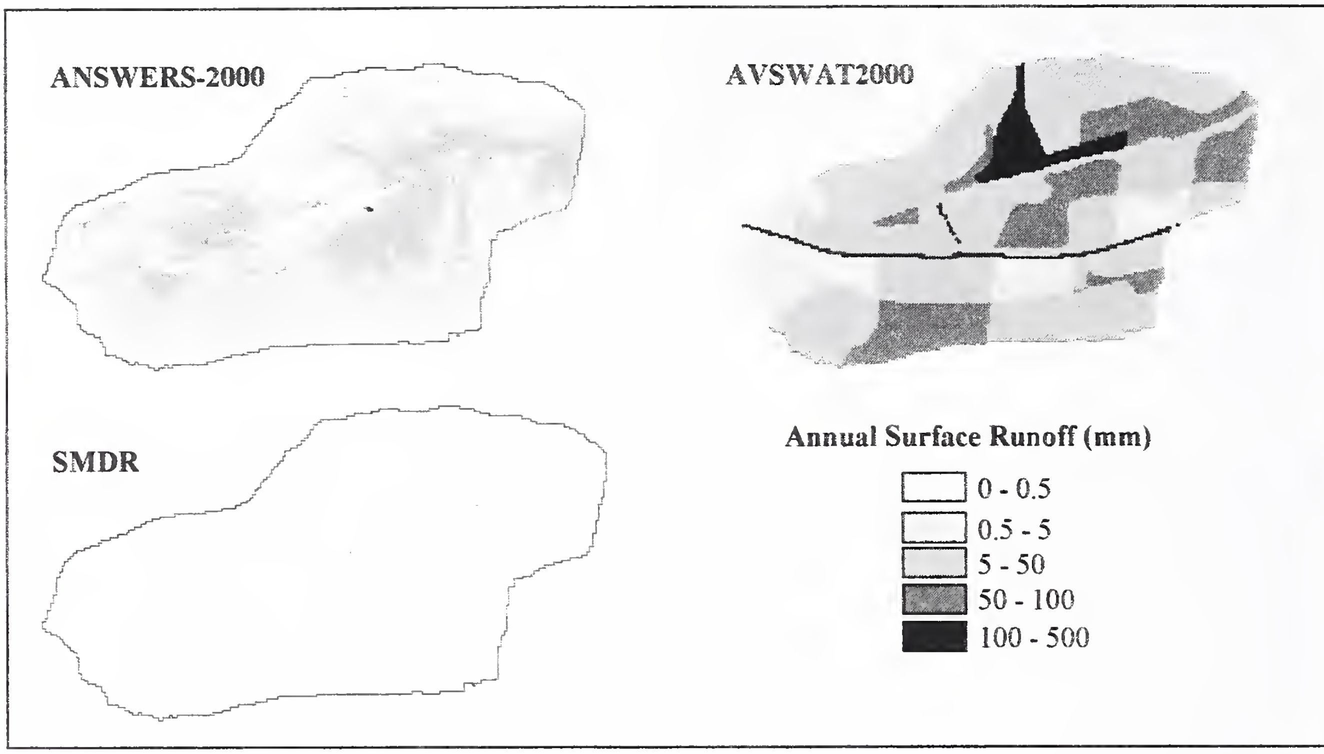


Figure 4. Spatial depictions of hydrological response during 1999: total surface runoff from each spatial unit.

before looking at pollution transport associated with this routing procedure.

AVSWAT2000 routes both surface and subsurface flow directly from HRU to stream, with no interaction between HRUs. Thus, an upland HRU may contribute more flow per unit area to a stream than a near-stream HRU (Figure 4). For example, from within each field AVSWAT2000 generated more surface runoff from some soils than others. Overall, in AVSSWAT2000, cropland produced the most runoff and forest the least, as might be expected. Particularly high runoff depths were seen on roads and on one field remaining in wheat stubble for the latter half of the year.

SMDR is appropriate for small watersheds, such as FD-36, where surface routing periods are less than a day. SMDR produces surface runoff only after a cell becomes saturated. Upon saturation, excess water is moved directly from the cell to the watershed outlet as surface runoff, with no surface interaction among cells. At the subsurface level, SMDR routes water from cell to cell at a rate of one cell per day. By routing subsurface flow from upland to near-stream cells in between storm events, SMDR causes near-stream cells to remain relatively wetter than upland cells. Due to simulating greater soil water storage than actually available in FD-36, SMDR did not produce the volumes of surface runoff that ANSWERS-2000 and AVSWAT2000 did (Figure 4). SMDR did identify near-stream areas as hydrologically more active than upland areas (data not shown). Including infiltration excess mechanisms may improve SMDR's performance.

Conclusions

This study assessed the ability of three models to depict spatial and temporal processes of a small, agricultural watershed with fragipan soils. All three models captured most major temporal variations seen in total surface runoff from the watershed in 1999; AVSWAT2000 achieved the strongest temporal statistical correlation. In contrast, spatial identification of runoff generation areas varied distinctly among the three models. Unlike SMDR, AVSWAT2000 and ANSWERS-2000 recognized differences in land use and soil characteristics within the watershed. This recognition is critical for making proper management recommendations. ANSWERS-2000 and, to a lesser extent, AVSWAT2000 depicted higher runoff depths from the near-stream, fragipan soils than from other areas. Differences were also seen in the ranges of simulated runoff depths. AVSWAT2000 produced as much as 100 mm of runoff per HRU while SMDR surface runoff depths did not exceed 5 mm over a 5-m grid cell.

ANSWERS-2000 runoff values ranged between 0 and 100 mm per 5-m grid cell, representing upstream flow accumulation from watershed boundary.

Although AVSWAT2000 uses less detailed process representations than ANSWERS-2000 or SMDR, AVSWAT2000 was chosen out of the three models as most accurately depicting the hydrological processes of the FD-36 watershed. Nevertheless, spatial distribution of runoff generation areas in AVSWAT2000 may need further analysis and refinement.

The success of AVSWAT2000 is likely due to a combination of factors. For example, AVSWAT2000 includes snowmelt and subsurface flow components not present in ANSWERS-2000 and, unlike SMDR, a mechanism for estimating infiltration excess. Also, AVSWAT2000 results may have benefited by the ability to calibrate the model specifically for characteristics of FD-36; unique aspects of this watershed's physical processes may not be adequately represented by the two physically-based models considered.

This study has improved understanding of how models with different temporal and spatial process representations simulate the characteristics of FD-36. By accurately modeling hydrologic response in this type of watershed, future efforts in modeling pollutant source and transport can build on a solid foundation. This work is an important step in developing and evaluating management techniques for water quality protection and improvement in small agricultural watersheds with fragipan soils.

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A Blueprint for an Integrated Watershed Hydrogeomorphic Modeling System

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Abstract

The hydrology of a river basin interacts with its geomorphic form. Despite significant advances in hydrological modeling, the dynamic interactions between geomorphic, hydrologic and ecological processes are not adequately captured at the basin scale. There is a need for advanced modeling tools that address both runoff and erosion prediction in sufficient spatial and temporal detail for exploring process couplings, feedbacks and complexity.

In this paper we present a blueprint for integrating the Channel-Hillslope Integrated Landscape Development (CHILD) and TIN-based Real-time Integrated Basin Simulator (tRIBS) models in a hydro-eco-geomorphic framework where process interactions and feedbacks are characterized. The integration of these models opens new avenues to explore the topographic, vegetative and climatic controls on watershed process interactions. Here, we present the capabilities of the two models and discuss a strategy for model integration. We also provide an example of the erosion effect on thunderstorm runoff response in a semi-arid basin as a proof-of-concept for model integration.

Keywords: distributed modeling, watershed, hydrology, erosion, vegetation, tRIBS, CHILD

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Introduction

Catchment hydrology depends on the geomorphic form of the basin and the hillslope, channel and ecological processes active at different spatial and temporal scales (Osterkamp and Toy 1997, Tucker and Bras 1998). The long-term river basin evolution is linked with the underlying mechanisms of hydrologic response, while the short-term storm response depends on topographic variability resulting from erosional processes. For example, gully erosion may increase drainage density, affect runoff generation and contribute to the formation of rapid runoff hydrographs (Ritter and Gardner 1993). Similarly, erosional perturbations can impact soil moisture distribution and vegetation growth, which can have important implications on hydrology.

Numerical models typically treat hydrologic and geomorphic processes independently due to differences in response time scale. Recent process-based erosion models, such as KINEROS (Woolhiser et al. 1990), EROSION 3D (Schmidt et al. 1999) and CASC2D (Ogden and Heiling 2001) simulate hydrology and use flow predictions for sediment transport. We think this approach is valid for "geomorphic equilibrium" when an approximate balance exists between soil production and removal by erosion. This period of equilibrium is often interrupted by external factors such as extreme climate events, disturbances and land-use change or by internal factors such as the exceedence of process thresholds (Schumm 1980). Erosional response to such changes can be quite rapid and have large impacts on the catchment hydrologic response.

The objective of this paper is to present and illustrate a hydrogeomorphic framework for continuous runoff and erosion modeling where interactions and feedbacks between hydrologic response and geomorphic form are explicitly characterized. We focus on the fluvial erosion of a small catchment in the Walnut Gulch Experimental Watershed. A disturbance regime due to climate variability and/or vegetation shifts is imposed on the current topography to induce erosional response in the channel network (i.e. propagation of an erosion wave) and on hillslopes due to rill incision. Short-term hydrologic response to summer thunderstorm events is then analyzed in the context of the evolving landscape. It is hypothesized that fluvial incision leads to increases in the drainage density that causes an intensification of the runoff response.

In the following sections, we discuss the framework for coupling the two distributed models for landscape evolution (CHILD) and continuous runoff prediction (tRIBS). The models currently share a computational platform and basin partitioning using a triangulated irregular network (TIN), but have yet to be utilized in a conjunctive fashion. This study is a preliminary investigation into the coupled use of the two modeling tools. In so doing, we outline a "blueprint" for model integration and illustrate its potential by addressing the effect of erosion on surface hydrologic response within a well-instrumented research catchment.

Integrated Hydrogeomorphic Modeling

Figure 1 presents the hydrogeomorphic processes represented in the integrated model. Basin storm and interstorm periods force an erosional, hydrologic and vegetative response which are both defined by and have an influence on the basin geomorphic and landuse properties. In the following, we discuss the elements of the model blueprint and their integration.

Triangulated model framework

The TIN data structure utilized in the two models is a surface representation defined by triangular elements of varying size. Tucker et al. (2001a) describes the data structure and geometry of the irregular mesh. Various factors motivate their use in basin modeling. The primary advantage is the variable resolution obtained through the irregular spacing of elevation nodes, which translates to computational savings. A second advantage is that TINs permit streams and boundaries to be preserved. In addition to the TIN, a Voronoi mesh is associated with the model domain. A unique set of Voronoi polygons is created by intersecting the perpendicular bisectors of each triangle edge (Tucker et al. 2001a). These irregular polygons surround a TIN node and form the basis for finite-volume computations in the two models.

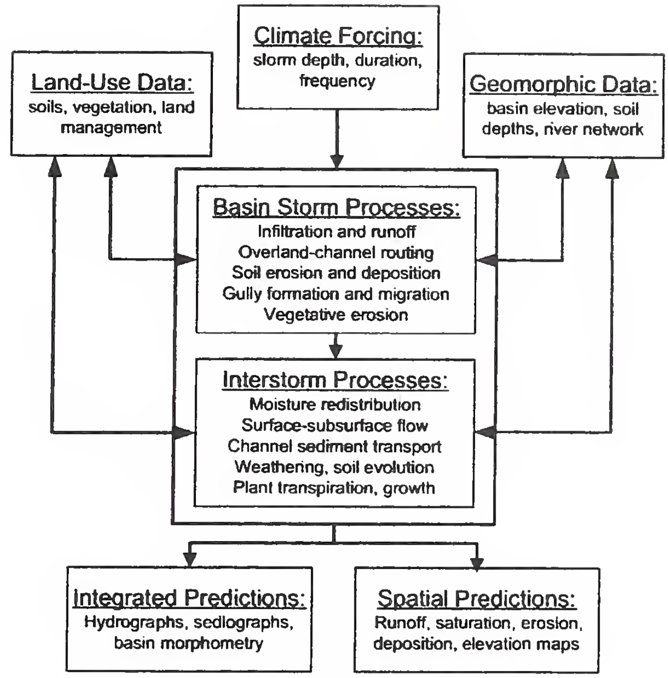

Figure 1. Integrated hydrogeomorphic model blueprint.

Landscape evolution: The CHILD model

The CHILD model simulates basin-scale changes in landscape morphology resulting from tectonic, channel and hillslope processes. The model incorporates climate forcing, steady-state runoff production, fluvial erosion and deposition, soil evolution due to soil creep and bedrock weathering, floodplain deposition, meander evolution, and a simple vegetation model (Tucker and Bras 2001, Tucker et al. 2001b). In the fluvial erosion component used here, the rate of change of elevation (z) is the difference between tectonic displacement (U) from either uplift or base level lowering and local erosion (E), as:

$$\frac{\partial z}{\partial t} = U - E = U - \begin{cases} \nabla \bullet Q_s \\ E_c \end{cases} \qquad (1)$$

where Q_s is the sediment transport rate, E_c is an erosion capacity and ∇ is the divergence operator. The erosion term can be modeled using two general types of erosion postulates: transport or detachment limited. The first postulate (the upper term in 1) dictates the conservation of mass, and erosion is represented as the difference between sediment outflux and influx. In the latter erosion law, the rate of local incision is assumed to depend on local shear stress or stream power.

The reader is referred to Tucker et al. (2001a, 2001b) for details on the various geomorphic processes simulated in CHILD. Currently, additional modules that simulate shallow landsliding, debris flow transport and gully erosion processes involving plunge-pool erosion, bank failures and sapping are being added to the model.

Distributed hydrology: The tRIBS model

tRIBS is a physically-based distributed hydrologic model designed for real-time, continuous forecasting. Over the TIN terrain, the model uses rainfall and meteorological forcing to predict basin response, including infiltration, runoff, evapotranspiration, soil moisture and aquifer recharge. Ivanov et al. (in review) provide detailed descriptions of the model, including the parameterizations for coupled unsaturated-saturated dynamics, runoff production and routing, and moisture redistribution through the hillslope system.

At the heart of the tRIBS model is a simplified, coupled system of equations leading to infiltration, lateral moisture fluxes and groundwater recharge. The sloped, heterogeneous, anisotropic soil column is characterized by a vertical decay in hydraulic conductivity (K_n):

$$K_n(z) = K_s e^{-fz} \tag{2}$$

where f is the conductivity decay parameter. Horizontal conductivity (K_p) is accounted for via an anisotropy ratio ($a_r = K_n/ K_p$). Ponded and unsaturated infiltration are taken into account. The interaction between propagating moisture fronts, water table fluctuations and lateral moisture exchanges leads to various runoff mechanisms produced within each model element:

$$R = R_I + R_S + R_P + R_G \tag{3}$$

where R_I is infiltration-excess runoff, R_S is saturation-excess runoff, R_P is perched return flow and R_G is groundwater exfiltration. The water table position determines the hydrostatic soil moisture profile in the unsaturated zone, which in turn controls the runoff partitioning. Routing of surface flow is achieved via overland and channel pathways through a hydrologic hillslope and hydraulic channel routines. The channel geometry is derived from a geomorphic relation:

$$W = \alpha A^{\beta} \tag{4}$$

where W is the channel width, A is the contributing upslope area, and α and β are parameters (Table 1).

Blueprint for model integration

Our strategy for integrating CHILD and tRIBS consists of loosely coupling the models through common landscape variables to simulate basin processes at both short (years to decades) and long term (up to millennia) time scales (Figure 1). Both models share the same TIN architecture and C++ object-oriented code structure. However, because the models focus on distinct basin processes operating over different times, they are executed at different time steps. For example, the CHILD time step is based on the duration of erosive storms (days – years), whereas tRIBS utilizes shorter time steps based on rapid unsaturated-saturated fluxes (minutes – hour). If tightly coupled by resolving all processes at a high temporal resolution, the resulting model would require a high computational demand. To increase efficiency, our strategy is to couple the two models through an adaptive time sequencing scheme for both short and long-term simulations.

For short-term simulations, we will incorporate the existing sediment transport algorithms in CHILD into the tRIBS model. This will permit hydrograph and sediograph predictions within the basin. To explore long term changes, such as climate variability or vegetation shifts, we will loosely couple the models by interchanging landscape state variables, which will translate the effects of geomorphic dynamics onto basin hydrology. We identify the important state variables as: topography, drainage density, soil depth, vegetation pattern and type, and stratigraphy. Over decades or centuries, CHILD could evolve these states through various geomorphic processes, while tRIBS could use these predictions to explore the hydrologic implications of basin evolution. This "magnifying" approach would add to our understanding of landscape process interactions and feedbacks across various time scales.

Illustrative Example

Here we present an example of the magnifying approach to coupled hydrogeomorphic modeling in a real world watershed. We impose fluvial erosion on an existing topography and explore the hydrologic consequences as an erosion wave propagates to the headwater channels. In the following, we describe the study area and the design of our numerical experiment, followed by the hydrogeomorphic simulations results.

Study area

To illustrate the coupled prediction of fluvial erosion and runoff, we selected a 2.27-km^2 subwatershed of the 149-km^2 USDA-ARS Walnut Gulch Experimental Watershed located in southeast Arizona (Figure 2). The basin topography, obtained from a 28.6-m USGS digital elevation model (DEM), is represented through a high-resolution triangulated irregular network. The hydrographic TIN model (3,600 nodes) captures terrain variability, and explicitly represents the basin boundary and embedded stream network (Vivoni et al. 2003).

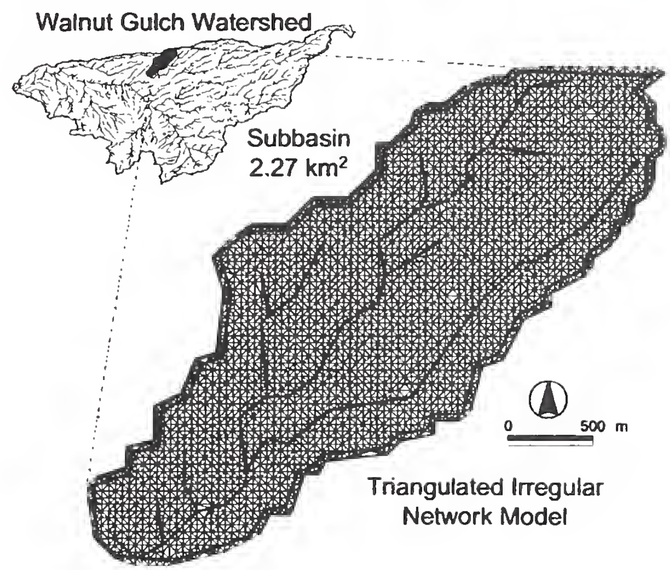

Figure 2. Initial TIN terrain model for study area with basin boundary and channel network.

The semi-arid basin is characterized by desert shrub vegetation and very gravelly, sandy loam soils (Houser et al. 2000). High-intensity, short-duration convective rainfall from summer thunderstorms produces the majority of the observed runoff measured at the outlet via a flume (W-4) (Syed et al. 2003). Infiltration-excess runoff is the primary mechanism due to the deep water table position and low soil infiltration capacity. Given the small basin area, spatially-uniform surface and rainfall conditions are assumed here. Only the spatial distribution of the evolving landscape topography impacts the runoff and erosion response in the catchment. These assumptions can be relaxed in the current framework, as shown by Ivanov et al. (in review). Table 1 lists the model parameters obtained from field and modeling studies (Grayson et al. 1992, Goodrich et al. 1997, Miller et al. 1996, Hernández et al. 2000).

Table 1. Uniform basin parameter values.

Parameter	Value
Saturated conductivity (K_s)	8.5 mm/hr
Soil porosity (ε)	0.45
Saturation soil moisture (θ_s)	0.41
Residual soil moisture (θ_r)	0.04
Conductivity decay (f)	0.007 mm^{-1}
Anisotropy ratio (a_r)	100
Vegetation cover (v)	0.1
Manning roughness (n)	0.3
Geomorphic width coefficient (α)	7.24
Geomorphic width exponent (β)	0.34
Erodibility (k_e)	3.7×10^{-4} ($kgs^{-1}m^{-2}Pa^{-p}$)
Excess shear stress exponent (p)	2.3
Storm duration (t_d)	1 hr
Rainfall intensity (i)	100 mm/hr
Interstorm duration (t_r)	60 hr

Experimental design

To demonstrate the potential for coupled modeling, we simulate the geomorphic processes associated with fluvial erosion and then its impact on watershed hydrologic response. In this study, CHILD and tRIBS are loosely tied via a common modeling framework and exchange of topographic data. The USGS DEM is utilized as an initial terrain condition. Operating over a long sequence of storm-interstorm periods, the hillslope and channel processes in CHILD lead to erosion throughout the basin. The initial topographic condition and a final instance of landscape evolution after 100 storms are then ingested into the tRIBS model to produce comparative runoff responses. The evolving geomorphic form will impart different signatures on the basin hydrograph and runoff spatial pattern.

Erosion

Fluvial erosion is modeled using (1) based on detachment limited erosion (Tucker et al. 2001b). The erosion rate is related to shear stress (τ) acting on soil grains in excess of a threshold (τ_c) as:

$$E_c = k_e(\tau - \tau_c)^p, \quad \tau = k_t q^{0.6} S^{0.7} > \tau_c \qquad (5)$$

where k_e is soil erodibility, and k_t relates shear stress to discharge (q) and slope (S) and is a function of soil roughness and channel shape (Istanbulluoglu et al. 2003). We obtained a value for k_t using the current topography of the study basin. Geomorphology literature suggests that unless an external disturbance is imposed, erosion is due to runoff rates (R) which recur every two years and at least once in five years (Wolman and Miller 1960).

Based on our hydrology simulations described next, a 2-year runoff rate, R has a magnitude of 70 mm/hr in this basin. Making the steady-state assumption of $q = R*a$, where a is the specific catchment area, and using $\tau_c = 5$ Pa (sediment diameter of 6 mm), a k_t value of ~500 produced limited erosion around the basin outlet. To simulate erosion due to external factors, we increased k_t by a factor of 5. Such an increase mimics a full scale disturbance of vegetation and/or an implicit increase in runoff rate due to higher rainfall (i.e. climate change). For comparison, Istanbulluoglu et al. (2002) found a factor of 3 difference in k_t between a forested and a partially burned basin in Idaho.

We used values for k_e and p in (5) (Table 1) reported by Nearing et al. (1999) for a site close to the study basin. Given these parameter values, we found that the selected runoff rate (R = 70 mm/hr) results in erosion that occupies the entire catchment, from outlet to headwaters, within a series of 100 thunderstorms (see next section). Within this period, CHILD simulates an erosion wave commencing in the basin lowlands and traveling upstream. In so doing, the propagating wave influences channel elevations throughout the network and leads to adjustments in the hillslope profiles. Upon reaching all basin locations within the specified erosion threshold, a stable state is reached in the elevation field. We test the hydrologic sensitivity of the initial and final landscapes in the following (Figure 3).

Thunderstorm runoff

Thunderstorm rainfall is modeled as a series of discrete random events separated by interstorm periods. The storm sequence is generated by sampling an exponential distribution constructed for the storm duration (t_d), intensity (i) and interstorm duration (t_r). Table 1 lists the storm parameters used for summer conditions in the basin (Chagnon 1998). Due to high rain rates over the gravelly sandy loam soil, infiltration excess runoff is the primary runoff mechanism. Prior to each simulation, the water table depth is set to a deep uniform value that determines the initial moisture profile (Ivanov et al. in review). A spin-up period of several months is allowed to minimize initialization errors. Subsequently, a one-month rainfall-runoff simulation is conducted. No plant interception or evapotranspiration are simulated as we concentrate on the runoff response to intense storms capable of eroding the land-surface.

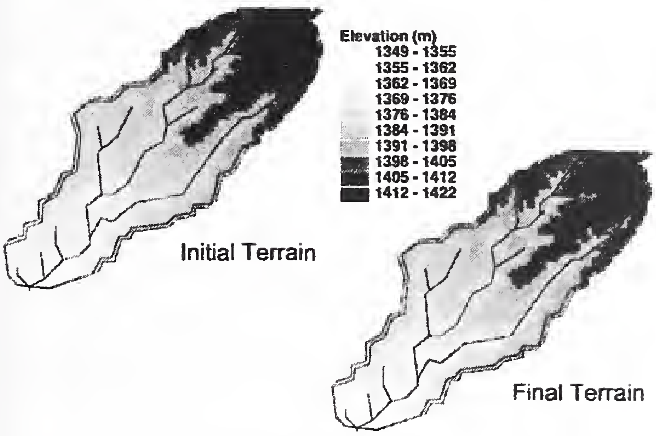

Figure 3. Evolving landscape elevation at initial and final time periods.

Figure 4 presents a runoff hydrograph comparison for the terrain models for two storm events. These cases represent two interesting scenarios. In Day 16, two individual, closely-spaced, storms produced a dual peak runoff response. In Day 25, a single peak event of large magnitude occurring after a long interstorm period produced a significant flood event at the basin outlet. Note that the runoff response to the same applied rainfall varies for each terrain. As the erosion wave advances, the flood peak increases in magnitude and decreases in response time, suggesting higher basin sensitivity to rainfall. Fluvial erosion leads to higher slopes along the hillslope-channel interface that affect the hydrograph. The overland flow velocity increases as well since it is tied to the channel discharge in the nearest stream link in the model. As erosion progress into hillslope regions, increased slopes also lead to higher lateral transport of the unsaturated zone moisture that also accelerates basin response.

While the hydrographs are indicative of the integrated catchment response, they do not provide information on the spatial distribution of runoff generation. Figure 5 presents a means for capturing these spatial patterns. Here, the temporal frequency of infiltration-excess runoff (R_I) (i.e. fraction total time), the primary mechanism occurring in the basin, is shown for the initial terrain model. Note that runoff is produced frequently within the channel network and along hillslope hollows. This is due both to rainfall rates exceeding the normal hydraulic conductivity (K_n) and lateral moisture transport in the unsaturated zone, which occurs due to the high anisotropy ratio (a_r). This preferential lateral flow in the absence of a high water table is plausible in soil horizons characterized by an exponential decrease in K_n (e.g. Beven 1982).

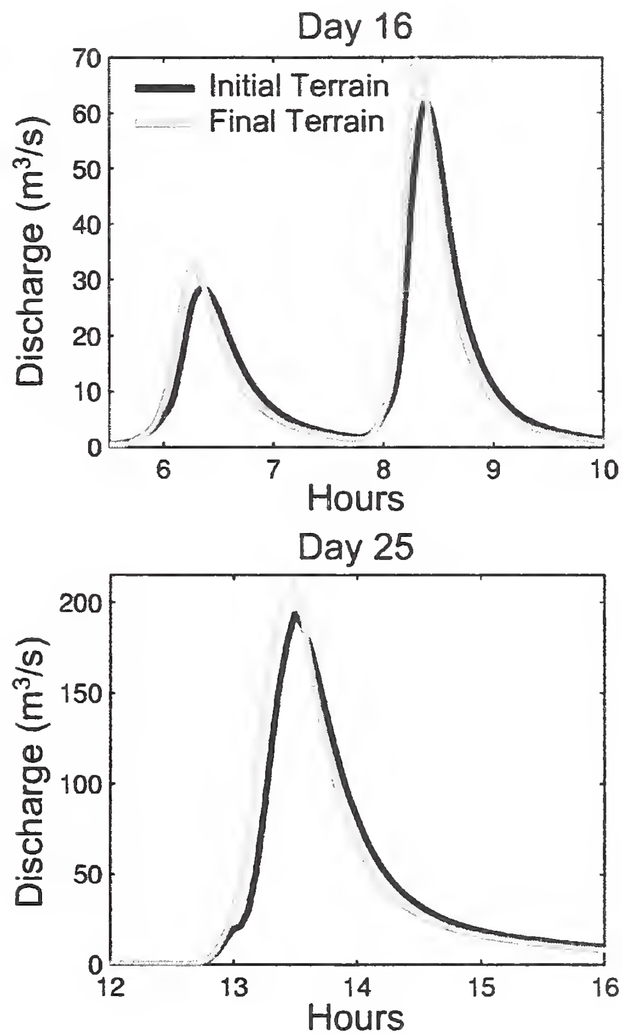

Figure 4. Runoff hydrograph response at the outlet ($2.27\ km^2$) for two selected events (days 16, 25).

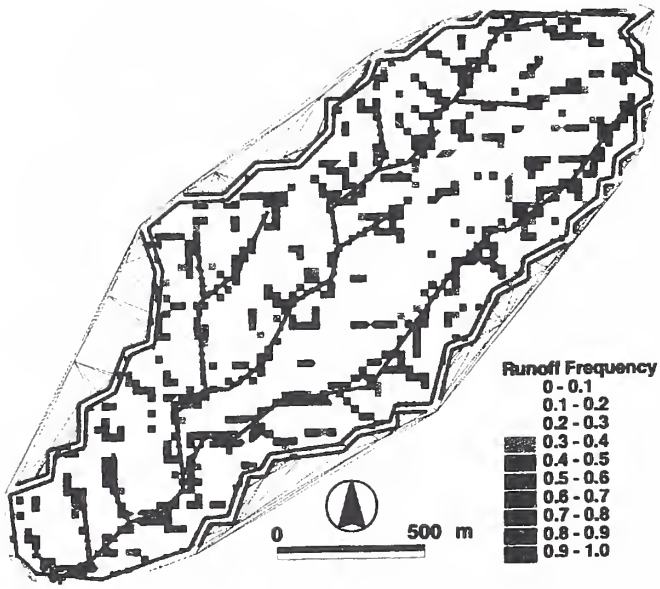

Figure 5. Temporal frequency of infiltration-excess runoff (R_I) for the initial topographic model.

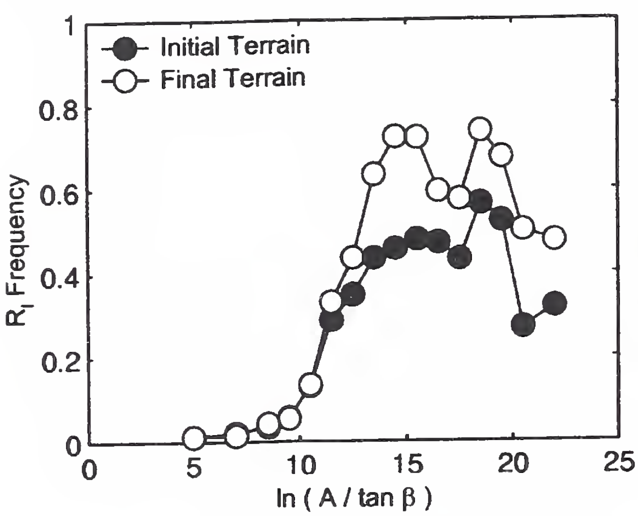

Figure 6. Topographic location of infiltration-excess runoff (R_I) frequency generated over simulation period.

The spatial organization of infiltration-excess runoff is captured in Figure 6 through a topographic analysis of the runoff production regions (Figure 5). Here, we utilize the topographic or wetness index (Beven and Kirkby 1979) to classify basin Voronoi polygons into hydrologically-similar units. The topographic index,

$$\lambda = \ln(A / \tan \beta) \tag{6}$$

where A is the upslope contributing area and $\tan \beta$ is the surface slope, illustrates the regions preferentially producing infiltration-excess runoff for each landscape model. Larger values of λ correspond to the flat channel areas, while smaller λ values are attributed to steep hillslope regions. Note that the frequency of runoff production increases between values $\lambda = 12$ to 17 as fluvial erosion shapes the basin geomorphic form. Figure 6 highlights the sharp differences in the runoff response in the basin as erosional processes adjust the landscape to the imposed surface disturbance.

In Figure 7, the effect of fluvial erosion on the surface moisture distribution is investigated for a selected transect in the upper basin reaches. Erosion impacts the elevation profile along the transect, as well as the soil moisture distribution. Note that the initial terrain has a broad channel region with a slightly higher moisture content in the low-lying area. As fluvial erosion incises and deepens the channel in the final terrain, a sharp difference in moisture content is exhibited between the channel and surrounding hillslope regions.

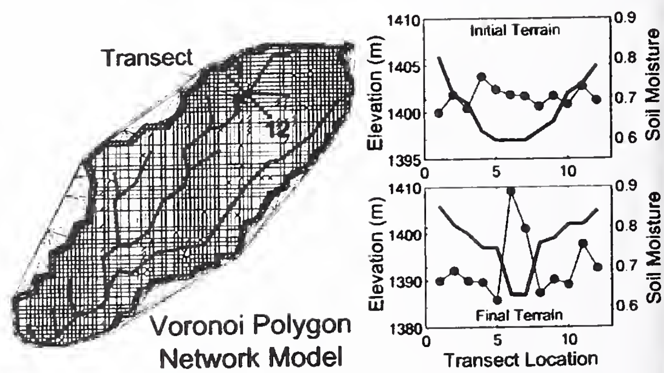

Figure 7. Transect distribution of elevation (solid line) and soil moisture (closed circles).

Conclusions

In this study, we have demonstrated the coupled use of a landscape evolution and a distributed hydrologic model for investigating the impact of fluvial erosion on basin hydrograph and spatial runoff response. Erosion processes are shown to increase the runoff intensity and decrease the response time to rainfall forcing. The application to a Walnut Gulch sub-basin illustrates the capabilities and potential of the proposed integrated framework for understanding the process interaction between basin hydrology and geomorphic form.

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Recent Progress in the Development of a SPARROW Model of Sediment for the Conterminous U.S.

Gregory Schwarz, Richard Smith, Richard Alexander, John Gray

Abstract

Suspended sediment has long been recognized as an important contaminant affecting water resources. Besides its direct role in determining water clarity, bridge scour and reservoir storage, sediment serves as a vehicle for the transport of many binding contaminants, including nutrients, trace metals, semi-volatile organic compounds, and numerous pesticides (U.S. Environmental Protection Agency 2000a). Recent efforts to address water quality concerns through the TMDL process have identified sediment as the single most prevalent cause of impairment in the Nation's streams and rivers (U.S. Environmental Protection Agency 2000b). Moreover, sediment has been identified as a medium for the transport and sequestration of organic carbon, playing a potentially important role in understanding sources and sinks in the global carbon budget (Stallard 1998).

Keywords: sediment, model, statistics, yield

Introduction

A comprehensive understanding of sediment fate and transport is considered essential to the design and implementation of effective plans for sediment management (Osterkamp et al. 1998, U.S. General Accounting Office 1990). An extensive literature addressing the problem of quantifying sediment transport has produced a number of methods for estimating its flux (Cohn 1995, Robertson and Roerish 1999). The accuracy of these methods is compromised by uncertainty in the concentration measurements and by the highly episodic nature of sediment movements, particularly when the methods are applied to smaller basins. However, for annual or decadal flux estimates, the methods are generally reliable if calibrated with extended periods of data (Robertson and Roerish 1999). A substantial literature also supports the Universal Soil Loss Equation (USLE) (Natural Resources Conservation Service 1983), an engineering method for estimating sheet and rill erosion, although the empirical credentials of the USLE have recently been questioned (Trimble and Crosson 2000). Conversely, relatively little direct evidence is available concerning the fate of sediment. The common practice of quantifying sediment fate with a sediment delivery ratio, estimated from a simple empirical relation with upstream basin area, does not articulate the relative importance of individual storage sites within a basin (Wolman 1977). Rates of sediment deposition in reservoirs and floodplains can be determined from empirical measurement, but only a limited number of sites have been monitored and net rates of deposition or loss from other potential sinks and sources is largely unknown (Stallard 1998). In particular, little is known about how much sediment loss from fields ultimately makes its way to stream channels and how much sediment is subsequently stored in or lost from the stream bed (Meade and Parker 1985, Trimble and Crosson 2000).

This paper reports on recent progress made to empirically address the question of sediment fate and transport on a national scale. The model presented here is based on the SPAtially Referenced Regression On Watershed attributes (SPARROW) methodology, first used to estimate the distribution of nutrients in streams and rivers of the US, and subsequently shown to describe land and stream processes affecting the delivery of nutrients (Smith et al. 1997,

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Alexander et al. 2000, Preston and Brakebill 1999). The model makes use of numerous spatial data sets, available at the national level, to explain long-term sediment water quality conditions in major streams and rivers throughout the US. Sediment sources are identified using sediment erosion rates from the National Resources Inventory (NRI) (Natural Resources Conservation Service 2000) and apportioned over the landscape according to 30-meter resolution land use information from the National Land Cover Data set (NLCD) (U.S. Geological Survey 2000a). Over 76,000 reservoirs from the National Inventory of Dams (NID) (U.S. Army Corps of Engineers 1996) are identified as potential sediment sinks. Other, non-anthropogenic sources and sinks are identified using soil information from the State Soil Survey Geographic (STATSGO) database (Schwarz and Alexander 1995) and spatial coverages representing surficial rock type and vegetative cover. The SPARROW model empirically relates these diverse spatial data sets to estimates of long-term, mean annual sediment flux computed from concentration and flow measurements collected over the period 1985-95 from more than 400 monitoring stations maintained by National Stream Quality Accounting Network (NASQAN) (Alexander et al. 1998), the National Water Quality Assessment (NAWQA) Program, and U.S. Geological Survey District offices (Turcios and Gray 2001). The calibrated model is used to estimate sediment flux for over 60,000 stream segments included in the River Reach File 1 (RF1) stream network (Alexander et al. 1999).

SPARROW uses statistical methods to calibrate a simple, structural model of riverine water quality, one that imposes mass balance in accounting for changes in contaminant flux. As applied here, the mass-balance approach facilitates the interpretation of model results in terms of physical processes affecting sediment transport, and makes possible the estimation of various rates of sediment generation and loss associated with stream channels and features of the landscape. The statistical approach provides a basis for assessing the error of these inferred rates, and of the error in extrapolated estimates of sediment flux made for streams in the RF1 network.

An important implication of the holistic modeling approach adopted in this analysis is that estimates of sediment production and loss are based on, and therefore consistent with, measurements of in-stream flux. Other ancillary information, such as direct measurements of long-term sediment storage and release from reservoirs (Steffen 1996), are incorporated into the analysis by specifying additional equations explaining these ancillary variables. The imposition of cross-equation constraints affords this information a statistically consistent weight in explaining in-stream sediment flux. Thus, the methodology described here represents a general framework for synthesizing a wide spectrum of available information relevant to the understanding of sediment fate and transport.

Methods

The SPARROW methodology (Smith et al. 1997) has been modified to incorporate greater spatial resolution. The primary spatial reference frame for the model continues to be the RF1 reach network: all point sources and landscape features are referenced to a particular RF1 reach. However, considerable internal structure has been added to each reach. Reach watersheds are delineated using the 1-kilometer HYDRO 1K digital elevation model (DEM) (U.S. Geological Survey 2000) and explicit pathways are defined between landscape features and their adjacent RF1 streams. The delineation method uses a "burn-in" process whereby the RF1 reach is first digitized in the 1-kilometer grid and then the elevations of RF1 grid cells are artificially lowered to insure that simulated flow from surrounding cells moves into them. Flow directions based on the steepest descent determine the extent of the reach watershed and the undefined tributary flow paths leading from the landscape to the RF1 channel cells. To insure the accurate determination of in-stream travel time, RF1 stream pathways continue to be defined by the line work of RF1 channels rather than by the grid-cell representation.

A schematic of a typical reach watershed, illustrating its spatial structure and associated features, is given in Figure 1. Flow directions, represented by the arrows crossing each adjacent grid cell, define the movement of water in undefined tributaries leading to the RF1 stream. The "burn-in" method insures that all flow paths intersect a reach cell at some point within the watershed, although inconsistencies between the RF1 reach and the DEM-defined stream channel may artificially lengthen "off-RF1" flow paths and shorten "on-RF1" paths (see Figure 1 for an example). The length of the flow path provides a rough estimate of the distance sediment must travel in smaller tributaries before reaching the larger streams included in the RF1 network. Travel time in small streams versus large rivers has been shown to be an important factor affecting the in-stream delivery of nutrients (Alexander et al. 2000) and could be of similar importance for sediment.

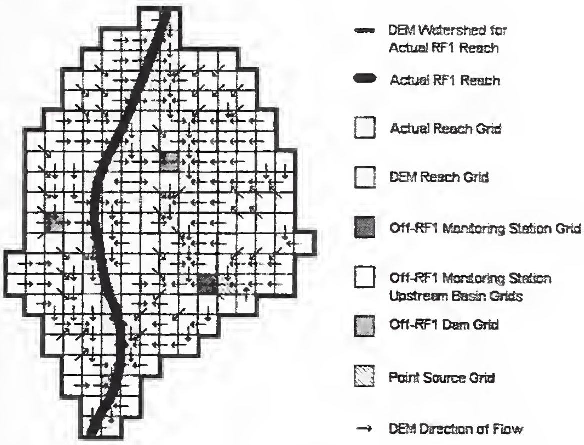

Figure 1. Schematic of a typical reach watershed illustrating the grid cell structure and identified attributes.

The enhanced spatial structure afforded by the DEM facilitates the incorporation of spatially integrating features into the model. "Off-channel" reservoirs, located on the grid net according to their geographic coordinates provided by the NID, act as potential sinks for sediment emanating from cells with flow paths that intersect the reservoir grid cell. Similarly, "off-RF1" monitoring stations can be located on the grid and given a basin representation. Although these stations are not useful for calibrating the delivery process within RF1 channels, they offer a high-resolution view of other processes affecting the movement of sediment across the landscape.

Other important spatial features identified in the model include point sources, located relative to RF1 streams based on geographic coordinates (Rubin 1999), and land associated with uses that serve as likely sources or sinks for sediment. Point source total suspended solids loadings are determined by methods developed by the National Oceanic and Atmospheric Administration (NOAA) for the National Coastal Pollutant Discharge Inventory (NOAA 1993). Land use is taken from the 21-class, 30-meter resolution NLCD, and summarized according to the number of 30-meter cells of a given land use class that are mapped to a corresponding 1-kilometer cell. NLCD land use is used to refine the areal extent of the various sediment erosion rates associated with different land covers identified in the NRI.

The mean annual suspended-sediment flux generated within and leaving reach watershed j, referred to as the incremental reach flux F_j, can be expressed as

$$F_j = \sum_{c=1}^{N_j} e^{-\boldsymbol{\delta}'\mathbf{d}_{c,j} + \boldsymbol{\alpha}'\mathbf{Z}_{c,j}} \boldsymbol{\beta}'\mathbf{S}_{cj} \qquad (1)$$

where N_j is the number of 1-kilometer grid cells, indexed by c, in reach watershed j, $\mathbf{d}_{cj}$ is a vector of factors describing the pathway from cell c to the outlet of reach j, $\boldsymbol{\delta}$ is a vector of coefficients associated with the pathway variables, $\mathbf{Z}_{cj}$ is a vector of landscape and climatic characteristics affecting the delivery of sediment within cell c, $\boldsymbol{\alpha}$ is a vector of coefficients associated with the $\mathbf{Z}$ variables, $\mathbf{S}_{cj}$ is a vector of sediment sources, and $\boldsymbol{\beta}$ is a vector of associated source coefficients.

The vector **d** consists of variables representing the landscape flow path distance traversed to reach the RF1 stream, the mean slope of the "off-RF1" flow path, the time of travel incurred along the RF1 stream, variables affecting the retention of sediment in any reservoir located along the landscape or RF1 flow path (such as streamflow, reservoir age, and NID estimates of surface area or storage volume), and other variables identifying possible sinks along the flow path such as forested land or land classified by STATSGO as wetlands or alluvium. Variables included in the **Z** vector include runoff, overland flow, slope and indicators of soils or other factors affecting the movement of sediment off the field to channels. The source vector, **S**, includes sediment erosion from the NRI and point source loadings.

The 1-kilometer spatial detail used to determine F_j, corresponding to nearly 8 million grid cells for the more than 60,000 reaches in the conterminous U.S., places a heavy computational burden on the iterative non-linear least squares calibration method. To reduce the number of computations, the reach model is simplified by assuming the **Z** variables take a single mean value $\overline{\mathbf{Z}}_j$ for all cells in the reach and, for the **d** variables, by substituting a second-order Taylor approximation about the reach-level mean $\overline{\mathbf{d}}_j$. The imposition of a common $\overline{\mathbf{Z}}_j$ value for all cells in a reach is not restrictive given the spatial coarseness of existing information. The resulting approximation is

$$F_j \approx e^{-\boldsymbol{\delta}'\overline{\mathbf{d}}_j+\boldsymbol{\alpha}'\overline{\mathbf{Z}}_j}\sum_{c=1}^{N_j}\Big\{\Big(1-\boldsymbol{\delta}'\big(\mathbf{d}_{c,j}-\overline{\mathbf{d}}_j\big)\Big)\mathbf{S}'_{c,j}\boldsymbol{\beta}+ (\boldsymbol{\beta}\otimes\boldsymbol{\delta})'\Big(\mathbf{S}_{c,j}\otimes\big(\mathbf{d}_{c,j}-\overline{\mathbf{d}}_j\big)\big(\mathbf{d}_{c,j}-\overline{\mathbf{d}}_j\big)'\Big)\boldsymbol{\delta}\Big\} \quad (2)$$

This approximation effectively converts the unit of observation in (1) from a 1-kilometer grid cell to a reach segment, replacing the non-linear terms dependent on individual cell values with non-linear and linear terms dependent on reach-level means, variances and covariances of the **d** and **S** variables.

To complete the model structure, individual reaches are combined to form a nested basin. Each nested basin i consists of the set $J(i)$ of reaches upstream from monitoring station i and below any monitoring station located further upstream (if such stations exist) (see Figure 2). The sediment load for nested basin i, denoted L_i, is equal to the sum of the incremental fluxes from the nested reach segments $j \in J(i)$, plus the monitored sediment discharged from the set $U(i)$ of nested basins bounding the upper drainage of nested basin i (there may be more than one) and delivered to monitoring station i. The sediment load L_i is related to the upstream incremental fluxes, F_j, and monitored loads, L_u, according to a log-linear relation

$$\ln(L_i) = \ln\left(\sum_{j\in J(i)} e^{-\boldsymbol{\delta}'\mathbf{d}_{j,i}} F_j + \sum_{u\in U(i)} e^{-\boldsymbol{\delta}'\mathbf{d}_{u,i}} L_u\right) + e_i \quad (3)$$

where $\mathbf{d}_{j,i}$ represents a vector consisting of the same variables in $\mathbf{d}_{c,j}$, but corresponding to the RF1-reach path extending from the downstream-end of reach j to the i^{th} monitoring station (accordingly, $\mathbf{d}_{j,i}$ has values of 0 for all variables pertaining to "off-RF1" flow paths). In (3), an independent error term e_i has been added to represent the combined effect of measurement and model error introduced at nested basin i.

Data on reservoir storage can be incorporated directly into the model by introducing an additional storage equation. Let $\mathbf{d}^*$ and $\boldsymbol{\delta}^*$ pertain to the subset of path variables determining the rate sediment is stored in reservoirs, and define R_k as the annual amount of stored sediment measured at a reservoir on reach k (a similar analysis can be done for "off-RF1" reservoirs). The reservoir storage equation takes the form

$$\ln(R_k) = \ln\left(\sum_{j\in J(k)}\left(e^{\boldsymbol{\delta}^{*\prime}\mathbf{d}^*_{j,k}}-1\right)F_j + \sum_{u\in U(k)}\left(e^{\boldsymbol{\delta}^{*\prime}\mathbf{d}^*_{u,k}}-1\right)L_u\right) + w_k \quad (4)$$

where w_k is a random error.

Joint estimation of (3) and (4), with the F_j and corresponding $\boldsymbol{\alpha}$, $\boldsymbol{\beta}$, and $\boldsymbol{\delta}$ parameters defined by (2), is by non-linear three-stage least squares. To insure robust estimates and to facilitate the estimation of prediction error, the calibration of the model is repeated 200 times employing a bootstrap estimation algorithm (Smith et al. 1997).

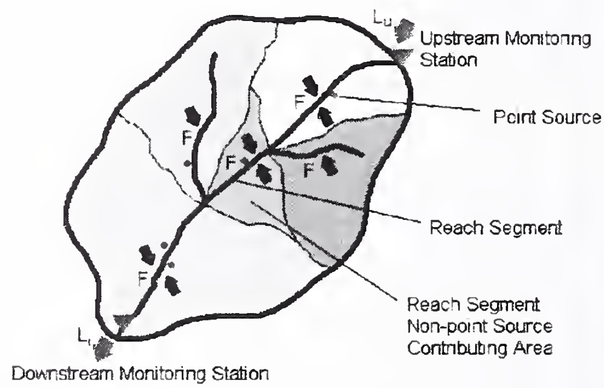

Figure 2. Schematic of a nested basin defined by upstream and downstream monitoring stations.

The flexible mathematical structure utilized in (1)-(3) is capable of accommodating a number of hypotheses concerning sediment fate and transport. Sites of sediment storage, identified in the model as a subset of the **d** variables, can act as sediment sources or sinks depending on the sign of corresponding δ coefficients. A random coefficient form of the model allows storage sites to serve as sources in some regions and sinks in others. Such behavior can be inferred statistically by relating the prevalence of storage sites in nested basins to the magnitude of the squared residual e in these basins (Godfrey 1988). Non-point sources of sediment, such as soil erosion included under **S**, are distinguished from sediment losses from storage (e.g., an alluvial plain) identified with **d**, on the assumption that the former is a primary process due to weathering whereas the latter is a consequence of the accumulation of previously weathered material which is later released to streams under changing hydraulic conditions. Accordingly, the potential for storage loss in the model depends on the extent of accumulated upstream soil erosion due to weathering. The empirical validity of the USLE estimate of soil erosion can be evaluated through statistical hypothesis tests conducted on the relevant β coefficients. Alternative measures of soil erosion can also be empirically evaluated in the model by substituting variables serving as determinants of the USLE for the USLE erosion estimate.

The estimation of long-term suspended-sediment load at a monitoring station is based on the regression of the natural logarithm of instantaneous suspended-sediment concentration on current and lagged values of the natural logarithm of daily flow and other variables representing seasonal and trend effects. If the station has concentration data collected more frequently than a weekly basis, the regression model is modified to account for serial correlation. To be included in the analysis, a station must have at least 3 years of data between 1985 and 1995. Only data within the period 1985-95 are included in the regression.

Mean-annual suspended-sediment load is estimated by first simulating load for each day over the 1985-95 period and then averaging daily values on an annual basis. Simulated loads are obtained by taking the exponential of the sum of the predicted daily load given by the calibrated regression model, with the time trend variable set to a base year of 1992, and a randomly selected residual from the regression model. For days having actual monitoring data, the daily load is computed by multiplying the measured instantaneous concentration by the daily flow. If a station has a data record with sufficient frequency to estimate a serial correlation parameter, the simulated daily load is based on the conditional prediction associated with past and future observed loads, plus a normally distributed random error having a correlation structure consistent with the conditional prediction and with the variance estimated by the regression model. The Monte Carlo process used to estimate simulated daily loads for the 1985-95 period is repeated 200 times, providing 200 values for estimating the mean and standard deviation of the average annual sediment load for a site.

Conclusions

The model described here is intended to empirically evaluate regional-scale processes affecting the long-term (i.e., decadal) transport of sediment in rivers. Additionally, the model will provide estimates of sediment mean annual flux for every reach included in the RF1 network. Error estimates for these process evaluations and stream predictions are determined using robust bootstrap methods. Future work will address the dynamic behavior of sediment flux associated with non-steady state streamflow conditions.

Acknowledgments

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Soil Water Dynamics Studies Using Image Analysis

Guillermo González Cervantes, Ignacio Sánchez-Cohen, Jean Pierre Rossignol

Abstract

Image analysis has become a powerful tool for describing soil porous media properties with hydrological purposes. In this study an experimental watershed in Northern Mexico was selected in order to characterize soil macro porosity and its variations in the soil profile for explaining runoff events. Preferential flow was determined using a colorant and then pore size and continuity was determined by means of image analysis. Four criteria were used for performing morphologic parameterization of pores: total porosity, pore size, pore shape and continuity and saturated hydraulic conductivity. Results have shown that horizons with low permeability have small pore size (<0.53 mm^2) and are rounded that function individually. On the other hand, horizons with higher permeability have both median size pores (form 0.53 to 1.58 mm^2) and big pores (>1.58 mm^2). The shape of these pores are enlarged and irregular with good continuity among them.

Keywords: soil pores, porosity, image analysis

Introduction

Distribution of water in soils takes place through void spaces (pores); in this way, organization and distribution of pores are of crucial importance for the transport and retention of water in the soil. This water it is used for crops evapotranspiration and or feeding aquifers. Also saturated hydraulic conductivity depends on the relative abundance and spatial distribution of void spaces. Volumetric description of soil porosity generally is not enough for explaining soil water dynamics (Hallaire et al. 1997). Therefore, soil porosity characterization based on the path of preferential water movement may be described and quantified based on three morphological criteria: a) size, b) shape and c) continuity. This may be pursued using image and analysis techniques which is a procedure that accounts for an important development in the study of soil porosity (Bouma et al. 1979; Stengel 1979, German and Beven 1981, Bullock and Mc Keague, 1984, Bruand 1986, Curmi, 1988, Grimaldi and Boulet 1989-90, Hallaire, 1997, Hallaire et al. 1997, González, 2002).

Objective

The overall objective of this study has been to describe soil porosity and the path of preferential soil water flux using image analysis through previously marked paths with a colorant (blue of methylene) over the main soil horizons in a watershed in northern Mexico.

Materials and Methods

Field work was carried out during 1999 in a watershed in northern Mexico known as Carboneras within the boundaries of the ranch Atotonilco located between parallels 24°33' and 24°50' north latitude, and the meridians 103°34' and 103°50' west longitude. Since a geological stand point Carboneras watershed is located over eruptive and sedimentary soil materials. Rocks and eruptive material are mainly located in the south part in the form of eruptive relieves and basaltic materials. Sedimentary materials are located due north in the form of dendrites materials of conglomerated and some calcareous materials (Figure 1).

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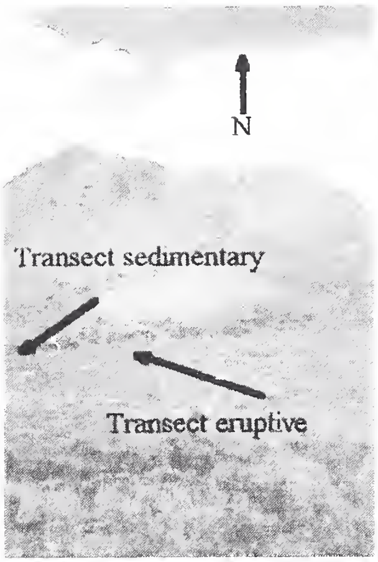

Figure 1. Carboneras Watershed and experimental sites.

Selection of representative soil horizons

After the soil description, six representative sites were selected in two transects as shown on Figure 1 (Gonzalez, 2002). These horizons are described in Figure 2.

Non disturbed soil monoliths of 1000 cm^3 were obtained using the Vergiere method (Bourier 1965). Saturated soil water conductivity (Ksat) was measured in the laboratory using a constant head infiltrometer (Beaudet 1998) and soil samples were obtained in order to determine physical and chemical characteristics (Table 1).

Soil paths characterization

Soil monoliths were saturated using a colorant blue of methylene ($C_{16}H_{18}ClN_{35}2H_{2}0$) in a concentration of 1 gr.l^{-1} during eight hours in order to highlight the pores that ensures preferential water flow in the soil (Bouma et al. 1979, Hallaire and Curmi 1994, Gonzalez 2002). Then the monoliths were saturated with acetone and impregnated with a polyester resin (Scott-Bader Crystic) containing a fluorescent pigment (Uvitex), (Murphy et al. 1977). Then horizontal thin layers of soil were obtained of approximately 3 cm in width.

Image analysis

Soil image analysis were performed using a camera (JVC 3CCD KY-F30B) with a data logger under the form of a rectangular matrix of 58 by 45 mm with a spatial resolution of 90μm by pixel highlighting soil samples by means of ultraviolet light for describing total porosity (Figure 3). Then using visible light for making evident marked paths with blue of methylene, image analysis was carried out using a special computer program (Optimas v5.2).

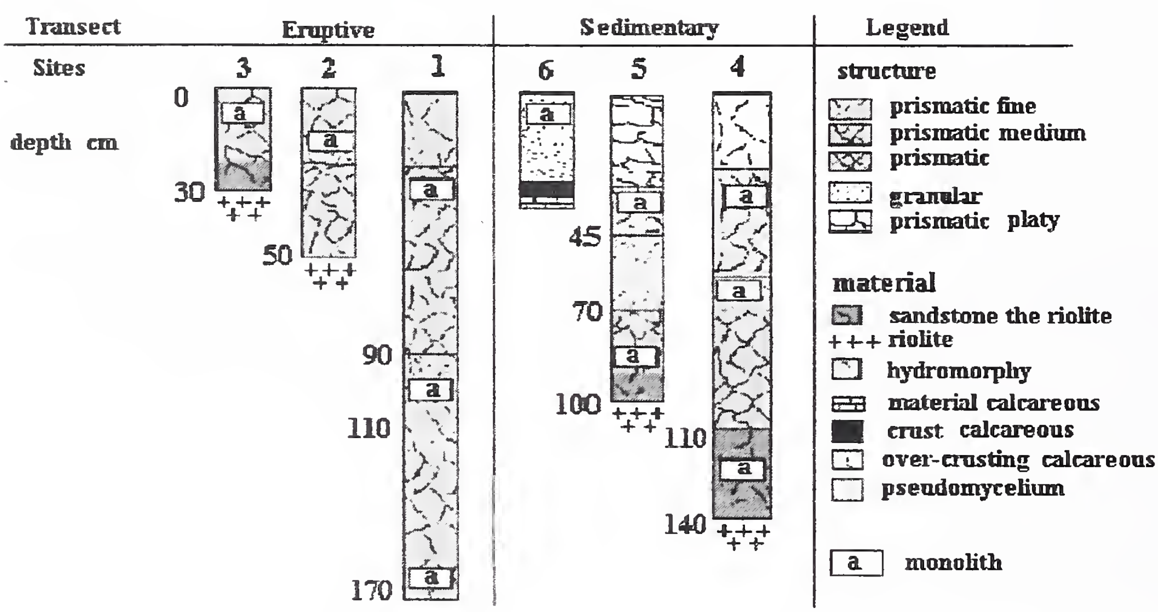

Figure 2. Site description of soil horizons.

Table 1. Physical and chemical characteristics of representative soil horizons of the eruptive and sedimentary transects.

Site	Depth	Clay	Silt %		Sand %		Carbon	pH water	Carbonate total
	cm	%	fine	thick	fine	thick	Total	2/1	%
3	15	22	4	11	40	23	1.5	7.7	0
2	15	18	16	12	37	17	1	7.4	0
	45	16	11	15	38	20	0.7	7.2	0
1	35	20	3	9	41	27	0.6	7.1	0
	115	7	27	12	34	20	0.3	7.5	0
6	5	14	16	13	34	23	6	8.2	34
5	15	17	16	12	28	27	3	8.1	9
	60	24	14	25	26	25	4	8.3	29
	95	18	21	12	25	24	0.6	8.4	3
4	10	12	25	12	26	25	2	8	1
	40	18	24	14	22	22	1	8	1
	65	26	18	14	20	22	2	8.3	15

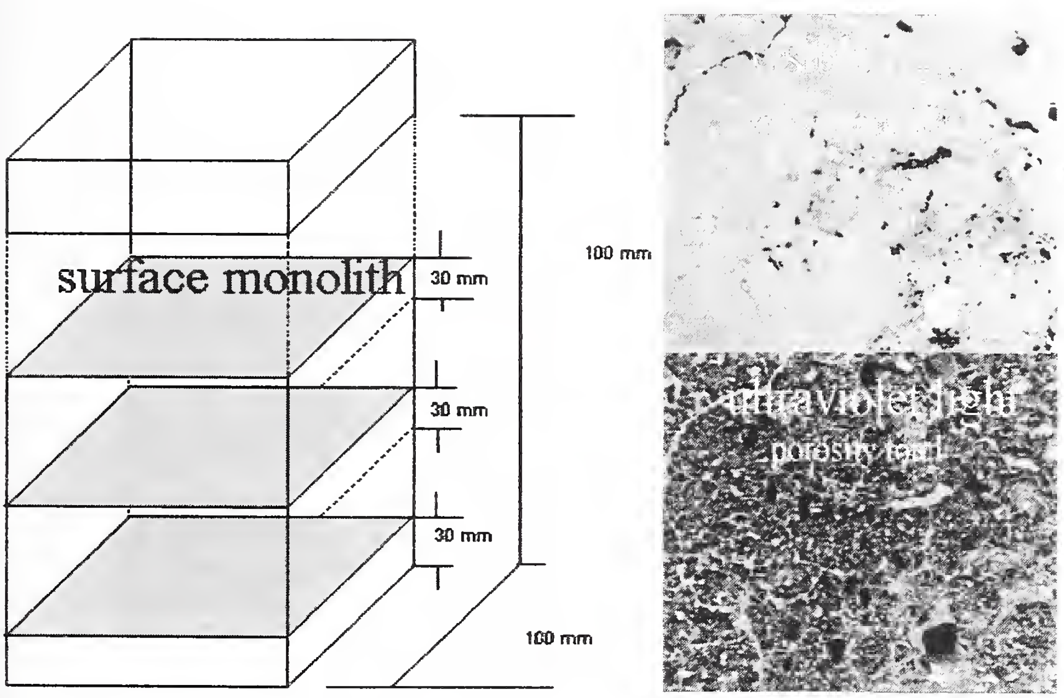

Figure 3. Monolith cutting and image sample.

Pores characteristics

Three morphologic parameters were used for characterizing soil pores. Pore size is computed using equation 1 which then was grouped in three different classes; T1 (small pores), T2 (medium sized pores) and T3 (big pores):

$$T = \frac{P^2}{4\pi \, x \, A} \qquad (1)$$

where T is size pores, and P is perimeter (cm) and A is pore area (cm^2).

Pore shape is expressed by an enlargement index (Ia) grouping three classes: F1 (round pores), F2 (enlarged pores) and F3 (irregular pores); see Table 2.

Table 2. Soil pore classification according shape and size.

Pore Shape		Pore Size		
		T1 small	T2 medium	T3 big
		area < 0.53 mm^2	0.53< area < 1.58 mm^2	area >1.58 mm^2
F1 round	Ia < 5	T1F1	T2F1	T3F1
F2 enlarged	5 < Ia < 10	T1F2	T2F2	T3F2
F3 irregular	Ia >10	T1F3	T2F3	T3F3

Continuity index is described by equation 2 (Serra 1982):

$$Ic = \frac{1 - Nc}{Un} \qquad (2)$$

were Ic represent image pore continuity and variates from 0 to 1; Nc is the number of convex voids and Un is the number of concave voids. Ic is small when pores are isolated and increases when pores are ordered in a series.

Results

Figure 4 shows results of saturated hydraulic conductivity (Ksat) for the two transects. Ksat variates from 252 to 108 mm hr^{-1} for the eruptive transect and from 144 to 28.8 mm hr^{-1} for the sedimentary transect. Based on this, three groups may be distinguished: horizons with high Ksat, sites 3, 2 and 1 of the eruptive transect; site 6 of the sedimentary transect and horizons with low Ksat of sites 5 and 4 of the sedimentary transect.

These Ksat variations lead to describe and characterize soil porosity using image analysis and then to correlate Ksat with morphological characteristics of soil pores considering the type of calcareous accumulation for the sedimentary transect and the sandy materials from reolite alterations for the eruptive transect.

Total visible porosity at 90 µm

Tables 3 and 4 shows distribution of total visible average of porosity at 90µm as a function of size (T1 to T3) and shape (F1 to F3) for surface horizons.

In eruptive transect (Table 3) a soil porosity variation from 6.2 to 8.4 % was observed for surface horizons (site 3, 2 and 1) and from 8.4 to 13.4% for deeper horizons (site 1). This porosity is mainly formed by small pores (T1 < 0.53 mm^2) and in less proportion by medium sized pores (T2 from 0.53 to 1.58 mm^2) and big pores (T3 > 1.58 mm^2). This former category has a different and irregular enlarged pore distribution. Calcareous accumulation en site 2 prevent soil porosity analysis.

On the sedimentary transect (Table 4) total soil porosity ranged from 8.1 to 10.4 % for surface horizons (sites 6, 5 and 4) and from 9.2 to 12.2 % for deeper horizons (sites 5 and 4). Soils in this transect are characterized by having a calcareous accumulation in the form of a crust at different depths.

Soil porosity analysis allows to describe pores distribution. Nevertheless, results interpretation does not allows to differentiate soil horizons with and without calcareous formation. On the other hand, results confirm that Ksat does not depend on the soil porosity only.

From there, the necessity of analyzing soil pores continuity index by means of equation 2.

Soil pores continuity index (*Ic*)

Table 5 shows values of porous media continuity index (*Ic*) for both eruptive and sedimentary transects. Horizons with calcareous crust formation (sites 2 and 6), with encrustment (sit 4 at 65 cm depth) and pseudo micelle (site 5 at 15 cm depth and site 4 at 25 cm depth) shows *Ic* values lower that sites with riolite - sand particles (sites 3, 1 and 4 at 130 cm). This result allows to differentiate horizons with and without calcareous formations and highlight the importance of pore continuity. Next a description of functional porous space is proposed.

Functional porosity

Table 6 shows the percentage of soil pores marked wit blue of methylene for the eruptive transect. Functional porosity ranges from 0.35 to 2.2% for surface horizons.

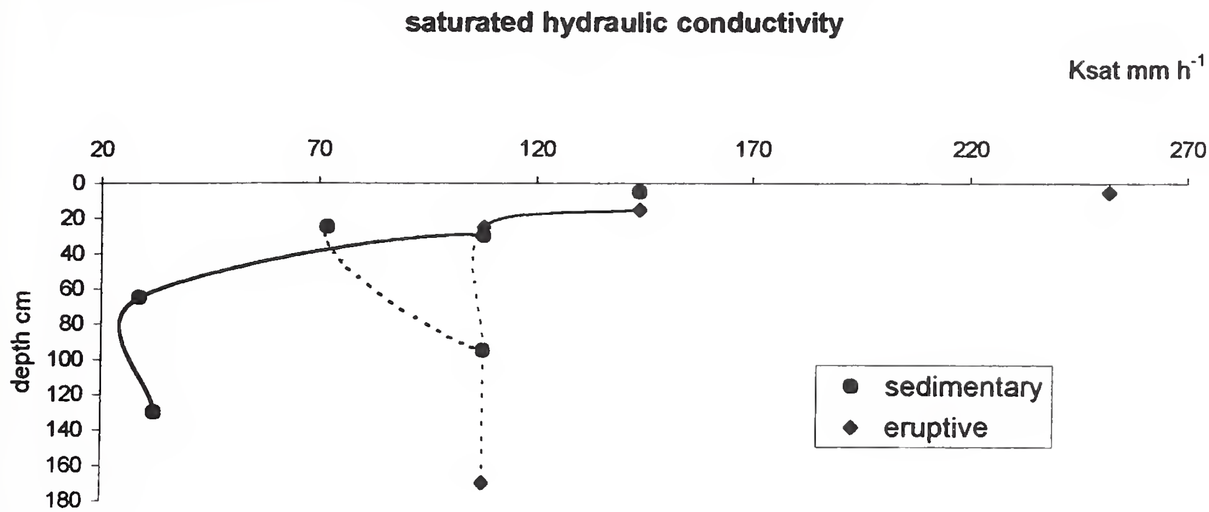

Figure 4. Values of saturated hydraulic conductivity (Ksat).

Table 3. Soil porosity characterization in the eruptive transect.

pore	Site 3 T1	Site 3 T2	Site 3 T3	Site 2 T1	Site 2 T2	Site 2 T3	depth cm		Site 1 T1	Site 1 T2	Site 1 T3	depth cm
round	2.9	0.4	0.3	3.5	0.1	0	15		5.5	0.5	0	25
enlarged	0.3	0.6	0.2	1.4	0.3	0.2			0.5	0.6	0.1	
irregular	0	0.2	1.3	0.4	0.9	1.3			0	0.2	1	
		total %	6.2			8.1					8.4	
								round	5.6	1.2	0.1	95
								enlarged	0.5	1.1	0.2	
								irregular	0	0.2	2.1	
										total %	11	
								round	6.2	1.7	0.5	170
								enlarged	0.5	1.2	0.9	
								irregular	0	0.2	2.2	
										total %	13.4	

Table 4. Soil porosity characterization in the sedimentary transect.

pore	Site 6 T1	Site 6 T2	Site 6 T3	depth cm	Site 5 T1	Site 5 T2	Site 5 T3	depth cm	Site 4 T1	Site 4 T2	Site 4 T3	depth cm
round	3.5	0.1	0	5	3.8	0.3	0.1	15	5.3	1	0.9	25
enlarged	1.4	0.3	0.2		1	0.9	0.1		0.4	1.1	1.2	
irregular	0.4	0.9	1.3		0.1	0.9	2		0	0.2	0.3	
		total %	**8.1**				**9.2**				**10.4**	
round					5.4	0.5	0	95	5.4	0.5	0.2	65
enlarged					1	0.7	0.2		0.6	0.6	0.3	
irregular					0.1	0.4	0.4		0	0.2	0.5	
						total %	**8.7**				**8.3**	
round									5	1.8	0.5	120
enlarged									0.4	1.1	1.2	
irregular									0	0.2	2	
										total %	**12.2**	

Table 5. Porous media continuity index (*Ic*).

Eruptive Transect site	depth	Ic	Sedimentary Transect site	depth	Ic
3	5	0.32	6	5	0.06
2	15	0.04	5	30	0.03
				100	0.04
1	25	0.07	4	25	0.1
	95	0.1		65	0.06
	170	0.1		130	0.11

Table 6. Characterization of soil porosity in the eruptive transect marked with blue methylene.

pore	Site 3 T1	Site 3 T2	Site 3 T3	Site 2 T1	Site 2 T2	Site 2 T3		Site 1 T1	Site 1 T2	Site 1 T3	depth
round	0.9	0.1	0.3	0.2	0	0		0.4	0	0	25
enlarged	0.1	0.1	0.1	0.1	0.02	0		0	0.1	0	
irregular	0	0	0.6	0.03	0	0		0	0	0.4	
		pF %	**2.2**			**0.35**				**1.2**	
		p T %	**6.2**			**8.1**				**8.4**	
							round	2.7	0.6	0	95
							enlarged	0.2	0.6	0.1	
							irregular	0	0	1.1	
									pF %	**5.3**	
									p T %	**11**	
							round	2.1	0.7	0	170
							enlarged	0.2	0.7	0.3	
							irregular	0	0.1	2.1	
									pF %	**6.2**	
									p T %	**13.4**	

Table 7: Characterization of soil porosity in the sedimentary transect marked with blue methylene.

	Site 6			Site 5				Site 4			
pore	T1	T2	T3	T1	T2	T3	depth cm	T1	T2	T3	depth cm
round	1.8	0.2	0	0.7	0	0		3	0.7	0	25
enlarged	0.1	0.5	0.4	0.1	0.1	0		0.1	0.7	0.8	
irregular	0	0.3	0.4	0	0.2	0.2		0	0.1	0.8	
		pF %	3.9		pF %	1.4			pF %	6.2	
		p T %	8.1		p T %	9.2			p T %	10.4	
round				0.1	0	0	95	0.9	0.1	0	65
enlarged				0.1	0	0		0.1	0.2	0.1	
irregular				0	0	0		0	0.1	0.1	
					pF %	0.2			pF %	1.6	
					p T %	8.7			p T %	8.3	
round								1	0.4	0.1	120
enlarged								0.1	0.3	0.2	
irregular								0	0.1	0.8	
									pF %	3	
									p T %	12.2	

Site three shows an important porosity (2.2% in relation to the image total) constituted by big pores (T3) and small pores (T1); on the other hand, site two shows a reduced porosity (0.35%) with small and rounded pores (T1); site one shows an intermediate soil porosity (1.2%) with a small and big pore distribution.

An important increment of marked pores with blue of methylene was observed in site one (5.3 and 6.2%) with pores of all classes and shapes but with a trend to big and irregular pores.

The sedimentary transect (Table 7) shows functional porosity variation from 1.4 to 6.2% for surface horizons; on site 6 a functional porosity of 3.9% was observed with a trend of small and rounded pores (1.8%); site 5 shows a marked porosity of 1.4% of rounded and small pores (0.7%); likewise, site 4 shows a functional porosity higher (6.2%), here also preferential pores are small and rounded. These data allows a correlation of Ksat with porosity and calcareous accumulation.

Morphologic parameters of porosity and saturated hydraulic conductivity

Table 8 shows Ksat values and the parameters of total porosity (pt), functional porosity (pf) and the soil pore continuity index (*Ic*) for both transects. According the results, horizons with calcareous accumulation shows a Ksat values ranging from medium to low with correlation to *Ic* values close to zero and with a distribution of functional pores of small size and rounded shape.

Table 8. Morphological and hydrodynamic soil horizon characteristics.

Site	depth cm	Ksat mm h^{-1}	pt %	Ic	pf %
3	5	252	6	0.32	2.2
2	15	144	8	0.04	0.3
1	25	108	8	0.07	1.0
	95	108	10	0.10	5.4
	170	108	16	0.10	6.2
6	5	144	10	0.06	3.9
5	25	72	9	0.03	1.4
	95	108	9	0.04	0.2
4	25	108	11	0.1	6.2
	65	28.8	8	0.06	1.6
	130	32.4	12	0.11	3.05

Non calcareous horizons (riolite material) shows a higher Ksat with good continuity among pores as shown by *Ic* values with a morphologic distribution of functional pores of size ranging from medium to big and with enlarged and irregular shape with a less abundant distribution but with a higher participation on soil water dynamics when these are interconnected.

Conclusions

Soil water dynamics and void spaces correlations lead to mark preferential flow under saturated conditions in a watershed in northern Mexico and allowed to establish a morphological characterization of soil functional porosity using image analysis. This characterization according size, shape and pore continuity allowed to establish a typology of soil horizons with and without calcareous accumulation coherent with the hydrodynamic functionality.

When soil horizons shows calcareous formation, functional pores are small (< 0.53 mm^2) rounded and generally abundant. In this horizons soil water dynamics is a function of pore size and shape but mainly to the lack of connection among them.

In horizons without calcareous accumulation, functional pores are from medium (0.53 to 1.58 mm^2) to big (>1.58 mm^2) size with a spatial distribution less abundant than small pores. Nevertheless, in this horizons soil water dynamics depends on the type of shape of the pores (enlarged and irregular) and the continuity among them.

In general terms the horizons of the sedimentary transect showed values of Ksat lower than the eruptive transect due that the calcareous material is transported by water and deposited in the void spaces affecting soil pores continuity as shown by *Ic* values.

Comparison of total porosity values prevent to explain differences between Ksat; on the other hand, the continuity index and functional porosity allows to describe soil water dynamics for these horizons.

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Modeling Phosphorus Transport in the Blue River Watershed, Summit County, Colorado

Paula Jo Lemonds, John E. McCray

Abstract

Lake Dillon in Summit County, Colorado, is a primary drinking-water reservoir for Denver. Eutrophication of Lake Dillon is a concern, primarily due to phosphorus (P) loading. There is little agriculture in the watershed. Thus, many local officials attribute the P loading to onsite wastewater systems (OWS). A watershed modeling effort using the SWAT model is underway to understand the potential influence of various point and nonpoint sources of P in the Blue River watershed (the most developed of three watersheds that supply Lake Dillon). The watershed model was calibrated to measured flow rates and P concentrations. The hydrologic model results are most sensitive to the physical parameters of snowmelt, and orographic effects on precipitation and evapotranspiration. However, uncertainties in chemical-hydrologic parameters preclude a rigorous assignment of relative contributions of various P sources. Rather, the effort has resulted in a better understanding of P chemical parameters required to simulate watershed-scale transport. The model was most sensitive to the P sorption coefficient, the P availability index, and the P enrichment ratio (a measure of P in runoff sediments compared to immobile sediments). Modeling results indicate that OWS are not significant sources of P to Lake Dillon.

Keywords: watershed, modeling, phosphorus, wastewater, hydrology, SWAT

Introduction

Numerical models are useful tools because they allow a quantitative assessment of the environmental impacts of wastewater pollutants and improve understanding of watershed-scale pollutant transport. Projecting future water quantity and quality is especially important in developing communities that rely on shallow groundwater as a source of drinking water while disposing of wastewater in the shallow subsurface. Some models capable of simulating watershed-scale pollutant transport include Soil and Water Assessment Tool (SWAT) (Arnold 1998), MIKESHE (Danish Hydraulic Institute 1999), Watershed Analysis Risk Management Framework (WARMF) (Chen et al. 1999) and Hydrologic Simulation Program – Fortran (HSPF) (Bicknell et al. 1996). SWAT was used for this effort because it is a public-domain model that can incorporate large amounts of data and simulate many hydrologic processes.

Several watershed-scale models have been developed using SWAT (Arnold and Allen 1996, Manguerra and Engel 1998, Srinivasan et al. 1998, Arnold et al. 1999, Santhi et al. 2001, Fontaine et al. 2002). However, these projects did not specifically address the watershed-scale impacts of wastewater pollutants from onsite wastewater systems (OWS). The goals of this study are to accurately simulate mountain watershed hydrology and to quantify the impacts of OWS-derived phosphorus (P) in the Lake Dillon Watershed.

The study area is the Lake Dillon watershed located in Summit County, Colorado (Figure 1). Lake Dillon is the main drinking-water storage reservoir for Denver. Towns in the watershed include Frisco, Dillon, Silverthorne, Breckenridge, and Blue River.

Model Setup

The ArcView Interface for SWAT was used in model development. Subwatersheds were delineated using SWAT and a USGS 300-m resolution, 1-degree Digital Elevation Model (DEM). Information extracted and calculated from the DEM includes overland slope, slope length, and elevation corrections for precipitation and evapotranspiration.

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The subwatershed delineation is illustrated in Figure 1.

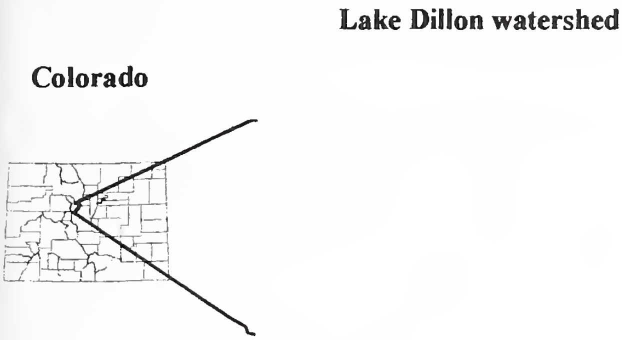

Figure 1. The study area is the Lake Dillon watershed in Summit County, Colorado.

The land-use information was derived from 1:250,000-scale Landuse/Landcover Geographic Information Retrieval Analysis System (GIRAS) spatial data. The land use/land cover digital data were collected by the USGS and converted to ARC/INFO by the USEPA. This information was used when simulating infiltration, runoff, ET, and natural sources of nutrients.

The soil attributes were taken from the State Soil Geographic (STATSGO) Database, which was developed by the National Cooperative Soil Survey (USDA-NRCS Soil Survey 2002). In the STATSGO database, soil attributes are stored in polygon format. Each polygon includes multiple soil series with information on its areal percentage of the polygon. In the SWAT ArcView Interface, the dominant soil series is selected, and the interface extracts properties for the model from a relational database. Examples of the properties extracted include soil texture, bulk density, hydraulic conductivity, available water capacity, organic carbon, and total depth of soil. These parameters are used in computations for infiltration, runoff, groundwater flow, and P transport.

Precipitation and temperature data were available from the National Climatic Data Center (NCDC) and the Natural Resources Conservation Service's (NRCS) Snowpack Telemetry (SNOTEL) Data Network. Six stations were available within the Lake Dillon watershed. Daily precipitation and minimum/maximum temperature values were incorporated into the model.

Other information defined in initial model setup included wastewater treatment plant point-source discharges into the Blue River and its tributaries, Lake Dillon water levels, reservoir outflow, and surface area of the reservoir. Information specific to each subwatershed, including stream-water chemical properties, groundwater-flow properties, stream-routing parameters, consumptive water use, and agricultural diversions were included. For details on the model setup and simulation parameters, the reader is referred to Lemonds (2003).

Incorporation of OWS

Currently, no algorithms exist in SWAT to specifically simulate OWS. Therefore, a fertilizer management practice was used to simulate OWS input. The mass input rate of OWS pollutants was set equal to the mass of nutrient input by the fertilizer. OWS inputs were established from reviews of OWS effluent flow rates and water-quality parameters completed by Kirkland (2001).

Fertilizer application input parameters were adjusted in SWAT to achieve the appropriate inorganic P mass input rate to the subsurface based on the number of OWS in each subwatershed. It was assumed that implementing the fertilizer management practice in seven-day intervals would adequately represent OWS effluent processes. SWAT allows the user to apply the fertilizer into the first soil layer. As a result, the simulated OWS nutrients are not affected by runoff and are allowed to percolate through the vadose zone. The source of percolation is the natural precipitation, which is orders of magnitude greater than OWS effluent input.

Results

Prior to simulating nutrient transport, physical hydrologic input parameters were adjusted to calibrate the model to stream flow rates. Adequate calibration of the physical hydrologic system was critical to simulating nutrient transport.

Hydrology simulation and calibration

Measured streamflow was obtained from two USGS gaging stations on the Blue River: one near the headwaters and the other located approximately one half mile upstream of Lake Dillon. The model simulation was executed for 11 years (1990-2000). The first two years were not used for model evaluation because parameters such as soil water content and residue cover are initially not in equilibrium with actual physical conditions (Santhi et al. 2001, Fontaine et al. 2002). Prior to calibration, comparison of annual-average streamflow data to simulated values show an under-prediction of flow (Figure 2).

Because SWAT was developed for watersheds in non-mountainous terrains, special adjustments were necessary to accurately simulate hydrologic processes that are strongly affected by elevation changes characteristic of this watershed. Fontaine et al. (2002), who applied SWAT to the mountainous Wind River Basin in Wyoming, discovered that orographic processes were very important. Processes that are affected by elevation include evapotranspiration, precipitation and snowmelt/snow-formation processes.

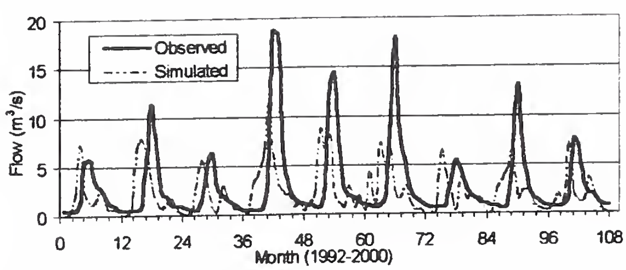

Figure 2. Initial simulation of monthly streamflow.

Lapse rates and elevation bands

Elevation in the Lake Dillon watershed ranges from 2681 m to 4350 m (8796-14,272 ft). To account for the orographic effects on precipitation and temperature (and thus evapotranspiration and snow processes), algorithms for elevation bands and lapse rates were used. In addition, several empirical parameters related to snowmelt and snow formation were adjusted.

The lapse rates were computed by relating elevation to mean annual temperature and mean annual precipitation at seven meteorological stations in the basin. The temperature decreases 4°C for an increase of 1km in elevation, and annual precipitation increases 5mm for an increase of 1km in elevation. Therefore, the temperature lapse rate was -4°C/1km (R^2 = 0.91, n = 7); the precipitation lapse rate was 5mm/1km (R^2 = 0.82, n = 6). These lapse rates were implemented by dividing the watershed into six elevation bands (2650-4150 m) based on the DEM. When lapse rates are defined in SWAT, subbasin temperatures and precipitation are adjusted for each elevation band in a subbasin as a function of the lapse rate and the difference between elevation of the meteorological gaging station and the average elevation specified for the band (Neitsch et al. 2000).

Adjustment of snowmelt/snow accumulation parameters

Parameters in SWAT that simulate snowmelt processes and control the formation of snow were also adjusted to create a better match to observed streamflow data. The parameters that were modified include a factor that accounts for snow pack characteristics and two empirical factors that account for the melting rate of snow.

A "lagging factor" accounts for temperature characteristics of the snow pack that influence the snow-pack density, snow-pack depth, exposure, and other factors (Neitsch et al. 2001). As the lagging factor approaches 1.0, the mean air temperature on the current day exerts an increasingly greater influence on the snow pack temperature, and the snow pack temperature from the previous day exerts less and less influence (Neitsch et al. 2001). In the model of the Lake Dillon watershed, the value was adjusted to 0.035. This value, which produced the best fit to observed data, is consistent with the findings of Fontaine et al. (2002) who observed values of the lag factor ranging from 0.0 to 0.5 for areas characterized by deep snowpack.

The other two factors influence the empirical relation used for snowmelt. Snowmelt is calculated as a linear function of the difference between the threshold temperature for snowmelt and the average snow-pack maximum air temperature. Two parameters in SWAT represent maximum and minimum melting values that occur on the summer and winter solstices, respectively. For the application of SWAT to the Lake Dillon watershed, these values were adjusted to 3.0 and 2.0 mm H_2O/day-°C, respectively.

Final hydrology calibration

The adjustment to snowmelt and snow formation parameters, as well as the inclusion of lapse rates and elevation bands, made a substantial improvement in the simulation of streamflow (Figure 3). The rising limb of each yearly hydrograph begins at the correct time. The recession limb of each yearly hydrograph begins at nearly the correct time. The years of higher discharge show improvement in the timing of the recession limb of the hydrograph (Figures 2 and 3, Months 18, 46, 70, and 92). The only problem that was not completely resolved was that the simulated streamflow approached 0.0 $m^3 s^{-1}$ for 2-3 months of the year. However, an improvement was made from the initial calibration. Comparison of Figures 2 and 3 show that the simulated hydrograph was smoothed considerably and better corresponds to the observed values of streamflow.

Statistics show the numeric improvement made in streamflow simulation. The initial R^2 value of monthly-averaged streamflow was 0.03. The simulation shown in Figure 3 exhibits an R^2 value of 0.70. R^2 values of 0.65 to 0.70 for monthly-averaged streamflow are appropriate considering the numerous potential measurement errors in data collection. For example, spatial variability in rainfall, soils, and land use, errors in measuring streamflow, and errors caused by sampling strategies are potential causes of inaccurate observed values (Santhi et al. 2001).

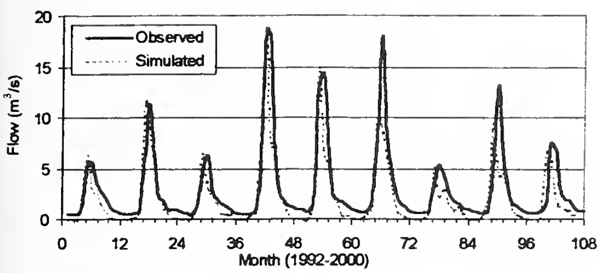

Figure 3. Model calibrated to monthly streamflow.

Phosphorus Calibration

To simulate pollutant transport, it is necessary to know the values for more than a dozen input parameters that influence the reaction, transformation, and interphase partitioning of the pollutants. Unfortunately, the available input data on these parameters, as well as observed data required to calibrate a model, generally are not available in the Lake Dillon watershed.

This is true in most watersheds. Thus, the first step in model improvement should include a sensitivity study to understand the relative importance of these parameters on model output. The next step is to use the parameters that are considered most important (in terms of the influence on the model) to evaluate the performance of the model in simulating actual, but limited, data. This exercise can also lend insight into designing a data-collection plan that would improve model performance.

Sensitivity study

For the sensitivity study, observed P concentration data for seven years were available at the Blue River station near Lake Dillon (the same location of the measured streamflow data). The observed P data are from the USEPA Storage and Retrieval (STORET) database (USEPA 2002) and from data collected by officials in Summit County, Colorado.

The automated calibration software, UCODE (Poeter and Hill 1998) was used to determine sensitivity of the model to several P transport parameters. Thirteen parameters in SWAT potentially affect P transport (Lemonds 2003). Of these parameters, the model was most sensitive to the P availability index (PAI), which specifies the fraction of fertilizer P that is in solution after a period of rapid reaction with the soil; the P enrichment ratio, which is the ratio of the concentration of P transported with the sediment to the concentration of P in the soil surface layer; the P-soil partitioning coefficient, which is the ratio of the soil concentration of P to the aqueous concentration of P at equilibrium; the initial P concentration in the soil; and the soil bulk density.

Best-fit phosphorus model

Parameters that had little affect on P transport were assigned reasonable values from the literature (Lemonds 2003). The parameters that most strongly affected the model were adjusted to yield a best-fit to observed values (Table 1). These values are all within reasonable ranges based on literature review (Soil Survey of Summit County Area, Colorado 1980, Sharpley 1984, Brady and Weil 1999, Kirkland 2001).

Table 1. P input parameters used in final simulation.

Parameter, Units	Value of Parameter for Best-Fit Model	
P availability index, unitless	0.7	
P-soil partitioning coefficient, m^3Mg^{-1}	175	
P enrichment ratio, unitless	Model calculates for each storm event	
Soil bulk density, $g\ cm^{-3}$	Soil Layer 1	0.80
	Soil Layer 2	0.90
	Soil Layer 3	0.85
	Soil Layer 4	0.9
Initial soluble P soil concentration, $mg\ P\ kg\ soil^{-1}$	Soil Layer 1	5
	Soil Layer 2	2
	Soil Layer 3	2
	Soil Layer 4	2

Figure 4 shows the observed P values versus the best-fit simulated values. The simulation produces P loading values that are generally within a factor of 10 of measured data and usually within a factor of 2. While the match is not rigorous for the entire simulation time, most of the important trends are captured. Simulations with no OWS input of P were also completed. The model-simulated P generally changed by less than 5%. Therefore, OWS is not likely to be an important contributor to P pollution in the Lake Dillon watershed. Rather, natural sources in runoff sediments are likely the most important contributor.

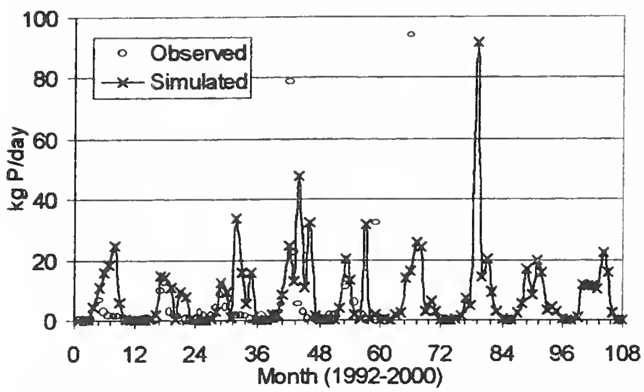

Figure 4. Best-fit model to observed P data.

Conclusions

Using public data that can be easily incorporated using the ArcView interface, SWAT accurately simulated mountain-watershed hydrologic processes. Variables associated with elevation-dependent temperature and precipitation (e.g. orographic) effects and snowmelt were adjusted. The orographic and snowmelt factors are particularly significant in the Lake Dillon watershed, where the elevation varies approximately 2000m.

A sensitivity study was completed to assess the influence of input parameters on simulated P transport. Several model input parameters were adjusted. Simulated P matched the overall trends of the limited measured data along the Blue River upstream of Lake Dillon. Because simulations without OWS contributions showed little change in the concentration of P in the stream, OWS are not believed to be the primary source of P in the lake. Instead, P in runoff sediments is the most likely contributor to surface water.

The uncertainty associated with the assignment of some chemical and hydrologic parameters indicates that additional information on the actual values and variability of pollutant-transport input variables is necessary. This is a feasible option, considering that most of the parameters containing approximated values (P soil-partitioning coefficient, mineral P concentration in the soil, and soil bulk density) may be quantified with additional collection and analysis of field data from the Lake Dillon watershed. However, it is not clear that additional measurement would benefit these particular simulations. For example, if parameter values varied greatly over the watershed, it may be impractical to collect enough measurements to obtain accurate values of input parameters. In such cases, sensitivity studies that use the reasonable range of parameters to assess a range in possible model outcomes still can be very useful and may be the only option. Despite the uncertainties related to model inputs, the model performs reasonably well. Thus, the model may be used to investigate different management options, such as using sewers versus OWS for a variety of pollutants, the influence of growth and increased OWS, or evaluating the effect of advanced OWS treatment systems on watershed water quality.

Acknowledgments

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October 27–30, 2003

Watershed Networks and Data Management I

Long-Term Forest Hydrologic Monitoring in Coastal Carolinas

D.M. Amatya, G. Sun, C.C. Trettin, R.W. Skaggs

Abstract

Long-term hydrologic data are essential for understanding the hydrologic processes, as base line data for assessment of impacts and conservation of regional ecosystems, and for developing and testing eco-hydrological models. This study presents 6-year (1996-2001) of rainfall, water table and outflow data from a USDA Forest Service coastal experimental watershed on a natural pine-hardwood forest in South Carolina (SC) and a small, 29-yr old intensively managed, drained pine forest owned by Weyerhaeuser Company in coastal North Carolina (NC). Results from this study showed a wide variation in annual outflows as affected by water table position, which is dependent upon both rainfall and evapotranspiration (ET). Although average annual rainfall was lower, the undrained watershed in SC had much shallower water table depths with higher frequent outflows compared to the drained NC watershed. The study emphasized the need for long-term rainfall and ET data and soil water properties in comparative assessments of the hydrology of poorly drained coastal watersheds.

Keywords: rainfall, outflow, water table, drained pine plantation, naturally drained forest

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Introduction

Scientists and researchers have long recognized the need for long-term hydrologic monitoring of various watersheds to understand the basic hydrologic processes that occur as a result of both natural events and anthropogenic disturbances. Long-term monitoring provides us with a database for evaluating responses and generating new scientific hypotheses, and wider range of observational data for testing of hydrologic and water quality models. Yapo et al. (1996) concluded that nearly eight years of data are required to obtain model calibrations that are relatively insensitive to the period selected, because of year-to-year variation in weather that may be sometimes either extremely wet or dry. This may also be true for understanding the water table (hydroperiod) dynamics, hydrologic and nutrient cycling processes, water and nutrient budgets as well as their interactions with the ecosystem.

Forests are an important part of the ecosystem and play a great role in regulating the regional hydrologic patterns of the southern US where 55% of the region is covered by forests (Sun et al. 2002). Long-term hydrologic data from small, paired, experimental forested watersheds at Coweeta Hydrologic Laboratory in North Carolina (NC) integrated with an ecosystem approach have provided basic understanding of eco-hydrological processes for regional upland watersheds (Swank et al. 2001). Tajchman et al. (1997) reported the water and energy balance of a 39 ha central Appalachian watershed covered with 80-yr old upland oaks and cove hardwoods using 40 years of hydrologic data. However, there are only a few such observational studies done for the forest ecosystems that occur along the lowlands of southeastern coastal plains. Unlike the upland watersheds dominated by hillslope processes, hydrologic processes on relatively low gradient poorly drained coastal plains are usually dominated by shallow water table positions. Most of the outflows (surface runoff and subsurface drainage) from these watersheds, in fact, drain from saturated areas where the water is either at the surface or a shallow water table is present. This means the total outflow is dependent upon the position of the water table. Water budgets as well as water table management and water quality studies for drained pine plantations in coastal NC were presented by McCarthy et al. (1991) and Amatya et al. (1996, 1998). Most of these studies were limited to artificially drained lands. Recently Chescheir et al. (2003) documented the baseline forest outflow characteristics of 41 forested watersheds from coastal NC. These watersheds varying in sizes from 7 ha to 6,070 ha were on natural and managed pine forests and included data spanning 25 years (1976-2000).

Millions of hectares under silvicultural management in the lower coastal plain along the southeast and Gulf Coast region (Figure 1), however, consists of natural lands with non-pattern drainage systems. The types of forests managed on these lands vary widely from loblolly pine to bottomland hardwoods to pine flatwoods to even short rotation woody crops. More than one million acres in the coastal area are classified as pine flatwoods alone (Sun et al., 1998). Bottomland hardwood forests occupy nearly 300 million acres in the southeast. With the growing demand on timber and increased coastal urbanization in the southeast and Gulf Coast, there is an increased potential for developments of these forested lands as well. Therefore, researchers and land managers are not only challenged to understand the hydrologic processes for these coastal forest systems but also to address the issues related with the climatic and management impacts on water quantity and quality.

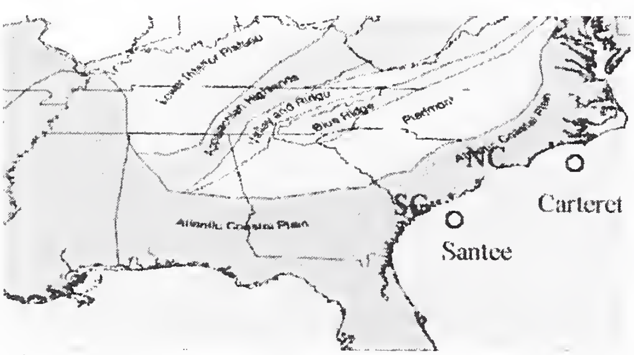

Figure 1. Location of Santee (WS 80) watershed in SC and Carteret (D1) watershed in NC.

A preliminary comparison of long-term water budgets for two paired coastal watersheds in headwater streams in South Carolina (SC) was recently conducted by Sun et al. (2000). Miwa et al. (2003) characterized stream flow dynamics and its relation to rainfall for the same two watersheds. A comparative study on long-term hydrologic characteristics (seasonal runoff patterns, water balances, stormflow patterns) of three watersheds in the southern US were presented by Sun et al. (2002). These watersheds included pine flatwoods of Florida, drained pine plantation site in coastal NC, and a hilly upland watershed at Coweeta in western NC. The study concluded that climate is the most important factor in the watershed water balances, and topography affecting streamflow patterns, is the key to wetland development in the southern US. Some long-term studies on pine flatwoods in Florida were conducted by Riekerk (1989). Most of the other long-term studies in the coastal plains were based on simulation models (Skaggs et al.1991, McCarthy et al. 1992, Amatya and Skaggs 2001, and Sun et al. 1998, 2000).

Recent report by Southern Forest Resource Assessment (Wear and Greis 2002) emphasized a need of research to assess the long-term cumulative non-point source impacts of silvicultural activities on water quality and overall watershed health. In order to evaluate the effects of these activities and develop database for reference wetlands on these various forest ecosystems, it is essential to understand the hydrologic processes and quantify the long-term hydroperiod dynamics and water and nutrient budgets. The objective of this study is to analyze the six-year (1996-2001) hydroperiod and outflow processes for two different watersheds in Coastal Carolinas (Figure 1) as affected by site specific climate regime and management practices. One of the watersheds is naturally drained and is on USDA Forest Service's Santee Experimental Forest in South Carolina established in 1937 for the long-term scientific study of the coastal forest ecosystems and their management. The other watershed, instrumented in 1987, is an artificially drained loblolly pine forest, owned and managed by Weyerhaeuser Company in coastal North Carolina. The site is being used to conduct long-term field studies to examine the impacts of different water management and silvicultural treatments on the hydrology and water quality during the life cycle of the intensively managed pine forest (Figure 1).

Methods

Site description

Santee watershed (WS 80)

This naturally drained watershed is located in the Santee experimental forest, which is part of Francis-Marion National Forest (USDA Forest Service) near Charleston, South Carolina (Figure 1). This is one of the paired

Table 1. Comparison of watershed characteristics for Santee, SC and Carteret, NC sites.

Parameter	Santee watershed (WS 80)	Carteret watershed (D1)
Location	Latitude 33°N, longitude 80°W	Latitude 34°N, longitude 76°W
Elevation above Mean sea level	7.0 m	3 m
Watershed size	206 ha	25 ha (ditched)
Long-term annual precipitation	1350 mm	1340 mm
Long-term mean annual air temperature	18.7°C	16.2°C
Topography/slope	<4%	<0.2%
Drainage type	Natural, first order stream	Artificially drained, pattern drainage

Note: Long-term climate data for Santee watershed taken from Charleston International Airport (1971-2000) and long-term data for Carteret taken from Morehead City (1971-2000).

watersheds (with WS77) on headwater stream draining to Turkey Creek on the lower Atlantic Coastal Plain. It is on a marine terrace of the Pleistocene epoch (Gartner and Burke, 2001). The area has low relief with surface elevations ranging from 4.0 to 10.0 m above mean sea level (Table 1).

Loblolly pine, longleaf pine, cypress, and sweet gum are dominant forest species in the watershed (Sun et al. 2000). Soils are primarily (loams) strongly acid, infertile Aquults, characterized by seasonally high water tables and argillic horizons at 1.5 meters depth with low base saturation (Gartner and Burke 2001). The climate of the research area is classified as humid subtropical with long hot summers and short mild winters (Sun et al. 2000). Mean annual precipitation is about 1350 mm with the highest rainfall in July-August and the lowest rainfall in the November-April period as the driest months. Meteorological data (daily maximum and minimum air temperatures and precipitation) have been collected since 1976 at Met-25 station inside the watershed and since 1946 at a weather station located at Santee Headquarters, which is about 10 km from the watershed (Sun et al. 2000). A stream gauging station consisting of a compound V-notch and a flat concrete weir with a recording gage inside a stilling well was installed at the watershed outlet in 1968. However, flow measurements were interrupted in 1981 and did not start again until after Hurricane Hugo in September 1989. Automatic ISCO-4210 flow data loggers were installed only in 1996. Since flow data were missing for intermittent periods, annual flow data reported herein were complete only for the years 1997 and 1998. Water table measurements were measured using manual wells at several locations in the watershed until 1995 and one automatic recording well (WL-40) was installed in late 1995.

Long-term data for daily precipitation, air temperature, and stream flow from this study site is recently being made available through HYDRO-DB, a WEB based data sharing and harvesting site hosted by Oregon State University and sponsored by USDA Forest Service and National Science Foundation's Long term Ecological Research (LTER) study. The site is located at www.fsl.orst.edu/climhy/hydrodb/ and data from various participating sites are posted here for inter-site comparisons and other useful hydro-ecological assessments including forest fire.

Carteret watershed (D1)

Field measurements on the Carteret 7 research site, located in Carteret County, North Carolina, have been conducted since early 1988 (Figure 1). The research site consists of three artificially drained experimental watersheds on flat, poorly drained soils with shallow water tables. The depth to the restricting layer is about three meters. The soil is a hydric series, Deloss fine sandy loam (fine-loamy mixed, Thermic Typic Umbraquult). Results analyzed in this paper were obtained from the control watershed (D1), which was managed in conventional drainage mode throughout the study period. The watershed is drained by four 1.4 to 1.8 m deep lateral ditches spaced 100 m apart.

Total rainfall was collected with an automatic tipping bucket rain gauge backed up by a manual gauge in an open area on the western side of the watershed. Air temperature, relative humidity, wind speed and net radiation were collected on an hourly basis at an automatic weather station located near the study site.

An adjustable height 120° V-notched weir, located at the outlet of the watershed, allowed measurement of drainage outflow by continuously recording water levels upstream of the weir kept at 1 m below average ground surface for free drainage.

Data on soil, hydrology and vegetation parameters were collected on three rectangular plots within the watershed. Water table elevations were measured by continuous recording wells in two plots midway between the field ditches. Daily water table elevation for the watershed was calculated as average of the midpoint wells. Details of hydrologic measurements and their procedures are documented elsewhere (McCarthy et al. 1991, Amatya et al. 1996).

Average annual rainfall as well as its variability for the six-year period was higher at NC Carteret site (1450 mm) compared to SC Santee site (1330 mm), although the long-term (1971-00) average was nearly the same for both sites (Table 1). This greater variability in NC site was primarily due to frequent summer-fall hurricanes and tropical storms that hit NC in 1996, 1998, 1999, and 2000. This is evident from the average monthly rainfall compared for the two study sites in Figure 2. Average monthly rainfall for both sites was comparable, except for the months of July to September. Year 2001 was the driest year with only 852 mm at Carteret site and 1106 mm at Santee, respectively (Table 2). The higher average annual temperature at

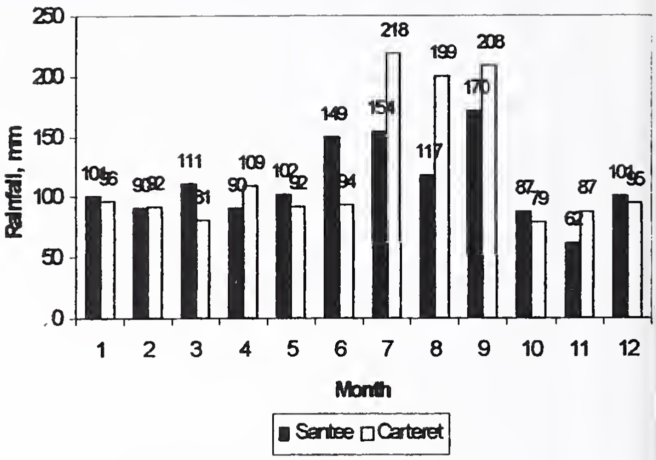

Figure 2. Comparison of average monthly rainfall at Santee (WS 80) and Carteret (D1) watersheds.

Santee site (18.8°C) compared to Carteret (16.4°C) indicates a higher potential ET at the former site. The highest difference was in 2001.

Results and Discussions

Annual temperature, rainfall, water table depths, outflow, and outflow as percentage of rainfall two measured at Santee (WS 80) and Carteret (D1) watersheds for 1996-2001 are presented in Table 2.

Table 2. Measured annual average temperature, total rainfall, average water table depths, and total outflow on Santee (WS 80), SC and Carteret (D1), NC watersheds for 1996-2001period.

	Santee watershed (WS 80)					Carteret watershed (D1)				
Year	Mean temp. °C	Total rain mm	Water table cm	Total outflow mm	Outflow as % of Rain	Mean temp. °C	Total rain mm	Water table cm	Total outflow mm	Outflow as % of Rain
1996	17.6	1303	N/A	372	29	15.8	1706	94	704	41
1997	18.7	1498	24	652	44	16.1	1408	107	397	28
1998	20.0	1463	35	858	59	17.7	1655	100	770	47
1999	18.5	1446	28	388 (i)	-	16.7	1362	88	614	45
2000	18.0	1167	30	280 (i)	-	15.6	1718	81	857	50
2001	19.8	1106	43	0 (i)	-	16.3	852	153	45	5

Santee watershed: Missing rainfall data were taken from Charleston International Airport. Total outflow was incomplete for full years of 1999 to 2001 and indicated as (i) beside total for available days. Outflow percent was not calculated for these years. Carteret watershed: Missing rainfall came from the gauges on adjacent watersheds and/or weather station.

Since water table position, as affected by rainfall and ET, is a dominating factor in generating outflow from these watersheds, measured water table depths for the Santee (WS 80) and Carteret (D1) watersheds were analyzed for the 1997 to 2001 period (Figure 3). One reason for the big difference in water table depths between the two watersheds during the summer was due to well depths, which was shallow (only 52 cm) for Santee watershed and 240 cm for the Carteret site. The water table remained within about 40 cm of the surface for most of the time during the winter and spring for the undrained Santee watershed, compared to drained Carteret site, which had deeper (~ 100 cm) depths for the same period. Average water table depth during this period at Santee watershed thus seemed to be shallower than about 47 cm reported for the 1992-95 study period (Sun et al. 2000). As a result of large event due to Hurricane Bonnie in late August of 1998, water table rose as high as 30 cm for a brief period at Carteret site in NC. Water table continued to decline below 150 cm from about May to the end of the year 2001 at Carteret site and, perhaps, at Santee watershed also, indicating a very dry period. The 5-year average water table depth at undrained Santee watershed was only 32 cm (probably, underestimated due to shallow well depth) compared to 106 cm at drained Carteret site, as expected (Table 2). A shallower depth to impermeable layer at Santee site may also explain this difference.

Earlier studies on Carteret site with the drained pine plantation (D1) had shown outflow occurring mostly as subsurface drainage when midpoint water table was within about 75 cm depth (Amatya and Skaggs 2001). Because of the depressional storage created by the plantation beds, surface runoff rarely occurs, except for the extreme events during the hurricanes such as Bonnie in August 1998 (Figure 3). For example, two years (1997-98) of daily flow data from both the watersheds were plotted together in Figure 4 to compare the flow regimes as affected by the water table depths. Outflows from naturally drained Santee (WS 80) watershed had very frequent and large storm outflows in both years compared to drained pine forest at Carteret site in NC. This coincides with much shallower near surface water table observed for a longer period on Santee WS 80 compared to Carteret site. Thus, the naturally drained Santee watershed yielded 46% and 59% of rainfall as outflow for the years 1997 and 1998, respectively (Table 2). Using five-years (1976-80; 1990-91) of data, Sun et al. (2000) found only 23% of the annual rainfall lost to outflow for the WS 80 watershed, with an average water table depth of 47 cm, which was deeper than those observed during this study period. These outflow values are also much higher than those observed for drained Carteret site (Table 2) for the same years. The six-year average annual outflow for

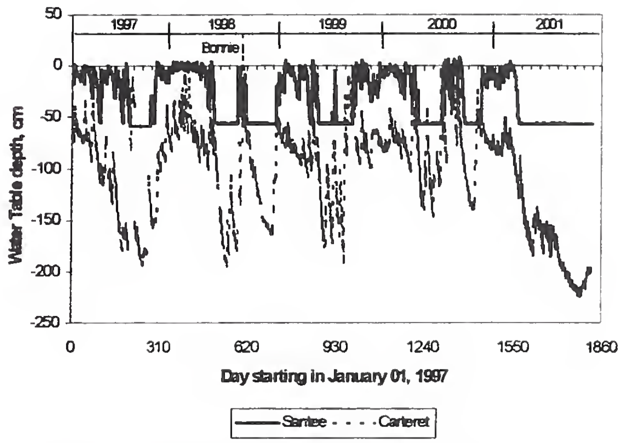

Figure 3. Measured water table depths for Santee (WS 80) and Carteret (D1) watersheds for 1997-01.

Carteret site was also higher than the 10-year data (Amatya and Skaggs, 2001), due to higher than long-term rainfall. The measured flows on Santee site, especially in wet winter (*La Niña*) of 1998 (Figure 4), may have been overestimated due to additional surface runoff from adjacent areas when the watershed was flooded. However, the event of Hurricane Bonnie in August 1998 had only a minimal effect on outflows from Santee compared to the Carteret site that resulted in a large storm outflow with a peak outflow of about 35 mm/day (Figure 4) due to 30 cm rise in water table (Figure 3). Flow ceased on both watersheds after this event.

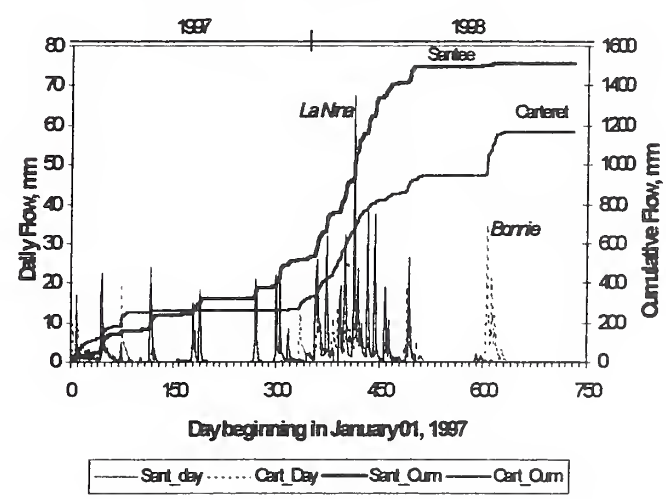

Figure 4. Daily and cumulative outflows at Santee and Carteret watersheds for the 1997-98 period.

Summary

A limited six years of rainfall, water table and outflow data from two long-term monitoring sites on coastal watersheds, a headwater stream draining a natural forest in SC and a drained pine forest in NC, showed a wide variation in annual rainfall pattern, especially in NC. As a result, watershed response, which depended upon the water table position, was found to be different from earlier studies on both of these sites. The drained NC site had deeper water tables than the undrained forest at the SC site. As a result of shallow near surface water tables, SC site yielded much higher outflows than the drained NC site for two years of data analyzed. Deeper water tables due to antecedent conditions resulted in low or near zero flows at both sites. Further study is needed to analyze the data in the context of rainfall, ET and soil conditions. This and past studies indicate that a long-term data monitoring is necessary for reliable comparative hydrologic assessments.

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Forest Service Watershed Research in the Southwest

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Abstract

Forest watershed research in the Southwest started in Arizona because of concerns about sediment inputs into the newly constructed Roosevelt Reservoir. The Summit Plots were established in 1925 to study the effects of vegetation establishment and mechanical soil stabilization on stormflow and sediment yields. In 1932 the Forest Service dedicated the Parker Creek Experimental Forest, later renamed Sierra Ancha Experimental Forest, for watershed research. In the 1950s and 1960s Forest Service research expanded to other areas such as Three-Bar, Brushy Basin, Whitespar, Mingus, Battle Flat, Beaver Creek, Castle Creek, Thomas Creek, and Willow Creek. The purpose of this expansion was to cover the full range of forest and woodland vegetation types from chaparral to pinyon-juniper, ponderosa pine, mixed conifer, and subalpine forest. Other smaller research sites were added in Arizona and New Mexico over the ensuing years to examine the watershed effects of grazing, wildfire, and riparian management. This paper examines the history of the Forest Service's watershed program in the Southwest and its usefulness for forest land management.

Keywords: watershed management, forests, ponderosa pine, mixed conifer forests, chaparral, Southwest

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Introduction

A common impression of the southwestern United States is that it is an area of vast deserts covered with cacti and low shrub species. Actually, the topography and vegetation of the Southwest are very diverse, including high plateaus and mountains that extend over 3,660 m in elevation.

The higher elevations receive relatively large amounts of precipitation, often in the form of snow, and support forests of ponderosa pine (*Pinus ponderosa*) and mixed conifers. Forested watersheds are the sources for much of the surface water for the major river systems within the region, including the Colorado, Salt and Verde in the Gila River Basin, and Rio Grande. The waters from the Salt and Verde Rivers in Arizona and Rio Grande in New Mexico are one of the reasons for the pre-historic American Indian and later European-American settlements and subsequent rapid growth of the Phoenix and Albuquerque Areas.

The need for watershed protection was recognized early. The Tonto National Forest in central Arizona, for example, was established in 1905 to protect the Salt River watershed and the Theodore Roosevelt Dam, the first reclamation project in the United States, which was under construction at the time.

Sound watershed management should be based on research. Watershed management research was initiated in Arizona in the 1920s when concerns developed that accumulations of sediment behind Roosevelt Dam would compromise the reclamation project. Research subsequently expanded to answer managers' questions about the hydrology of upland watersheds and efforts to manage forests and woodlands for augmented stream flows in the context of multiple resource management. Forest managers incorporated research results into management activities and into multiple-use planning.

Current watershed management efforts are aimed at maintaining or improving watershed conditions and water quality and at assessing the impacts of recent disastrous wildfires on hydrology and watershed condition. This paper reviews past forest watershed management research in the Southwest, management to reduce erosion and improve water quality from forested areas, and the impacts of fire on forest watershed values.

The Setting

Topography

The topography of the Southwest is very diverse. Elevations in Arizona vary from less than 42 m near the border with California and Mexico, to more than 3,862 m on top of the San Francisco Peaks near Flagstaff. Twenty-three mountain ranges are identified within New Mexico; the highest mountain has an elevation of 4,012 m. Arizona is divided into the Colorado Plateau, the intermediate Central Highlands, and the Basin and Range geological provinces that are located from the north to the south. Each of these provinces contains mountains, canyons, and valleys, cliffs and plains that were formed and influenced by geological events (Lowe 1964). New Mexico is divided into three topographic zones. These include a Rocky Mountain zone, the Plains zone of the eastern border with Texas and Oklahoma, and the Intermountain Plateau and Valley zone.

Climate

The climate is characterized by variable frontal precipitation in winter from the Pacific Ocean, an arid pre-summer, and summer rains that are predictable in timing and amount at a given station but vary from site to site. The summer moisture comes primarily from the Gulf of Mexico in a monsoon-like seasonal pattern. The proportion of winter to summer moisture varies throughout the region with southern and eastern areas being more dependent on summer precipitation than more northern and western sites. The summer season is characterized by convective storms that are often associated with high intensity rainfall events. Occasional tropical storms enter the region from the Pacific Ocean in late summer.

Desert areas of the Southwest average less than 25 mm of annual precipitation (Sellers and Hill 1974); however mountainous regions may average between 760 and 1,015 mm as rain and snow. Elevation is a key factor influencing the amounts of precipitation recorded in the Southwest. The highest mountains receive the largest amounts of moisture, primarily as snow. Higher precipitation and lower temperatures provide the necessary environment for forests to occur. Some of the plateaus, which are more than 1,830 m in elevation, are quite arid due to rain shadow effects.

Vegetation

The Southwest contains six life zones from deserts to alpine tundra based on vegetation types and varying by elevation and aspect (Lowe 1964). Ponderosa pine and mixed conifer forests are of prime interest because of their importance to watershed and natural resource management.

Ponderosa pine forests occupy about 2.38 million ha in the region (Van Hooser et al. 1993, O'Brien 2002). Ponderosa pine forms almost pure stands in association with a variety of grasses and other herbaceous species. The species is found from about 1,600 to 2,600 m in elevation and is often associated with pinyon (*Pinus edulis*), alligator juniper (*Juniperus deppeana*), and Gambel oak (*Quercus gambelii*) at its lower range, and with Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) at higher up.

The southwestern mixed conifer forests occupy about 809,000 ha between 2,400 and 3,400 m in elevation. These forests are a combination and intermixture of forest species and types. The dominant tree species are Douglas-fir, ponderosa pine, white fir (*Abies concolor* var. *concolor*), Engelmann spruce (*Picea engelmannii*), blue spruce (*P. pungens*), corkbark fir (*A. bifolia*), southwestern white pine (*P. strobiformis*), and quaking aspen (*Populus tremuloides*). Limber pine (*P. flexilis*) stands are only found in northern New Mexico.

The Southwestern forests were noted historically for their production of timber when the railroads entered the region in the 1870s. The value of forested watersheds was recognized in the nineteenth century. Current management on non-reserved lands is aimed at multiple-use of the forest resources for wildlife habitat, a number of threatened, endangered, and sensitive species, livestock, and recreation. The latter has grown in importance with the recent increases in the Southwest's population.

Watershed Management Research

Research scope

A complete coverage of the research conducted on the Forest Service watersheds in the Southwest is beyond the limitations of this manuscript. Work was conducted on the complete scope of watershed management topics. This includes management topics such as harvesting, thinning, site preparation, chemical vegetation management, grazing, road construction, recreation, and cumulative effects. Specific process-level studies examined water yield, snow hydrology, water quality, erosion, heavy metals, sedimentation, aquatic biology, nutrient cycling, etc, A more complete description of the details of this research program can be found in Baker (1999) and Gottfried et al. (2003). The focus of this narrative is on the history of Forest Service's key watershed research sites.

History of watershed research

Initial watershed research efforts were directed in determining methods of controlling erosion from lower elevation chaparral sites surrounding Roosevelt Reservoir. Chaparral is a vegetation type dominated by evergreen, sclerophyllous shrubs. Soil erosion was seen as a threat to the longevity of the dam and the general reclamation project along the Salt River. The Summit watersheds were established in 1925 to address the problem.

The Forest Service's Southwestern Forest and Range Experiment Station (now the Rocky Mountain Research Station) established the Parker Creek Experimental Forest in 1932 in the Sierra Ancha Mountains northeast of Roosevelt Reservoir and enlarged and renamed the forest in 1938 as the Sierra Ancha Experimental Forest. The 5,190 ha experimental forest, because of its broad elevational range (1,080 to 2,355 m), includes seven vegetation types from desert-shrub to mixed conifer forests (Pase and Johnson 1968). The research objective was to study the effects of grazed and ungrazed vegetation on water yields, and to learn more about water cycle relationships within the diverse vegetation zones of the Southwest (Gottfried et al. 1999a).

Both plot and watershed research was initiated at Sierra Ancha. Much of the early work concentrated on grazing effects, primarily in the chaparral shrublands that cover about 57% of the Experimental Forest. However, major watershed studies were begun in the 1930s at the Workman Creek watersheds in the mixed conifer-ponderosa pine forests and on Parker Creek and Pocket Creek that supported a mixture of chaparral stands and conifer forests, depending on aspect. The initial effort at Sierra Ancha was designed to determine the basic hydrologic relationships for forested watersheds in the Southwest.

An effort was launched in 1955, during a period of extended drought, to determine the feasibility of increasing stream flow by manipulating the plant cover in the different vegetation types within the Salt and Verde River Basins (Fox et al. 2000). Although watershed treatments were in progress at the time in the mixed conifer stands at Workman Creek, additional research was initiated to evaluate the chaparral, pinyon-juniper, and ponderosa pine forests of central Arizona and other mixed conifer watersheds in eastern Arizona. The goal of this work was to determine the effects of vegetation manipulations on the other natural resource products and uses as well as on runoff. A review of research results determined that augmented streamflow was possible where annual precipitation exceeded 460 mm (Hibbert 1979).

Research at a number of chaparral sites in Arizona (Three Bar, Whitespar, Mingus, Battle Flat, etc.) demonstrated that runoff could be increased when the vegetation was growing on deep soils or porous parent materials (DeBano et al. 1999). However, research in coniferous woodlands dominated by pinyon and junipers (*Juniperus* spp.) generally did not show increased water yields, even when the entire tree cover was removed (Baker 1984). The best opportunities for increasing water production in the context of multiple resource management were demonstrated to be in the ponderosa pine and mixed conifer forests.

Workman Creek

The initial research in these forests was conducted on the three Workman Creek watersheds in the Sierra Ancha Experimental Forest to evaluate the hydrology of higher elevation mixed conifer forests and to determine the changes in streamflow and sedimentation from manipulating the forest cover (Gottfried et al. 1999a). The treatments were selected to cover the range of water yields possible through manipulating or removing the forest vegetation. Some treatments were designed to obtain basic hydrological information, and others were designed to test common or potential silvicultural prescriptions.

The basic experimental design was to treat the North Fork and South Fork and hold Middle Fork as the hydrologic control. Studies on North Fork were designed to evaluate streamflow responses to clearing the forest cover in stages, starting on the wettest and progressing to the driest sites (Rich and Gottfried 1976, Gottfried et al. 1999a). The first treatment in 1953 removed the riparian vegetation along the stream channel and around springs. In 1958, the mixed conifer stands nearest the channels were commercially harvested and small trees and unmerchantable material were pushed and burned. The final treatment in 1966 was the harvesting of drier site ponderosa pine stands.

The initial treatment on South Fork in 1953 was to evaluate the common single-tree selection prescription. A wildfire burned through part of the watershed in 1957, and the two events resulted in the removal of 45% of the initial stand basal area. The second treatment in 1966 was designed to convert the mixed conifer forest into a pure ponderosa pine forest by harvesting the Douglas-fir and white fir and planting pine seedlings. The eventual goal was to thin the resulting stand to 9.2 $m^2 ha^{-1}$ of basal area to determine if this density would optimize both tree and water production.

The two treated watersheds and the control watershed were burned by the Coon Creek Fire in 2000 and were re-instrumented after a 17-year hiatus. Current research is evaluating the impacts of the severe wildfire on mixed conifer hydrology and erosion and sedimentation.

White Mountain watersheds

Watershed management research moved into the White Mountains of eastern Arizona. soon after initial results from Workman Creek were reviewed. The initial objective was to determine if results from Workman Creek could be confirmed and if they were transferable to other mixed conifer areas (Gottfried et al. 1999b). One concern was that the relatively large openings of more than 32 ha would be unacceptable for multiple-use forest management and compromise prescriptions that benefited water and timber production and wildlife habitat. Three sets of paired experimental watersheds were established on Castle Creek, which supported high elevation stands of ponderosa pine and mixed conifers, and on Willow and Thomas Creeks, which supported dense mixed conifer stands.

The results from Workman Creek indicated that even-aged management could maintain long-term timber production and improve water yields (Rich 1972). The experiments on Castle Creek, where the West Fork (364 ha) was harvested and East Fork (471 ha) served as the hydrologic control, were designed to test this hypothesis. The watersheds are located between 2,388 and 2,616 m in elevation. In 1965, one-sixth of the stand was harvested in irregular blocks fitted to stand conditions and the remaining stand was put into optimum growing condition by thinning and sanitation operations. The idea was to create conditions where trees would achieve a desired size within 120 years and where one-sixth of the stand would be harvested every 20 years. The harvest reduced the stand basal area from 31.0 to 14.5 $m^2 ha^{-1}$. Harvested blocks were planted with ponderosa pine seedlings to ensure adequate regeneration.

A second treatment was initiated in 1981 to test the impacts of pre-harvest prescribed fire as a method of reducing heavy natural fuel loadings. Aggressive fire suppression has been partially responsible for the accumulations of heavy fuel loadings within many southwestern ecosystems that have increased the potential of severe, stand-replacing wildfires. The stable increases in streamflow since 1967 allowed the fire treatment to be applied to East Fork and the West Fork to become the control watershed. The fire reduced surface fuels on approximately 43% of the watershed, but caused little mortality to the overstory.

The second test of the effects of timber management on water augmentation was conducted on the East and West Fork watersheds of Willow Creek, a relatively short distance from Castle Creek. West Fork has an area of 117 ha and East Fork contains 198 ha; elevations on the experimental area range from 2,682 to 2,835 m. The silvicultural prescription on East Fork was similar to the one used on the West Fork of Castle Creek, but the Willow Creek watersheds are at a higher elevation site that is dominated by mixed conifer stands containing an important spruce-fir component. Heavy logging removals, which contributed to excessive wind damage, compromised the original research objectives. The designated openings were often indistinguishable from the areas that had been marked as residual thinned stands; 62% of the watershed was in openings (Gottfried 1983).

Regeneration numbers and stocking recovered because of vigorous quaking aspen suckering.

The third watershed management experiment in the White Mountains was conducted on the two Thomas Creek watersheds that supported an undisturbed, old-growth mixed conifer forest. The South Fork (227 ha) and the North Fork (189 ha) range from 2,560 to 2,835 m in elevation. The objective was to demonstrate and evaluate the knowledge of integrated resource management for southwestern mixed conifer forests (Gottfried 1991).

Beaver Creek watersheds

The southwestern United States experience a drought period during the 1950s, and land owners and managers were concerned that the increases in stand densities in Arizona's ponderosa pine and pinyon-juniper woodlands contributed to reduced surface runoff and amounts of forage for livestock (Fox et al. 2000). A review of existing information determined that replacing high water-using trees and shrubs with low water-using grasses and forbs would increase water yields. However, there were concerns about the effects of vegetative manipulations on other natural resource products.

The Beaver Creek watershed study became a significant component of the effort to evaluate the feasibility of manipulating vegetation by silvicultural treatments to increase water yields and other multiple resource benefits. The Beaver Creek complex encompasses 111,289 ha, south of Flagstaff in the Verde River Basin. Average elevations ranged from 2,054 to 2,225 m. Average winter precipitation ranges from about 550 to 510 mm at Beaver Creek with 60% occurring mostly as snow during then winter (Baker 1986). A multi-discipline team, including forest, wildlife, and range scientists, hydrologists, and economists was assembled in 1960 for the project. A large number of researchers and managers from the Forest Service, other federal and Arizona State agencies and universities cooperated with the Beaver Creek Project throughout the research effort.

Twenty pilot experimental watersheds were instrumented to determine the effects of a variety of land management treatments on stream flow and erosion and on the other natural resources. The 12 watersheds that supported ponderosa pine stands and range from 66 to 730 ha in size are the focus of this discussion. Sub-drainages were also instrumented within the pilot watersheds to refine the findings from the larger watersheds. The remaining eight watersheds included six that were covered by pinyon-juniper woodlands and two large, untreated ponderosa pine watersheds of more than 4,856 ha in size. Since basalt is the main parent material at Beaver Creek, additional watersheds were established on limestone and sandstone sites in eastern Arizona to determine the hydrology of these areas but these were not treated.

Results from the Beaver Creek experiments have been reported in nearly 700 publications, including USDA Forest Service publications, journal articles, and special publications on specific topics, and dissertations and theses (Baker and Ffolliott 1998). The Beaver Creek experiments demonstrated that manipulating the forest vegetation can produce short-term increases in stream flow and that increases generally occur during years with above average precipitation (Baker 1986). However, these increases also occur when the reservoirs are near capacity and it is difficult to effectively control the additional runoff. Vegetation manipulations for runoff augmentation can benefit wildlife, forage production, timber and amenity values. Results from Beaver Creek have not been limited in applicability to the Southwest but are of national and international interest.

Summary

Forest watershed management has been an important aspect of forestry in the southwestern United States since the early 20th century. A watershed research program was initiated in the 1920s to gather fundamental information about the hydrology of forested watersheds. In the future, watershed management will emphasize watershed improvement practices, sustaining riparian ecosystems, and managing watersheds to meet society's growing demands for limited watershed resources. Sound land stewardship, now and in the future, requires partnerships among the general public, land users and watershed managers and a continued investment in watershed science.

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A Proposed International Watershed Research Network

W.R. Osterkamp, J.R. Gray

Abstract

An "International Watershed Research Network" is to be an initial project of the Sino-U. S. Centers for Soil and Water Conservation and Environmental Protection. The Network will provide a fundamental database for research personnel of the Centers, as well as of the global research community, and is viewed as an important resource for their successful operation. Efforts are under way to: (a) identify and select candidate watersheds, (b) develop standards and protocols for data collection and dissemination, and (c) specify other data sources on erosion, sediment transport, hydrology, and ancillary information of probable interest and use to participants of the Centers.

The initial focus of the Network will be on water-deficient areas. Candidate watersheds for the Network are yet to be determined although likely selections include the Ansai Research Station, northern China, and the Walnut Gulch Experimental Watershed, Arizona, USA. The Network is to be patterned after the Vigil Network, an open-ended group of global sites and small drainage basins for which Internet-accessible geomorphic, hydrologic, and biological data are periodically collected or updated. Some types of data, using similar instruments and observation methods, will be collected at all watersheds selected for the Network. Other data from the watersheds that may reflect individual watershed characteristics and research objectives will be collected as well.

Keywords: network, research watersheds, data, monitoring

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Introduction

The Sino-U. S. Centers for Soil and Water Conservation and Environmental Protection were established in May 2002, at the Northwest Sci-Tech University for Agriculture and Forestry, Yangling, Shaanxi Province, China (Gray et al., in press). The Centers provide a formal mechanism to enable cooperation between Chinese and American scientists in developing methods and strategies to conserve soil and water and to maintain environmental protection. Scientists participating at the Centers will conduct research and develop techniques to promote education and outreach of common interest to peoples of the two countries, implemented by mutual understanding and in conformity with their policies.

The Centers are at the Northwest Sci-Tech University of Agriculture and Forestry in Yangling, Shaanxi, China, and at the Institute for the Study of Planet Earth, the University of Arizona, Tucson, Arizona, USA. Each Center manages and funds research conducted from that country. A Joint Oversight Committee (JOC) of members from both countries provides independent scientific review and guidance. Because activities of research, education, and outreach within the earth sciences require reliable landscape data, an initial objective of the JOC is to establish an "International Watershed Research Network" (IWRN), from which hydrologic, geomorphic, agronomic, and biological data, actively collected and archived, will be available to staff of the Centers and to the research community at large.

The IWRN will be comprised of gauged (instrumented) sites and small watersheds, generally in, but not limited to, water-deficient areas, where relatively long-term records of reliable (validated) data have been collected. Initial focus is likely to be on established research sites and watersheds in China and the United States. Establishment of an IWRN requires agreement among Chinese and American representatives for criteria used for watershed selection, protocols for the collection of data and the unconstrained availability of data, responsibility for data collection, network construction, and testing of protocols at selected watersheds. Information describing the principal worldwide data sets that may assist research conducted under the auspices of the Centers will be compiled, in part to aid in the selection of additional watersheds for the IWRN, and partly to augment databases for research conducted under the auspices of the Centers.

To address these prerequisites for establishing a network of research watersheds, interaction among Chinese and American scientists and representatives from other countries is essential. Thus, the general purposes of this paper are to inform the international watershed-research community of these intentions and to solicit worldwide input from scientists who desire to participate in Network development. The IWRN is to be a source of global information for assessing and mitigating physical, chemical, and biological sediment damages, estimated to exceed 16 billion dollars annually in North America alone (Osterkamp et al. 1998). The principal goal, therefore, is to accumulate consistent, validated (quality-assured) watershed data, both actively collected and archived, for use by researchers of the Centers and by the global research community in general.

An International Watershed Research Network: Description and Implementation

Watersheds comprising the initial Network are expected to be up to approximately 1000 km^2 in size; data collection at each will be conducted using established standards for observations of change, management and compilation of watershed information, and data access. This approach applies to the strategies of observation, the objectives of ongoing research, the instruments used, and the procedures used, such as frequency of observations in time and space. The intent of these standards is to ensure an ability for intercomparison of watershed data and processes worldwide. Prior to establishment of the Network, efforts will be made to: (1) identify and select candidate watersheds for the Network; (2) identify acceptable, and preferably, consistent instrumentation and techniques for the collection, dissemination, and archival of data; and (3) specify other data sources on erosion, sediment transport, hydrology, climate, vegetation, and other information of probable interest to and use through the Centers. In preparation to implement the IWRN, the following activities must be completed:

- establishment of selection criteria for watersheds that are either presently gauged or to be gauged by representatives of the Centers and their designated points of contact
- assessment of the candidate watershed observation and research plan to represent the watershed processes
- determination of standards for the types and quality of instrumentation to be maintained or installed at the various gauged watersheds
- determination of protocols for data assembly, management, and access, and for designations of responsibility
- selection of a small number of charter watersheds to be included in the Network
- identification of reliable, quality-assured data sets that may contain erosion, sediment-transport, hydrologic, and related information useful to Centers researchers, and that may help guide selection of additional watersheds for the Network
- planning for the testing of protocols at selected watersheds and the development of a consistent means for storage and dissemination of watershed data

Among the criteria to be established for the selection of watersheds in the IWRN will be: a satisfactory watershed research and observation plan; a record of the types, uniqueness, length, amount, and quality of data available at a watershed; documentation of the size (area) of the watershed, its climate, the country(s) involved, the availability of logistical support at or near the watershed, and other relevant information such as presently occurring natural and induced surface processes. Priority will be given to paired, or comparative, watersheds in China and the United States that, for example, are similar in all respects except for land-use practices. Regardless of the criteria established, the intent will be to require that data from the watersheds included are comparable and permit consideration of critical research issues, or questions, of watershed management, especially of arid and semiarid regions in or relevant to China and the United States. These questions include but are not limited to identifying what the water and sediment yields of the representative watersheds are under natural conditions and how land-use practices have altered these yields, how different agricultural practices affect water and sediment movement to stream channels in landscapes of specified characteristics, how rampant erosion in arid/semiarid watersheds can be reduced by re-vegetation after land-use stresses have caused gully initiation, which dryland crops provide the best resistance to erosion under specified conditions of climate, soil, and topography, and which practices of hillslope modification (such as terracing, construction of levees, contour tillage) are most likely to minimize erosion under specified conditions of climate, soil, and topography, thereby leading to landscape rehabilitation.

Although these questions are basic, the solutions are exceedingly complex; all have been addressed by previous studies, but no question has been answered comprehensively. The development of a global data resource – through a watershed research network -- is necessary if the questions are to be answered satisfactorily.

Implementation of an IWRN must incorporate a means to provide information freely to the watershed-research community. Data compiled from the proposed IWRN are intended principally as a resource for the Centers, but also will be available to all natural scientists via an established USGS web site for the Vigil Network (http://wwwpaztcn.wr.usgs.gov/vigil/) (Orr and Osterkamp 1999). The Vigil Network, which was begun and continues to be managed by the USGS (Leopold 1962, Osterkamp and Emmett 1992), is a system of sites and small drainage basins where long-term geomorphic, hydrologic, and biological data are collected periodically, and the information is added as an update to the accumulating data file. Although data collected for the IWRN will vary with local conditions, as has been the case for Vigil Network records, certain standard data types will be collected at all watersheds of the Network and will likely include measures of climate, geology and soils, topography, vegetation, and land use.

Existing Databases

Databases relevant to the Centers' objectives are available from many countries. For example, as of January, 2000, the NWISWEB database of the USGS (Turcios et al. 2000, Osterkamp and Gray 2001, Turcios and Gray 2001) contained about 15,400 sites in the United States and Puerto Rico for which fluvial-sediment data are available. The most useful databases should be identified and information on their characteristics and availability should be obtained for selecting IWRN watersheds, and to represent a resource for scientists of the Centers.

Research Watersheds and Network Design

Initially the IWRN will be comprised of established watersheds and sites in semiarid, water-deficient areas where relatively long-term, high-quality data have been collected and compiled. Following this criterion alone, the Ansai Research Station, northern China, the Walnut Gulch Experimental Watershed, at Tombstone, Arizona, USA, and the Eshtemoa Experimental Watershed, northern Negev Desert, Israel, are candidates, as are presently unspecified watersheds in Australia, India, and northern Mexico.

The Ansai Research Station (36° 51' 39"N, 109° 19' 23"E) of the Institute of Soil and Water Conservation, Chinese Academy of Science, and the Ministry of Water Resources was established in 1973 and became a member of the Chinese Ecosystem Research Network in 1991. Isohyetal maps of the area, based on long-term data, suggest that mean annual precipitation at the town of Ansai is about 450mm, but shorter-term records collected at the research station indicate that precipitation averages 498 mm (Guobin Liu, Chinese Academy of Sciences, written communication, 2003). The station is in the highly erodable gullied region of the Loess Plateau (mean annual sediment yield about 13,500 t/km^2), has an elevation of 1010 to 1431 m, and, before major anthropogenic modification, had temperate forest-steppe vegetation (Guobin Liu, Chinese Academy of Sciences, written communication, 2003).

The Walnut Gulch Experimental Watershed of the U. S. Department of Agriculture, Agricultural Research Service, was established in 1961 as a research site representative of 600,000 km^2 of semiarid rangeland in the southwestern United States. The watershed (31° 43' N, 110° 41' W) is 149 km^2 in area, has an elevation ranging from 1250 to 1585 m, and mean annual precipitation (at Tombstone, Arizona, in the west-central part of the watershed) of 324 mm. Vegetation is mostly shrubs and grasses. Instrumentation includes numerous rain gages and 11 runoff flumes; data are collected intensively in 12 sub-basins in which rainfall, runoff, and sediment yield are related to land use through erosion modeling (Renard et al. 1993; Alonso 1997).

The 112-km^2 Eshtemoa watershed (31° 22' N, 34° 54' E) in the northern Negev Desert, Israel, has been in operation 10 years. Native vegetation, which has been altered through land use, consists mainly of herbs, thorny shrubs, and, in the upper parts of the watershed, pines. Mean annual precipitation is 280 mm and the main gravel-bed channel drains soils derived principally from limestone, chalk, and loess exposed on the southwest flank of the Hebron Mountains (Reid et al. 1995). Available information includes event-based bed-load data from sub-basins of the catchment.

The process of site and watershed selection, based on agreed protocols for data collection and the implementation of those protocols by representatives of the JOC, will be extended to include a network design. Watershed monitoring involves repeated observation through time at a site or watershed, generally to facilitate the regulation or control of those operations for which the time-series data are collected. In contrast, a network defines areal variability of the measured properties and, therefore, is designed to identify change in both space and time.

Thus, site and watershed selections and the data collected at the watersheds must be compatible with network objectives, permitting a global analysis for each data type compiled. Descriptions of the various global data sets, and analyses of those data sets, relevant to the Centers' interests will be compiled and entered in the USGS–Vigil Network web site, partly to aid in selection of watersheds for the Network, and partly to add resources for the Centers' research. Information about the Sino-U. S. Centers may be found on the Centers' web site http://www.ispe.arizona.edu/sino.

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Descriptions of the Ansai Research Station, and permission to include it as a possible part of the intended Network, were provided by Guobin Liu, Chinese Academy of Sciences. Information of the Walnut Gulch Experimental Watershed was supplied by colleagues of the Agricultural Research Service, Tucson, AZ, and papers describing the Eshtemoa watershed were provided by Jonathan Laronne, Ben Gurion University of the Negev, Beer Sheva, Israel. Very helpful reviews of drafts of the paper were given by William Sprigg, Institute for the Study of Planet Earth, University of Arizona, and by Craig Perl, Office of Surface Water, U. S. Geological Survey.

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History of Small Watershed Research in Non-Forested Watersheds in Arizona and New Mexico

K.G. Renard, M.H. Nichols

Abstract

Small watershed research to support land use and conservation technology on non-forested areas in Arizona and New Mexico has a long history with many reasons for failure. The programs originated from need for solutions to land degradation in arid and semiarid areas of the southwestern United States. Speculation was that degrad-ation resulting from intense grazing of delicate vegetation and that other land uses led to excessive erosion and loss of soil resources. Efforts began in the early 1930s to establish experimental programs on watersheds to understand hydrologic and erosion phenomenon. Technical and administrative factors leading to research termination are discussed with emphasis on the importance of long-term programs to ensure conservation of semi-arid lands.

Keywords: hydrology, watersheds, rainfall, runoff, semiarid

Introduction

Watershed research to quantify rates and amounts of the factors in the hydrologic cycle and the consequences of varying land uses on environmental conditions became a research priority throughout the U.S. in the mid-1930s, including Arizona and New Mexico. The early research programs in the western United States were often complicated by attempts to apply measurement technology that worked in more humid areas to the unique conditions of semiarid regions. In sparsely vegetated areas such as those in the southwestern United States, runoff measurement was often difficult/inaccurate because of heavy sediment loads which confounded measurements. The efforts are summarized in this paper including information on land uses, and some information on why the work was terminated. While Walnut Gulch is currently one of the premier semiarid experimental watershed in the world, the research program has benefited greatly from earlier watershed studies in Arizona and New Mexico. A summary table lists the characteristics of research sites described below (Table 1).

Mexican Springs, New Mexico

Research on the Navajo Experiment Station began in 1934 in an effort to provide information on water spreading and the role of land use on water rates and amounts. Work at the station included soil conservation demonstration areas and instrumented watersheds (see Table 1). The watersheds located in the Navajo Section of northwestern New Mexico were the first on larger instrumented watersheds in the Southwest and were among the earliest research watersheds in the United States. A comparatively dense rain gage network with runoff measurements from integrated subwatersheds provided data critical to quantifying rainfall-runoff-erosion information in a comprehensive way. The Soil Conservation Service Division of Operations and Research operated the watersheds. Progress of research at the location is given in Lowdermilk (1936).

Soil Conservation Service Experimental Watersheds

The Soil Conservation Service (SCS) in the U.S. Department of Agriculture initiated early watershed studies across the U.S. in collaboration with the Soil Erosion Service (a USDA agency that preceeded the formation of ARS in 1954). This early precipitation and runoff work often also included soil erosion

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measurements. Most of the instrumentation at these sites was selected to represent the climate-topography-cropping (land use) combinations on the lands of the U.S.. The instrumentation was constructed in the late 1930s by workers in the Civilian Conservation Corp (CCC camps) and used broadcrested V-notch weirs (Brakensiek et al. 1979, Johnson et al. 1982). The use of such weirs and design considerations is described in Ruff et al. (1977). The weirs were subsequently found to be inadequate for accurate measurement in highly erodible areas where sediment accumulation negated weir principles of neglible velocity head at the depth measuring point.

Table 1. Experimental watersheds - Arizona and New Mexico.

Location / City	Altitude (ft.)	Predominant Land Use	Cover	Record Years	Number of Watersheds	Responsible Group
1. Navajo Experiment Station						
(20 mi N of Gallup, NM) Mexican Springs	5000-6000	Grazing	Mixed Grass-Brush	1934-50	12	USDA-Plant Industry and Navajo Indian Nation
2. Albuquerque Watersheds						
(40 mi NW of Albuquerque – middle Rio Grande)	5000	Grazing	Grassland	1939-75	3	USDA-SCS and Laguna Indian Reservation
3. Alamogordo Creek						
(40 mi East of Santa Rosa, NM)	4500-5500	Grazing	Mixed Grass-Brush	1954-79	3	USDA-SCS USDA-ARS
4. Fort Stanton Watershed						
(70 mi SE of Capitan, M)	6000	Grazing	Mixed Grass	1966-83	3	USDA-ARS NMSU
5. Jornada Experimental Range						
(25 mi N of Las Cruces, NM)	4000-5000	Grazing	Mixed Brush	1906-Present	2	USDA-ARS
6. Safford Experimental Watersheds						
(circle of 40 mi from Safford, AZ - San Carlos Basin)	4000-5000	Grazing	Mixed Grass-Brush-Cacti	1939-75	4	USDA-SCS USDA-ARS USDI-BLM
7. Walnut Gulch Experimental Watershed						
(Tombstone, AZ)	4200-5000	Grazing-Urban	Mixed Brush-Grass	1954-Present	>20	USDA-SCS USDA-ARS
8. Santa Rita Experimental Range						
(30 mi E of Green Valley, AZ)	3800-4500	Grazing	Mixed Brush-Grass-Cacti	1975-Present	8	USDA-ARS Univ. of AZ
9. Santa Fe, NM						
(Upper Rio Grande basin)	3200-3500	Grazing	Sparse grass	1940-47	3	USDA-SCS
10. Atterbury, AZ						
(East Tucson)	2560	Grazing-Urban	Sparse grass -Brush	1955-70	5	U of A Water Res.
11. Tucson, AZ						
(Tucson Metro)	2400	Urban	Mixed Landscape	1971-80	3	U of A Water Res.

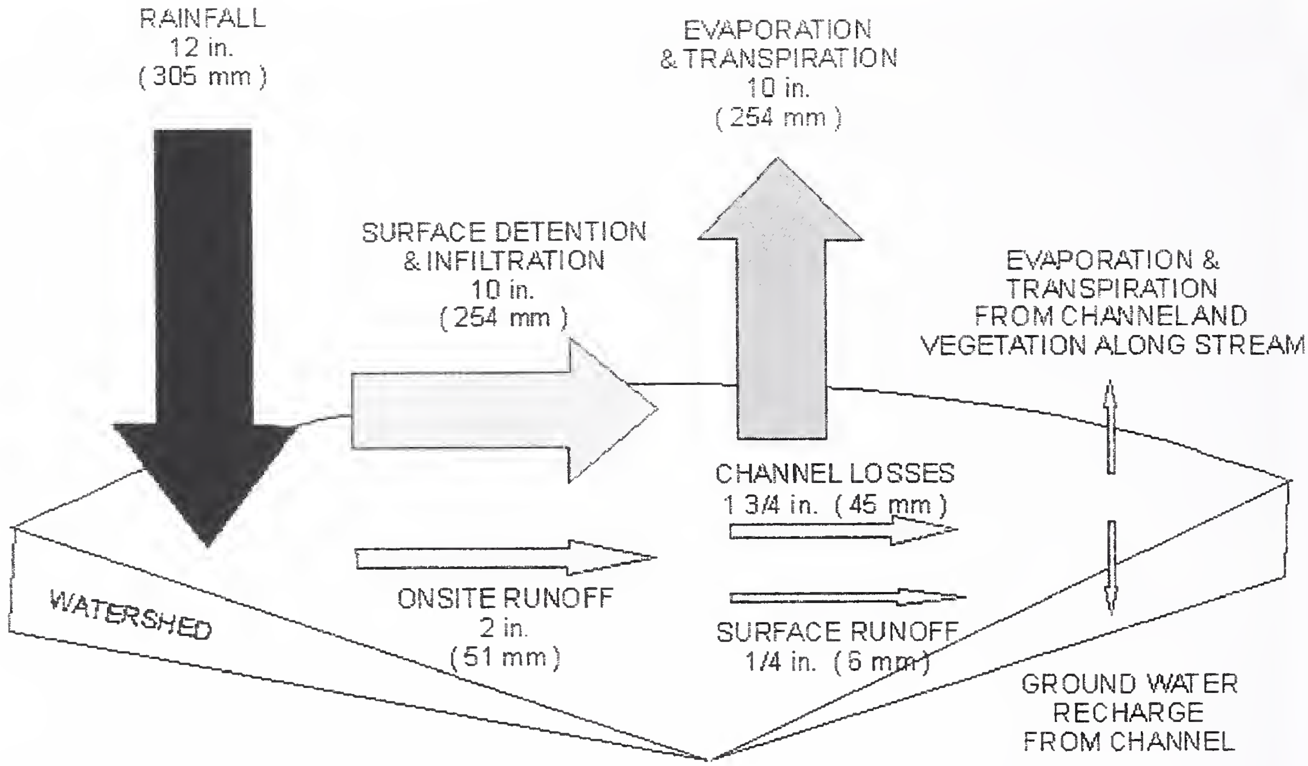

Figure 1. Annual water balance for the USDA-ARS Walnut Gulch Experimental Watershed (Renard et al. 1993).

In 1979, the research program on Alamogordo Creek was terminated because of resource and staff limitations. This unfortunate scenario restricted valuable information on thunderstorm dynamics in eastern New Mexico that are often quite different from those in southeastern Arizona (Osborn et al. 1980).

Many major conclusions have been reported from the Walnut Gulch Experimental Watershed Program. These include:

1) Precipitation amounts and models developed to describe such (Osborn 1983; Osborn et al. 1980).
2) Instrument developments to monitor the hydrologic and erosion cycle in semiarid areas (Renard et al., 1993).
3) The role and magnitude of transmission losses in ephemeral streams (Lane 1990).

The precipitation and runoff monitoring network and the innovations and research results have allowed researchers to prepare a water balance for Walnut Gulch that is typical of semiarid rangeland watersheds (Figure 1). Research and data collection continues at the WGEW and the core hydrologic and erosion monitoring networks have expanded to include meteorologic and remotely sensed data.

Conclusions

Considerable precipitation-infiltration-runoff data are available in the southwestern arid watersheds in AZ-NM as described herein. Published material describing such work is extensive, especially as it pertains to precipitation and the research of Walnut Gulch Experimental Watershed.

Experimental watersheds with comprehensive monitoring and measurement infrastructure are a critical resource for research to understand semi-arid hydrologic, geomorphic, and ecosystem processes. The long-term data sets collected at the Walnut Gulch Experimental Watershed and at other experimental watersheds in Arizona and New Mexico have been used throughout the world to quantify rainfall and runoff relationships, to develop runoff and sediment yield prediction technologies, and to support soil and water conservation projects. The continued support of the currently operating experimental watersheds is critical to future soil and water conservation efforts.

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Historical and Current Hydrological Research at the USDA/ARS Jornada Experimental Range in Southern New Mexico

Albert Rango, Keirith Snyder, Jeff Herrick, Kris Havstad, Robert Gibbens, John Wainwright, Tony Parsons

Abstract

The USDA/ARS Jornada Experimental Range (JER) (738 km^2), north of Las Cruces, NM, was established in 1912 to assess the impact of grazing in an arid land environment. The majority of rainfall occurs during June-September with an annual mean of 241 mm. Ecophysiological studies employing stable isotopes are underway to identify the sources of water uptake for shrubs and grasses and how the temporal and spatial variability affects the amount and sources of water used by various species. Infiltrometer and rainfall simulation studies are being used to quantify the role of soil biota in controlling soil surface hydrology in arid and semi-arid environments, and to define the resistance and resilience of different soils and plant communities to different disturbance regimes. Previous work at Jornada quantified the interception of rainfall for different shrubs and infiltration rates in rootplowed areas. Runoff was measured with a 2.8 m^3/s critical depth flume on a shrub dominated 7.4 ha watershed from 1977-1986. These flow measurements were re-activated in 2003. Runoff and sediment measurements were also made on plots, microwatersheds, and stock ponds. Because of the aridity of Jornada, there have been numerous rangeland rehabilitation treatments with the goal of slowing or reversing the shrub encroachment into grasslands. The most effective treatments have revolved around redistribution of surface runoff and its effects on infiltration and soil moisture. Simple, low profile, water ponding dikes seem to have had the best success in achieving a positive vegetation response.

Keywords: Jornada Experimental Range, grazing, rangeland rehabilitation, watershed studies, ecophysiology

Introduction

The U.S. Department of Agriculture (USDA), Agricultural Research Service (ARS), Jornada Experimental Range (JER or Jornada) in Southern New Mexico has a long history of research, experimentation, and monitoring on rangeland vegetation change. The JER was established in 1912 with some research plots and data records having been maintained since then. In 1977, the site was selected as a Biosphere Reserve as part of the United Nation's International Man and the Biosphere program. In 1981, the National Science Foundation selected Jornada as a Long-Term Ecological Research (LTER) site. These different programs and research efforts at Jornada have produced a 90+ year history of ecological research on processes related to vegetation change, desertification, hydrology, and range management.

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The Jornada Experimental Range (783 km^2) lies 37 km north of Las Cruces, NM on the Jornada del Muerto Plain in the northern part of the Chihuahuan Desert. It is located between the Rio Grande floodplain (elevation 1,186 m) on the west and the crest of the San Andres Mountains (2,833 m) on the east. Jornada is typical of the Basin and Range physiographic province of the American Southwest and the Chihuahuan Desert. The Jornada basin is an internal drainage system with no surface runoff exiting the basin to the Rio Grande.

The climate of Jornada is characteristic of the northern region of the Chihuahuan desert with abundant sunshine, low relative humidity, wide ranges of daily temperature, and variable precipitation both temporally and spatially. Precipitation, which averages 241 mm yr^{-1}, mainly occurs as localized thunderstorms during July, August, and September. The average monthly maximum temperature ranges from 13°C in January to 36°C in June. Potential evaporation is approximately 10 times the average precipitation.

Jornada is the most arid North American grassland. Grasses on the plains are entirely C4 plants. Shrubs and suffrutescents are commonly C3 plants. More than 490 plant species have been identified on Jornada.

Grass communities of black grama [*Bouteloua eriopoda*], which once dominated the landscape, have been susceptible to encroachment by shrubs during the last century. Vegetation surveys made in 1855, 1915, 1928, and 1963 show that total area dominated by grass had decreased from 90% in 1858 to 23% in 1963 (Buffington and Herbel 1965). Droughts, grazing by livestock and native fauna, and shrub seed dispersal by livestock have all contributed to the spread of shrubs. Conversion from grass-dominated to shrub-dominated vegetation on these deep coarse texture soils characteristically has resulted in the formation of coppice dunes (Buffington and Herbel 1965), resulting in increased spatial heterogeneity of critically limited nutrients (especially N) required for plant growth (Schlesinger and Pilmanis 1998) and increased wind erosion.

To complement the programs of ground measurements, a campaign called JORNEX (the JORNada EXperiment) began in 1995 to collect remotely sensed data from aircraft and satellite platforms to provide spatial and temporal data on physical and biological states of the Jornada rangeland. A wide range of ground, aircraft, and satellite data have been collected on the physical, vegetative, thermal, and radiometric properties of the major ecosystems (grass, grass/shrub transition, and shrub) typical of the Jornada rangeland and of southwestern U.S. deserts. Data from different platforms have allowed the evaluation of the landscape at different scales. These measurements are being used to quantify hydrologic budgets and plant responses to change in components in the water and energy balance at Jornada. Data have been acquired twice a year from 1995 through 2003.

Sources of Plant Water Use

The interactions between plants and hydrology are critical in aridland systems. Plants are a pathway for water transport from soil to atmosphere. Plants determine the subsurface location of water lost to transpiration by having roots accessing water stored in various soil layers. Plant available water is water stored within the soil profile at soil water potentials that plants can extract. Differences in soil characteristics and topography create heterogeneous patterns of plant available water across the landscape. Highly variable rainfall events may also make water temporarily available to plants in shallow soil layers. The ability of different plant species or plant functional types (e.g. shrubs versus grass) to use these various water sources is a function of plant characteristics such as plant age, rooting characteristics, and carbon assimilation capacity. Plant water source use also is also a function of environmental factors such as the relative availability of water sources and the recent patterns of absolute soil moisture. The ability of various plant species or plant functional types to exploit various water sources will in part determine species success and competitive interactions.

We sampled dominant shrub species at the JER on various geomorphic surface and soil types throughout the growing season to determine which water sources the three dominant shrub species tarbush [*Flourensia cernua*], honey mesquite [*Prosopis glandulosa*], and creosotebush [*Larrea tridentata*] were using. Water source use was determined with stable isotopic methodology. Rainfall in this region is bimodal, and because of this seasonal variation, the natural abundance ratios of hydrogen and oxygen stable isotopes vary considerably among different source waters. Winter rainfall is isotopically distinct from summer monsoon rainfall. Winter rainfall percolates deeper and is the primary source of recharge for deep soil water. Summer rainfall is usually in isolated storms that recharge only shallow soil water. Since there is no isotopic fractionation during plant uptake of water from the soil (White et al. 1985, Ehleringer and Dawson 1992, Brunel et al. 1995), stable isotope methodology is a tool for determining plant water source (Ehleringer and Dawson 1992).

Stable isotopic methodology determines the ratio of heavy hydrogen (deuterium) to light hydrogen (H^2:H) in extracted water samples. Stable isotope ratios of hydrogen in water are expressed using delta notation (δ) in parts per thousand (‰) as:

$$\delta D = (R_{sample}/R_{standard} - 1) \times 1000 \quad \text{(Eq. 1)}$$

where R_{sample} and $R_{standard}$ are the molar ratios of D/H or $^{18}O/^{16}O$ of the sample and standard water. We extracted plant and soil water with cryogenic vacuum distillation (Ehleringer and Osmond 1989, Smith et al. 1991). Plant water and soil water samples were analyzed for hydrogen isotope ratios (δD) using a dual inlet isotope ratio mass spectrometer (Delta-S, Finnigan –MA, Bremen, Germany). A chromium reduction furnace attached on-line to the mass spectrometer was used to convert liquid water to hydrogen gas (HD-Device, Finnigan-MAT, Bremen, Germany). Plant water was compared with soil water values to determine from which soil layers plants obtained their water.

Preliminary results indicate species-specific differences in water source use. Creosote, which grows on gravelly sandy loam soils on the JER, appears to be highly responsive to summer rainfall and resultant pulses of shallow soil moisture. Mesquite, growing on deep loamy fine sands and loamy sands, did not appear to use substantial amount of shallow soil water and relied primarily on water stored deeper within the soil profile. In addition to these preliminary results, analyses of a thirteen-year data set on soil moisture patterns collected from the Jornada LTER showed as much variability within three replications of these different community types as between the different communities in patterns of volumetric soil moisture measured with a neutron probe.

Effects of Soil Biota

Tension and single ring infiltrometers are being used together with rainfall simulation to quantify the role of soil biota in controlling soil surface hydrology in arid and semi-arid environments, and to define the resistance and resilience of different soils and plant communities to different disturbance regimes. We used tension infiltrometers to quantify the effects of macropores generated by ants and termites on water infiltration capacity. The effects of soil biota on hydrology are also being evaluated as part of a long-term study on the effects of disturbance. This study was initiated in 1997 and is replicated on five different soils in southern New Mexico. We are using 12.5 cm single-ring constant-head infiltrometers to quantify relative changes in infiltration capacity in canopy and intercanopy zones in response to single and repeated trampling by humans and livestock, and to off-road vehicle traffic. Small-plot (0.5 m^2) rainfall simulation is used to calibrate the infiltrometers within a soil series, where possible, and to calibrate a field soil aggregate stability kit and to explore the relationship between soil aggregated stability, soil microbiotic crusts and erodibility. We are also using small plot rainfall simulation to define the effects of antecedent soil moisture on infiltration capacity reduction by off-road vehicles.

The tension infiltrometer studies showed that both termite and harvester ant activity significantly increases soil water infiltration capacity, and that much of the increase is due to macropore formation. Preliminary results from the long-term surface disturbance studies show that the effects on soil surface hydrology depend on disturbance type, timing, frequency, and intensity, and that these effects vary with soil type and vegetative cover. Soil aggregate stability values from the field kit (Herrick et al. 2001) are negatively correlated with sediment production from small-plot rainfall simulation. Disturbance of wet soils reduces infiltration capacity more than disturbance of dry soil, as predicted by theory and data from cultivated systems.

Preliminary results of the long-term, comprehensive study on the effects of different disturbance regimes on soil surface hydrology demonstrates the importance of integrated, multi-factor, long-term experiments applied across multiple soils: in most cases, the variable effects among sites can be explained by interactions between site characteristics and one or more characteristics of the disturbance regime. The results of the study will be analyzed and published following collection of an additional data set at all 5 sites in fall 2003 and spring 2004. The soil stability kit (Herrick et al. 2001) appears to be relatively sensitive to short-term changes in soil erodibility associated with dynamic soil carbon fractions, supporting its use in rangeland monitoring.

Vegetation Influence on Hydrologic Cycle Components

Tromble (1988) reported on a comparison of rainfall interception by creosotebush and tarbush at Jornada using rainfall simulators. Native stands of creosotebush had 30% crown cover and rainfall loss by interception was approximately 12%. Tarbush had 15% crown cover and intercepted about 6% of the rainfall. Infiltration rates were measured over undisturbed creosotebush stands and areas where creosotebush was rootplowed and seeded. Infiltration rates were greater on untreated plots than treated plots (Table 1). This demonstrated the potential for increased surface runoff and erosion from areas not adequately protected with vegetation cover, especially right after treatments have been performed (Tromble 1980).

Table 1. Difference between treatment means for infiltration rates after 60 minutes (Tromble 1980).

	Treatment **			
	CB wet	RP2 wet	RP6 dry	RP6 wet
Means (cm/hr)	1.39*[a]	1.80[a]	2.25[a]	3.39[ab]
	CB dry	RP2 dry	CC wet	CC dry
	4.68[bc]	5.51[bc]	6.72[c]	9.41[d]

Values followed by the same letter indicate no significant difference at the .95 level treatment means according to Duncan's multiple range test. ** CB = control, bare soil; RP2-rootplow, 1972; RP6-rootplow, 1976; CC = control, cresotebush.

Runoff Measurements

A 7.4 ha rangeland watershed dominated by creosotebush and mesquite on the eastern side of Jornada was first gauged in 1977 with a 2.8 m^3/s critical depth flume installed with assistance of ARS personnel at the Southwest Watershed Research Center in Tucson, AZ. The gauge was deactivated after 1986. During the 10 year period, sediment samples were collected with Coshocton wheels during each runoff event for four years. For the 10 year period of record, on average 6 storm flows a year were produced by precipitation over the watershed with almost all occurring during the summer months. The flow measurements were discontinued until May 2003 when the flume was reactivated with remotely telemetered data.
In the seven-year period, 1988-1994, plot runoff was measured from shrubland (dominated by creosotebush) and grassland (in black grama areas) as part of a study on nutrient losses in the Chihuahuan Desert of southern New Mexico. Runoff began at a lower rainfall threshold in shrubland than in grassland. In the shrubland, the runoff coefficient was 18.6% over the seven-year period. In contrast, in two different types of grassland plots, the runoff coefficient ranged from 5.0–6.3% (Schlesinger et al. 2000). The runoff plot dimensions were 2x2 m, and the plots were surrounded by a metal frame on three sides to prevent overland flow from crossing the plot. The total annual runoff averaged 57 mm from the shrubland runoff plots and only 15 mm from the grassland plots. In these studies, only natural rainfall events were studies. Additionally, rainfall simulation experiments were performed on other plots located within grassland, creosotebush shrubland, and mesquite dunefields. Rainfall intensities for these studies were typically 144 mm/hr.

Two small watersheds on the alluvial piedmont, both within creosotebush shrubland, have been instrumented to monitor natural runoff events. These instrumented watersheds have been collecting data since 1995. One watershed may be characterized as a typical 'dendritic' network in which two tributaries join to form the master stream of the catchment. Flow in both the tributaries and the master stream has been monitored. The second watershed characterizes the discontinuous drainage pattern, which typically occurs on alluvial piedmonts. In this case, two instrumented tributaries discharge into an area of diffuse flow. The outflow from this area of diffuse flow is also monitored. These areas of diffuse flow are significant sinks of runoff, particularly for small events, and appear also to be 'islands of fertility' favorable for plant growth (Wainwright et al. 2002).

In addition to the plot-based field experiments, we have undertaken field experiments to determine (i) the effects of creosotebush on rainfall energy and disposition of rainfall, and (ii) transmission losses in rills on the alluvial piedmont of Summerford Mountain. In 2001, ninety-six miniature flumes and bedload samplers were installed throughout the Jornada Experimental Range. This instrumentation is designed to sample the fluxes of interrill water, nutrients and sediment across ecotones. In 2002, instruments to record maximum flow stage and to sample water for nutrient analysis were installed in 11 rill locations. In 2002, five stock ponds located to represent the range of vegetation communities – creosotebush, tarbush, mesquite, grass, and creosotebush/mesquite mixed – were instrumented to record rainfall and rate of water inflow. The stock ponds allow the investigation of the integration of fluxes at larger scales.

Vegetation plays a very significant role in the hydrology of the Jornada Basin. Beneath creosotebush, mean rainfall intensity was reduced by up to 90% of that falling outside the canopy in rainfall simulation experiments, and mean kinetic energy was reduced by 30% (Wainwright et al. 1999). The reduction in kinetic energy is particularly important, because it weakens sealing beneath the shrubs, thereby enhancing infiltration, compared to intershrub areas. In contrast, these shrubs also direct much of the rain falling onto their canopies through stemflow to a small area of the ground surface surrounding their root crown. In this locality, the rate of stemflow is so high that the local infiltration rate is readily overwhelmed. During high-intensity storms, therefore, a high proportion of stemflow will run off as overland flow (Abrahams et al. 2003). Data for mesquite dunefields are available only for high intensity rainfall-simulation experiments, which indicate an average runoff coefficient at 41.8% at 144 mm/hr (Parsons et al. 2003). However, given the very low infiltration rates for sealed interdune surfaces, this figure suggests that mesquite dunefields may have the highest runoff coefficient of the three vegetation communities.

Rangeland Remediation Treatments

In the 1930s and early 1940s, extensive rangeland treatments were carried out in the Jornada basin in an attempt to reverse the advance of shrubs and re-establish grass dominance. The extensive treatments were conducted by hand, thanks to the presence of a highly organized and an inexpensive labor force in the form of the Civilian Conservation Corps (CCC) in the Jornada basin. After it became evident to Jornada scientists, that reduction of grazing alone would not cause a return to grass, a number of attempts were made to modify surface runoff patterns to slow the runoff and increase infiltration and surface soil moisture, with the hopes of encouraging grass growth. The common treatments using this approach were contour terraces, brush water spreaders, check dams for water redistribution, rootplow seeding, water ponding dikes, and water spreader systems.

Contour terraces and Brush water spreaders were installed throughout Jornada in the 1930s with little or no maintenance of these treatments, they both had lost effectiveness within about 35 years (Rango et al. 2002).

After the CCC period, numerous implements were designed to exploit the power of agricultural and civil engineering machinery to remove shrubs, prepare seedbeds, create small pits where water could accumulate, and plant seeds. A machine which accomplished all these operations in a single pass, was developed at Jornada in 1967 and tested during the 1960s and 1970s (Herbel et al. 1973, Abernathy and Herbel 1973). Commonly know as the "arid land seeder," it consisted of a 2.4 m wide rootplow pulled by a Caterpillar D-7. A 1.2 m wide conveyor with two side delivery rake wheels to gather severed shrubs from the 2.4 m rootplow swath was pulled behind the rootplow which picked up the severed shrubs and elevated them in the air. A power-operated blade was mounted underneath the conveyor to gouge out shallow basins about 3.6 m long. A press wheel seeder able to handle both small and chaffy seed planted grasses in the basins, and the shrubs dropped off the conveyor to provide temperature-reducing shade for emerging grasses.

In 1972 approximately 9 ha dominated by creosotebush was treated on the JER with the arid land seeder. An excellent stand of seeded grasses was obtained. Production in 1978 was 997 to 1,566 kg/ha compared to practically nothing on untreated areas. Contour strips 7.3, 14.6, and 29.2 m wide with either a 1-1 or 2-1 watershed above the strips were rootplowed and seeded in 1976. In 1978 the water harvesting strips yielded 2,817 kg/ha of forage vs. practically nothing on a control area. However, these treatments resulted in relatively high levels of soil surface disturbance that increased erosion susceptibility (Herrick et al. 1997).

Range water spreaders are systems of dikes and berms constructed to automatically divert flood water flows from gullies or arroyos, and to spread flow over adjacent rangeland to promote a positive vegetation response. In 1974, a serpentine water spreader was constructed below Yarbrough Dam in the southeastern part of JER in the foothills of the San Andres Mountains. Also, a diversion dam was constructed in the adjacent Lion's Den Canyon to divert more water into Yarbrough Dam. Operations were started in 1976, only to cease in 1977 because of a leak in the dam. Despite operating for just a year, vigorously growing vegetation was recorded on aerial photography where soil moisture was increased because of ponding and retention of water for longer time periods than allowed by natural, flashy runoff in these arid regions.

At about the same time as the construction of the Yarbrough Dam water spreader, smaller water ponding structures were being installed at four other sites at Jornada from 1975-1981. These water ponding dikes were of varying sizes and heights, usually about 60m in length and from 7.5-30 cm high. Various associated treatments were combined with the dikes including rootplowing, seeding, and application of municipal biosolids. These experiments were abandoned by 1984 because the principal investigator retired, there was little plant response, and the dikes required periodic maintenance. But, when the dikes were re-examined and re-measured in 1997 (Walton et al. 2001), significant recovery of native species was noted. In arid regions, significant rainfall events and subsequent runoff are not frequent, so it may be necessary to maintain treatments for longer periods than would be necessary in humid regions in order to encounter the precipitation events that would activate the treatments or support plant response.

Future Work

Future work in sources of plant water will investigate the water use behavior of mesquite and black grama across a larger range of soil type with differing depths to petrocalcic. Experimental additions of precipitation to plots of mesquite and black grama will be employed to understand the minimum storm size requirements for species-specific response and the importance of petrocalcic depth and development on plant community structure. We plan to continue to work on the effects of disturbance regime on soil surface hydrology, and to expand it to determine how disturbance at different spatial scales can be used to promote the restoration of degraded systems. A new project designed to define how petrocalcic horizons control soil water availability to different plant species was initiated in 2003. The focus of future runoff measurements is *i*) monitoring of natural events, and *ii*) upscaling of field measurement and monitoring to the landscape scale. Investigations will be made to determine the best ways to conduct spatially explicit sampling of surface properties and soil moisture in order to scale up to landscape units.

The most effective rangeland treatments to provide a positive rangeland vegetation response in the Jornada basin seem to revolve around the manipulation and redistribution of the surface runoff process, which in turn affect infiltration and soil moisture. Some of the best example rehabilitation treatments (water ponding dikes and water spreaders) will be repeated in the future and extended to larger areas, such as a series of arroyos.

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StreamStats: A U.S. Geological Survey Web Site for Stream Information

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Abstract

The U.S. Geological Survey has developed a Web application, named StreamStats, for providing streamflow statistics, such as the 100-year flood and the 7-day, 10-year low flow, to the public. Statistics can be obtained for data-collection stations and for ungaged sites. Streamflow statistics are needed for water-resources planning and management; for design of bridges, culverts, and flood-control structures; and for many other purposes. StreamStats users can point and click on data-collection stations shown on a map in their Web browser window to obtain previously determined streamflow statistics and other information for the stations. Users also can point and click on any stream shown on the map to get estimates of streamflow statistics for ungaged sites. StreamStats determines the watershed boundaries and measures physical and climatic characteristics of the watersheds for the ungaged sites by use of a Geographic Information System (GIS), and then it inserts the characteristics into previously determined regression equations to estimate the streamflow statistics. Compared to manual methods, StreamStats reduces the average time needed to estimate streamflow statistics for ungaged sites from several hours to several minutes.

Keywords: surface water, flood frequency, low flow, statistics, United States

Introduction

Estimates of the magnitude and frequency of floods (flood-frequency estimates), such as the 100-year flood (1 percent recurrence probability), are needed for floodplain mapping and for the design of bridges, culverts, and flood-control structures. Estimates of other streamflow statistics, such as the mean annual flow, the 7-day, 10-year low flow, and flow-duration statistics, are also needed for design of infrastructure and for water-resources planning and management. These estimates are commonly needed at gaged stream sites, where data are available to determine the statistics, and more often at ungaged sites, where no observed data are available.

The U.S. Geological Survey (USGS) has developed and published numerous equations to provide simple methods for estimating flood frequencies and other streamflow statistics at ungaged sites. The equations are developed from regression analyses on data for gaged sites, using streamflow statistics as the dependent variables and measured physical and (or) climatic characteristics of the drainage basins for the sites as explanatory variables. Flood-frequency equations are available for every State, Puerto Rico, American Samoa, and a number of metropolitan areas in the United States (for example, see http://me.water.usgs.gov/99-4008.pdf). Equations for estimating other streamflow statistics are also available in many areas (for example, see http://water.usgs.gov/pubs/wri/wri004135/).

The USGS has developed a prototype Web application named StreamStats that provides estimates of streamflow statistics for gaged and ungaged sites. StreamStats users can point and click on data-collection stations shown on a map in their Web browser window to obtain streamflow statistics and other information for the stations. Users also can point and click on any stream shown on the map to get estimates of streamflow statistics for ungaged sites. StreamStats uses a GIS to automatically determine the watershed boundary and physical and climatic characteristics for the ungaged site, and then it inserts the characteristics into previously determined regression equations to estimate the streamflow statistics for the site. Compared to

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manual methods, StreamStats reduces the average time needed to estimate the streamflow statistics for ungaged sites from several hours to several minutes. Estimates provided by StreamStats for ungaged sites assume rural (non-regulated) flow conditions only.

StreamStats Development

An initial StreamStats prototype that was released in 2000 for Massachusetts can be accessed on the Web at http://ststdmamrl.er.usgs.gov/streamstats/. The Massachusetts StreamStats Web site provides full documentation (Ries et al. 2000) and instructions for use of the application. The USGS and MassGIS, the Massachusetts Geographic Information Systems (GIS) Agency, jointly developed Massachusetts StreamStats in cooperation with the Massachusetts Departments of Environmental Management and Environmental Protection. The cooperating agencies wanted to use StreamStats primarily as a planning and management tool. The State of Massachusetts requires applicants for water-withdrawal permits to use StreamStats to determine the natural flow of any stream from which 378.5 cubic meters (100,000 gallons) per day or more of water would be withdrawn.

Representatives from many states have expressed interest in implementing StreamStats; however, Massachusetts StreamStats is not easily modified for use in other areas because it was built to use GIS data sets that are unique to Massachusetts and because the technology on which it was based is now obsolete. Because of this, the USGS has developed a new StreamStats prototype that will be able to provide streamflow statistics and other information for any State. The national StreamStats prototype was designed to have all of the features of Massachusetts StreamStats, but it is easier to use and has several new features. The new program takes advantage of advances in the technology for serving maps and data over the Web, and in other advances in computer technology, since Massachusetts StreamStats was developed.

The new prototype currently has been implemented only for Idaho (Figure 1). It is still being tested, and it is not yet available to the public. Plans have been made to make it available to the public and to add several more states within a year.

StreamStats has four major components, including (1) a user interface that displays maps and allows users to select stream locations where they want streamflow statistics; (2) a database of previously published streamflow statistics and descriptive information for 725 USGS data-collection stations; (3) a GIS database that contains all the map layers needed for measuring the watershed and climatic characteristics in addition to numerous map layers that can be used to identify locations of interest; and (4) an automated GIS procedure that measures the watershed and climatic characteristics needed to solve the regression equations and inserts those characteristics into equations to estimate the streamflow statistics. Each of these components is described below.

User interface

The StreamStats user interface and the automated GIS procedure were developed using ArcIMS (http://www.esri.com/software/arcims/index.html) and ArcGIS 8.1 (http://www.esri.com/software/arcgis/ index.html). The new user interface appears within a Web browser window rather than as a separate window, as in Massachusetts StreamStats.

The map frame for displaying default and user-selected map layers is the largest feature in the user interface. For the Idaho prototype, the map frame initially shows a shaded-relief map of Idaho and the surrounding area, with locations of data-collection stations. When users zoom in to locate sites of interest, different scales of digital topographic maps are shown depending on the selected scale. When more states are added to StreamStats, a map of the United States and State boundaries will be shown initially. Clicking on a state will cause a map of the state to appear in the map frame.

Above the map frame is a set of buttons to activate tools that allow users to (1) zoom in, zoom out, and pan to areas of the map; (2) get information about features displayed on the map; (3) zoom to the full or last extent shown in the map; (4) delineate a drainage basin for an ungaged site; (5) get streamflow statistics for the delineated basin; (6) download GIS data displayed in the map; and (7) print the map.

The map legend frame is at the right of the map frame, and shows the names and the map symbols for the map layers that can be displayed in the map frame. Check boxes next to the layer names in the legend frame allow users to turn on and off the display of the layers. A locator map can be shown instead of the map layers by clicking on the Locator Map tab at the top of the frame.

Above the map layers frame is the Zoom To tool, which allows users to zoom to an address, a water feature fromthe National Hydrography Dataset (NHD) (http://nhd.usgs.gov/), or a geographic name from the Geographic Names Information System (GNIS) of the USGS (http://geonames.usgs.gov/). A scale bar above the Zoom To tool allows users to zoom to a specific scale.

Streamflow statistics database

The streamflow statistics database, named StreamstatsDB, was developed using Microsoft Access. The database has a custom user interface that allows for easy entry and update of data. The initial release of StreamstatsDB contains 144 types of physical and climatic basin characteristics and 492 types of streamflow statistics. Each streamflow statistic has an associated standard error of estimate in the database. All USGS-developed regression equations for estimating peak-flow frequency statistics were reviewed to determine the basin characteristics needed initially in StreamstatsDB.

StreamstatsDB was released internally to each of the 48 District offices within the Water Discipline of the USGS so local staff can populate the database with whatever previously published data are available. District staff can easily add any new basin characteristics or streamflow statistics that they need in the database.

StreamStatsDB has not yet been linked to the StreamStats prototype we application. When the linking is completed, users will be able to click on data-collection sites shown on the map and get information for the sites from the database.

GIS database

The GIS database contains map layers needed to assist users in locating sites of interest and map layers needed to measure basin and climatic characteristics used as explanatory variables in the regression equations used to estimate the streamflow statistics. StreamStats will use nationally available GIS data layers as much as possible for measuring basin characteristics and for use as base map layers. In many areas of the United States, however, local data layers will be used instead of or in addition to the national data layers because they have better resolution than the national data layers or they provide information that is not available from a national data layer. Some of the national data layers used in StreamStats include:

- Digital Raster Graphics (DRG) (http://mcmcweb.er.usgs.gov/drg/) scanned USGS topographic maps are used as the primary base layer for precise site selection
- National Elevation Dataset (NED) (http://edcnts12.cr.usgs.gov/ned/),
- Elevation Derivatives for National Applications (EDNA) dataset (http://edcnts12.cr.usgs.gov/ned-h/),
- National Hydrography Dataset (NHD) (http://nhd.usgs.gov/),
- Watershed Boundary Dataset (WBD) (http://www.ftw.nrcs.usda.gov/huc_data.html),
- National Land Cover Dataset (NLCD) (http://landcover.usgs.gov/),
- Parameter-elevation Regressions on Independent Slopes Model (PRISM) climate data (http://www.ocs.orst.edu/prism/), and
- State Soil Geographic Database (STATSGO) soil survey data (http://www.ftw.nrcs.usda.gov/stat_data.html)

StreamStats requires three data layers to determine watershed boundaries for ungaged sites: (1) a Digital Elevation Model (DEM) (Elassel and Caruso 1983), (2) a networked data layer of streams with lines connecting through the centers of wetlands and water bodies, and (3) a watershed boundary data layer. In much of the United States, StreamStats will use the EDNA data layers to determine watershed boundaries and to measure several watershed characteristics. In some areas, however, StreamStats will use other data layers, such as the NHD, the WBD, the NED, or locally developed data to determine watershed boundaries and characteristics. The ENDA data are used in the Idaho prototype.

The EDNA data layers were developed through an interagency effort, and are derived from the NED, a DEM with grid cells of 30 meters on a side and a vertical accuracy of about 1.5 meters. The EDNA includes a hydrologically correct DEM, a networked synthetic streams layer, a catchment layer that includes catchments derived for each reach in the synthetic stream network, and several other layers. The DEM was made hydrologically correct by filling spurious depressions so that surface runoff could not collect in the depressions on the DEM land surface. The synthetic streams were obtained by setting a minimum flow-accumulation value of 5,000 DEM grid cells (about 4.5 square kilometers) to generate a stream. The catchments define the drainage boundary that is unique to each synthetic reach.

All local and national GIS data used in StreamStats will be formatted to conform to the Hydrology Data Model developed by the GIS Water Resources Consortium (http://www.crwr.utexas.edu/giswr/). The data will be stored in geodatabases (http://www.esri.com/news/arcuser/0701/migrating.html) organized by major watersheds. Watershed characteristics needed to solve the regression equations in StreamStats will be pre-calculated for each catchment and attached as attributes to the catchment data layer to increase the efficiency of the application.

Automated GIS procedure

The StreamStats automated procedure was developed as a Dynamic Link Library (DLL), a bundle of Visual Basic subroutines that operates within ArcGIS 8.1. The DLL transfers information to and from ArcIMS to allow implementation over the Web.

When users select an ungaged site along the stream network, the coordinates for the site are sent to a computer on which the StreamStats automated GIS procedure is running. The procedure then determines the watershed boundary for the location from the DEM up to the points at which the DEM-defined boundary coincides with existing catchment boundaries. StreamStats then combines the new boundaries with the existing boundaries for any upstream catchments and dissolves the internal boundaries to obtain the complete drainage-basin boundary for the location. Use of the existing catchment boundaries minimizes errors associated with determining boundaries for ungaged sites from the raw DEM and minimizes processing time.

StreamStats displays the watershed boundary for the user-selected site in the map frame, with the stream network and digital images of USGS topographic maps shown as the base data layers. Users should verify that the boundary determined by StreamStats is correct before proceeding with estimating streamflow statistics for the site.

After the user verifies the drainage-basin boundary, StreamStats determines the applicable watershed and climatic characteristics and solves the regression equations to obtain estimates of streamflow statistics for the ungaged site. The estimates are computed by the National Flood Frequency (NFF) computer program (Ries and Crouse 2002), which is linked as a subroutine to StreamStats. Information on NFF can be obtained and the program can be downloaded at http://water.usgs.gov/software/nff.html.

Output from StreamStats is provided in a pop-up Web browser window. The output consists of the date, the coordinates of the site, the measured watershed and climatic characteristics, the estimated streamflow statistics, and either standard errors or 90-percent prediction intervals as indicators of the reliability of the estimates. Approximately two-thirds of estimates for ungaged sites will have errors within the standard errors provided. There is a 90-percent probability that the actual streamflow statistics for a site are within the 90-percent prediction interval. The estimates assume natural flow conditions at the ungaged site.

Figure 1. View of national StreamStats prototype user interface showing a delineated drainage basin for a selected site imposed over a digital topographic map.

StreamStats Implementation

Work by the USGS District offices will be needed to implement StreamStats. Usually, cooperative agreements for cost sharing with state and local agencies will be needed to complete the work.

Steps involved in implementing StreamStats include at least (1) preparation of GIS data needed for the application, (2) population of StreamstatsDB with streamflow statistics and other information for data-collection stations, and (3) verification that the watershed and climatic characteristics are measured and the streamflow statistics are estimated without bias and with the same accuracy as that indicated in the USGS reports that describe the regression equations for each State. Because regression equations for many states were developed using basin characteristics that were not measured using a GIS, it is likely that new equations will need to be developed before the ungaged site process can be implemented for these states.

After StreamStats is made available to the public for Idaho, work will begin on implementing it for other states. Massachusetts, New Hampshire, and Vermont are the next states scheduled for implementation. Though several states have started the implementation process, full national implementation is expected to take several years to complete. Additional information on the development approach and implementation plans for StreamStats can be obtained at http://water.usgs.gov/osw/programs/streamstats.html

Note

The use of trade, firm, or product names in this paper is for identification purposes only and does not imply endorsement by the U.S. Geological Survey.

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Watershed Modeling II

Assessment of Two Physically-Based Watershed Models Based on Their Performances of Simulating Water and Sediment Movement

Latif Kalin, Mohamed M. Hantush

Abstract

Two physically based watershed models, GSSHA and KINEROS-2, are evaluated and compared for their performances on modeling flow and sediment movement. Each model has a different watershed conceptualization. GSSHA divides the watershed into cells, and flow and sediments are routed through these cells in a cascading fashion. Conversely, KINEROS-2 divides the watershed into sub-watersheds and channel segments having uniform properties. GSSHA requires much longer simulation times depending on what is simulated. KINEROS-2, on the other hand, entails relatively less data and effort. Simulations were performed with each model over a small watershed for several events. Models were calibrated using the same events and the differences in estimated parameters were discussed. Both models have resulted in different calibration parameters although the underlying physics are similar. The differences in model behaviors are discussed.

Keywords: sediment, distributed models, watershed, GSSHA, KINEROS-2

Introduction

Hydrologic models are useful tools in understanding the natural processes in a watershed. For instance, one practical application is analyzing the effect of land use changes, such as urbanization, on runoff and sediment yield. There are numerous watershed scale hydrologic models varying from lumped such as the unit hydrograph concept (Sherman 1932) to highly complex distributed models such as MIKE-SHE (Refsgaard and Storm 1995). Each of those models has their own advantages and disadvantages. Depending on needs, sometime a simple lumped model might suffice. However, to achieve TMDL targets and implement BMPs, use of distributed models is inevitable. The availability of high power computers relaxed the burden of long simulation times. Among the distributed models the physically based ones always have edges over the empirical ones, since the model parameters have physical meanings and can be measured in the field. When measurements are not available model parameters can be still be deduced from published data in literature based on topography, soil and land use maps. When flow is concerned, to our knowledge three models seem to be the most physically based and separate themselves from others: GSSHA (Downer and Ogden 2002), KINEROS-2 (Smith et al. 1995) and MIKE-SHE (Refsgaard and Storm 1995). In this study we examined and compared the former two. In what follows is a brief discussion of each model.

GSSHA

Gridded Surface Subsurface Hydrologic Analysis (GSSHA) is a reformulation and enhancement of CASC2D (Downer and Ogden 2002). The CASC2D model was initiated at Colorado State University by Pierre Julien as a two dimensional overland flow routing model. In its final form, it is a distributed-parameter, physically-based watershed model. Both single event and continuous simulations are possible. The U.S. Army Waterways Experiment Station considered this model as very promising and therefore fully incorporated this model into

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Watershed Modeling System(WMS). Watershed is divided into cells and water and sediment is routed from one cell to another. It uses one and two-dimensional diffusive wave flow routing at channels and overland planes, respectively. Although only Hortonian flows were modeled by employing Green-Ampt (G-A) infiltration model in the initial versions, GSSHA considers other runoff generating mechanisms such as lateral saturated groundwater flow, exfiltration, stream/groundwater interaction etc. GSSHA offers two options for long-term simulations: G-A with redistribution (Ogden and Saghafian 1997) and the full Richards' equation. The latter requires tremendous amount of simulation time and is very sensitive to time step and horizontal and vertical cell sizes (Downer and Ogden 2003). Modified Kilinc and Richardson equation (Julien 1995) is used to compute sediment transport capacity at plane cells. A trap efficiency measure is used to determine how much material is transported from the outgoing cell. Details on theory and equations used can be found in Julien et al. 1995, Johnson et al. 2000, and Downer and Ogden 2002.

KINEROS-2

This is the improved version of KINEROS (Woolhiser et al. 1990). It is event based since it lacks a true soil moisture redistribution formulation for long rainfall hiatus and more importantly it does not consider evapotranspiration (ET) losses. This model is primarily useful for predicting surface runoff and erosion over small agricultural and urban watersheds. Smith et al. 1995 suggest watershed size smaller than 1,000 ha for best results. Runoff is calculated based on the Hortonian approach using a modified version of Smith- Parlange (Smith and Parlange 1978) infiltration model. KINEROS-2 requires the watershed divided into homogeneous overland flow planes and channel segments, and routs water movement over these elements in a cascading fashion. Mass balance and the kinematic wave approximations to the Saint Venant equations are solved with implicit finite difference numerical scheme in a 1-D framework. KINEROS-2 accounts for erosion resulting from raindrop energy and by flowing water separately. A mass balance equation is solved to describe sediment dynamics at any point along a surface flow path. Erosion is based on maximum transport capacity determined by Engelund-Hansen equation (1967). The rate of sediment transfer between soil and water is defined with a first order uptake rate. A detailed description of the model and the equations used can be found in Smith et al. 1995 and at the official URL of the model: http://www.tucson.ars.ag.gov/kineros.

Data

The data used in this study comes from a small USDA experimental watershed named W-2, which is located near Treynor, Iowa. It is approximately 83 acres. Figure 1 depicts the location and topography of this watershed. This watershed is one of the 4 experimental watersheds established by USDA in 1964 to determine the effect of various soil conservation practices on runoff and water-induced erosion. Runoff and sediment load has been measured since then. There are two rain gauges (115 and 116) around the watershed. W-2 has a rolling topography defined by gently sloping ridges, steep side slopes, and alluvial valleys with incised channels that normally end at an active gully head, typical of the deep loess soil in MLRA 107 (Kramer et al. 1990). Slopes usually change from 2 to 4 percent on the ridges and valleys and 12 to 16 percent on the side slopes. An average slope of about 8.4 percent is estimated, using first-order soil survey maps. The major soil types are well drained Typic Hapludolls, Typic Udorthents, and Cumulic Hapludolls (Marshall-Monona-Ida and Napier series), classified as fine-silty, mixed, mesics. The surface soils consist of silt loam (SL) and silty clay loam (SCL) textures that are very prone to erosion, requiring suitable conservation practices to prevent soil loss (Chung et al. 1999). Corn has been grown continuously on W-2 since 1964.

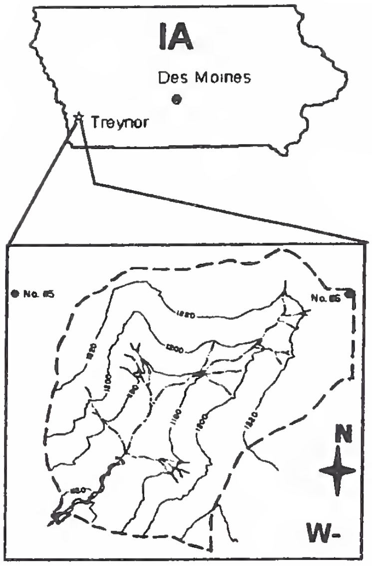

Figure 1. Study watershed.

Methodology

KINEROS-2 was already calibrated for W-2 watershed in a previous study using 3 rainfall events (Kalin and Hantush 2003). In that study average values were used for net capillary drive, G (35,20 cm), pore size distribution index, λ (0.6,0.6), porosity, ϕ (0.47,0.50), and median particle size diameter, D_{50} (7 μm). The two values given in parentheses represent SCL and SL soil types, respectively. Table 1 lists the parameter sets used after calibration of KINEROS-2. In the table, n is Manning's roughness, K_s is saturated hydraulic conductivity, I is interception depth, S_i is initial saturation, C_g is soil cohesion coefficient and C_f is rainsplash coefficient. For simplicity channel and overland roughness were assumed to be same. Since corn has been grown on W-2, the parameters n, C_g and C_f were allowed to vary with season where C_g and C_f were assumed to decay exponentially with the growing season. This assumption was justified over four independent verification events (see Kalin and Hantush 2003).

Table 1. Parameter sets used in KINEROS-2.

event	n	K_s*	I**	S_i	C_g	C_f
6/13/83	0.055	(1.8,6.5)	2.0	(0.44,0.27)	0.15	160
5/30/82	0.040	(1.5,6.0)	0.0	(0.90,0.86)	0.25	200
8/26/81	0.080	(2.0,7.0)	1.0	(0.84,0.60)	0.05	100

* K_s: mm/hr
** I: mm

Flow simulations

GSSHA was run with the above events. KINEROS-2 values were directly substituted for parameters common to both models i.e. λ, ϕ, n, I, S_i, and K_s. Other parameters were adjusted accordingly. The infiltration scheme in GSSHA is the Green-Ampt (G-A) model, whereas KINEROS-2 uses Smith-Parlange infiltration model. G-A capillary head (Ψ) needs to be provided in GSSHA. We approximated Ψ as equal to G in KINEROS-2. Figure 2 shows the comparison of the simulation results for flow with two models. It is clear that both models behave very differently when similar parameter sets are used as inputs. The most striking observation is that, in all cases GSSHA generates later responses and lower peak flows than KINEROS-2. For instance, the difference in time to peaks for the event 8/26/81 is around 25 minutes which is very significant considering the fact that the base time is around 150 minutes. Similarly, the peak flow generated by KINEROS-2 is about 45 % larger than the peak flow generated by GSHHA. One possible rationale to this might be the different watershed conceptualizations involved in each model. Flow routing in GSSHA is only in x-y directions. In other words, flow from a cell is allowed only in the four principal directions. Diagonal neighboring cells can not be receivers which well might be the reality. This results in overestimation of the travel lengths of water particles which might be up to 41 %. On the other hand, the travel paths used to compute the average travel lengths of each element in KINEROS-2 were determined based on the D-8 methodology using the TOPAZ algorithm (Garbrecht and Martz 1999) which allows flow in 8 directions. Considering the fact that flow in the study watershed is mostly diagonal, the overestimation of travel lengths by GSSHA resulted in longer travel time leading to more resistance to flow, and consequently lower and retarded peaks.

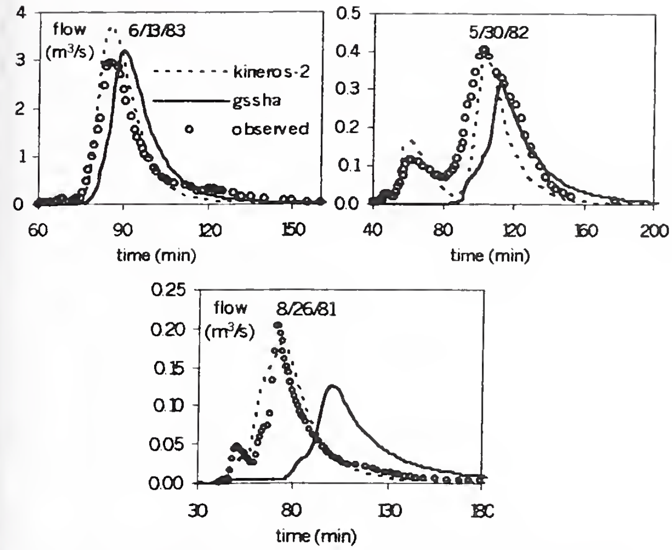

Figure 2. Comparison of hydrographs generated with GSSHA and KINEROS-2 based on KINEROS-2 calibrated parameters.

The differences in flow volumes do not seem to be significant. With this set of parameters KINEROS-2 seems to simulate events having multi-modal shapes, such as the one in 5/30/82, better than GSSHA. In fact GSSHA completely misses the first and second humps in 5/30/82 as opposed to KINEROS-2. KINEROS-2, to some extent, performs better than GSSHA in simulating the small hump seen on the observed data of 8/26/81.

It is important to keep in mind that all these observations are based on simulations with the parameters calibrated for KINEROS-2. Therefore, we recalibrated the GSSHA parameters for the same events. This time each event was calibrated individually and parameters were compared to KINEROS-2 calibrated parameters. We accept that we did not follow the traditional model calibration/ verification methodology. However, we need to mention that the aim of this study is basically a comparison of the two models rather than a model calibration effort. Keeping this in mind, we kept I, S_i and the overland plane roughness (n_p) same and recalibrated channel roughness (n_c) and K_s. Figure 3 shows the hydrographs after calibration. For the event 6/13/83 both model performs equally. For 5/30/82 GSSHA is still underestimating the first and second humps, but interestingly it does a better job than KINEROS-2 in representing the shape. Although KINEROS-2 could not simulate the first (the smallest hump in the figure) happening approximately at 45 minutes, GSSHA does a fairly good job in catching the both humps. Finally, when we look at the last event we see that GSSHA almost perfectly reproduces the observed hydrograph shape while KINEROS-2 suffers to simulate the first peak.

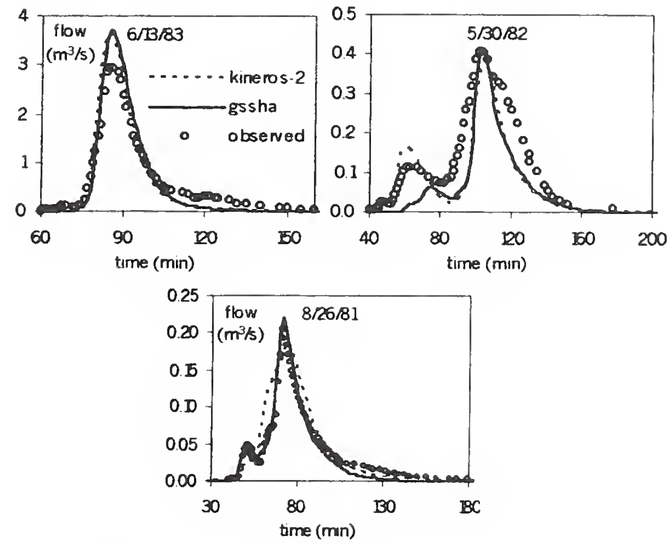

Figure 3. Comparison of hydrographs generated with GSSHA and KINEROS-2. GSSHA is recalibrated.

The recalibrated parameters for GSSHA are summarized in Table 2. In the table C is the USLE crop factor which will be discussed later. The value of n_c had to be decreased dramatically for each event that is clearly expected from Figure 2 as GSSHA generated later responses in each case. One remarkable observation is that n_c values are very close to each other which confirms the comments of Larry Kramer (personal communication) who has extensive experience on Treynor watersheds. He stated that channels are covered with bromegrass and they are cultivated such a way that channel roughness can be assumed invariable year around. K_S values are very close to KINEROS-2 values.

Table 2. Calibrated parameters with GSSHA.

event	n_c	K_s (mm/hr)	C
6/13/83	0.025	(2.0, 7.7)	0.042
5/30/82	0.020	(1.5, 6.0)	0.150
8/26/81	0.025	(1.8, 6.5)	0.050

Erosion simulations

GSSHA requires silt and sand percentages for sediment computations. Assuming that D_{50} is 0.25 mm for sand, 0.016 mm for silt and 0.003 for clay, compositions of each soil class were determined as sand % (10,25) and silt % (56,61) so that the overall average D_{50} is 7 mm, which is the value used in KINEROS-2. The sediment routine in GSSHA is based on the USLE concept that requires three parameters K, C and P. It is not practical to infer estimates of these parameters from the KINEROS-2 soil parameters; i.e., C_g and C_f. Therefore, by keeping KP product constant C was calibrated for each event, since it is only the product of K, C, and P that matters. The values of K and P are (0.37,0.48) and (0.01,0.01), correspondingly. The estimated C values are listed in Table 2. The pattern observed in KINEROS-2 that is erodibility decreases with the growing season, is not observed between the C values here. The C values obtained for the event 8/26/1981 is unexpectedly high, even higher than the value of 6/13/83. Figure 4 compares the sedimentographs obtained by KINEROS-2 and GSSHA. The general observation is that GSSHA generates narrower sedimentographs than KINEROS-2 generates. We do not have a clear reasoning for this. Further, this can not be attributed to flow, since such a behavior is not monitored in Figure 3.

It is interesting to note that the erosion parameters, c_f and c_g, found after calibration for KINEROS-2 are well above the recommended values given in Woolhiser et al. (1990) and the calibrated C parameters for GSSHA are well below the literature values. What this means is that when literature values are used, GSSHA overestimates erosion compared to KINEROS-2. Slope is an important factor in both models' erosion formulation. The smaller the computational element, which is the grid size for GSSHA and the average length of overland flow planes in KINEROS-2, the greater the erosion. Because, as the element size increases the tendency of smoothing the topography increases, and this results in loss of areas with steep slopes meaning reduction in erosion. KINEROS-2 uses far less elements than GSSHA, thus leading to loss of local slope information in the former. This probably elucidates the difference in estimates of soil erosion. A very good discussion on this topic can be found in Rosalia 2002.

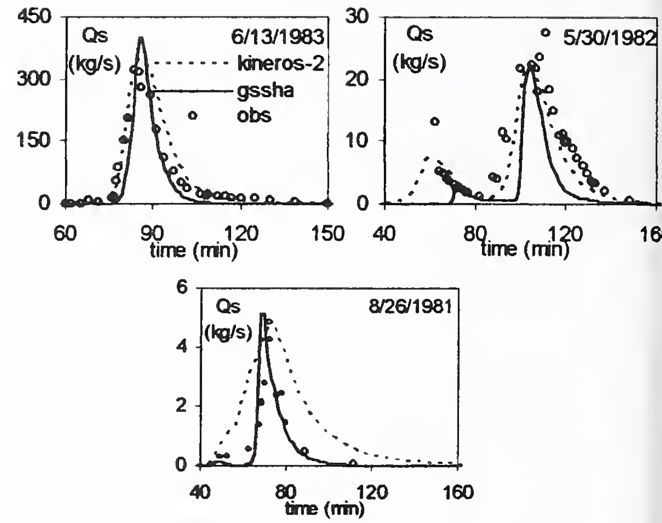

Figure 4. Comparison of sedimentographs generated with GSSHA and KINEROS-2.

Discussions and Conclusions

It is known that in numerical solutions involving finite difference schemes, as the grid size decreases the required time interval should also decrease. In fact, this is reflected in the Courant Condition as a stability criteria which can be stated as $U<\Delta x/\Delta t$ where U is velocity, and Δt and Δx are time and space increments, respectively (Chapra 1997). The grid size used for W-2 in GSSHA simulations was 10 m. This is an unusually small grid size for such simulations. In fact, 5 m horizontal resolution DEM data is also available for this area, but because of the interaction between Δt and Δx we decided to use 10 m. Using coarser grid size than 10 m would lead to inaccurate representation of the watershed since it is only 83 acres. In a review of several watershed scale hydrologic and non-point source pollution models, Borah (2002) refers to a study on CASC2D, the older version of GSSHA, where Molnar and Julien (2000) found that for a 150 m grid size the required time step was about 5 seconds. This number decreased to 1 second when the grid size was reduced to 30 m. The smallest time interval allowed by GSSHA is 1 second which is the value used in our simulations. This might have introduced additional uncertainty.

One of the deficiencies of the GSSHA is that erosion in channels is not transport limited. GSSHA can generate sediment which has a volume larger than flow. This is physically impossible; however there is nothing in the GSSHA formulation to prevent this from happening once sediment reaches the channels (Downer, personal communication). When we initially used the literature values for C, K, and P parameters we observed this effect. Eventually we had to decrease these parameters dramatically to get more realistic results. This itself is enough to claim that the sediment routine in KINEROS-2 is more robust than the routine used in GSSHA. In fact, there is a contract between US Army Corps of Engineers and Fred Ogden, University of Connecticut, one of model developers, to completely reformulate the sediment routine of GSSHA (Downer and Ogden, personal communication). It would be interesting to redo this whole exercise once that project is completed.

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Past and Prospective Effects of N Deposition on Carbon Cycling and Nutrient Retention in the Delaware River Basin

Yude Pan, John Hom, Kevin McCullough

Abstract

This research will analyze the effects of nitrogen deposition, cation depletion, climate change, and land use change in the Delaware River Basin (DRB) using a spatially explicit process model, PnET-CN, with high specificity for forested watersheds in the Northeast and mid-Atlantic regions. The model predicts the net primary productivity (NPP), carbon storage, water yield, nitrogen export, nitrogen retention in forested watersheds. We are using the model to investigate how changes in environmental variables, separately or interactively, affect carbon and nutrient processes, and the function and structure of forest ecosystems using scenarios of N deposition, elevated CO2, climate change and forest management within the Delaware River Basin. Preliminary results suggest that the chronic N increase in the past 70 years has increased forest productivity by 22%, forest biomass by 11% and soil organic matter by 22%. Increased N deposition is the major cause for enhanced C sequestration in the forested watershed of the DRB, with elevated CO2 secondary in importance for enhancing growth (12%). The historic increase of atmospheric CO2 resulted in the largest C gains in forest living biomass (18%). The combination of rising N deposition and CO2 has increased C in forest biomass by 38% and in soil organic matter by 28%. The current N leaching losses from the DRB forested watersheds is estimated at 1.7 Kg/ha/yr with N retention rate at 83%. The impact of ozone and calcium depletion due to acidic deposition will be added as enhancements to this modeling study for sensitive areas in the Basin.

Keywords: nitrogen deposition, forest productivity, nitrogen retention, water yield

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Entropy-Based Assessment of Two Hydrologic Models

Mariano Hernandez, David C. Goodrich, Leonard J. Lane

Abstract

The purpose of this paper is to evaluate the effects of watershed segmentation and rain gage network density on simulated watershed response using informational entropy concepts. Two hydrologic models are used to simulate runoff from a small watershed in the USDA-ARS Walnut Gulch Experimental Watershed while varying the number of rain gages in the network and the contributing source area to generate different drainage network configurations. The first model is an event-based, physically distributed parameter model and the second is a continuous simulation model based on the hydrologic Curve Number. Both models are calibrated using the maximum number of rain gages in the network and the highest resolution configuration of the drainage network. The efficiency of both models is evaluated at each optimal rain gage network density and drainage network configuration. In this research, it is argued that there is a relationship between the maximum efficiency of the hydrologic models and the amount of information conveyed by both the existing rain gage network and the topological representation of the watershed. Based on the optimization of the informational content, a maximum contributing source area is identified as a function of the optimal spatial distribution of rain gages that maximizes model performance. This result can represent a new approach in evaluating the informational content associated with hydrologic data and the hydrologic model performance in order to develop a watershed segmentation criterion that defines geometric model complexity.

Keywords: informational entropy, hydrologic uncertainty, watershed, complexity

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Modeling Uncertainty of Runoff and Sediment Yield Using a Distributed Hydrologic Model

Mohamed M. Hantush, Latif Kalin

Abstract

The event- and physically-based runoff and erosion model KINEROS2 is applied to assess the impact of uncertainty in model parameters on simulated hydrographs and sediment discharge in a small experimental watershed. The net capillary drive parameter, which affects soil infiltration, is shown to be approximately lognormally distributed, and related statistics of this parameter for all soil texture class are computed and tabulated. The model output response to uncertainty in soil hydraulic and channel roughness parameters is evaluated by performing Monte Carlo (MC) simulations based on the parameters' statistics obtained from the literature. The results show the extent to which uncertainty in the saturated hydraulic conductivity, net capillary drive, and initial relative saturation influences the simulated cumulative distributions of peak sediment discharge and sediment yield. The distribution of the simulated time to peak sediment discharge is dominated by uncertainty in the channel and plane Manning's roughness coefficients. Comparison of the simulated median and uncertainty (± 25% and 45% quartiles) with observed values of runoff and sediment discharge, for two, large and small rainfall events, indicate that the model performs properly and can be calibrated based on the range of soil parameters reported in the literature.

Keywords: sediment, hydrologic model, KINEROS2, Monte Carlo simulations, uncertainty, net capillary drive

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Introduction

Sediments are cited as the third leading cause of stress for lakes, reservoirs, and ponds, behind nutrients and metals (U.S. EPA 2000). Agriculture land use activities are the leading source for sediment stress. Sediment runoff carries with it adsorbed toxic chemicals and nutrients that have the potential to cause major environmental problems to aquatic ecological systems and water quality impairment in streams and lakes (e.g., eutrophication and hypoxia). Distributed hydrologic runoff/sediment models are increasingly relied upon by scientists and resource managers as cost-effective tools for linking hillslope soil erosion and erodable surfaces to receiving waterbodies, thereby, allowing for direct assessment of the impact of land use practices on water quality in streams and lakes.

The Kinematic Runoff and Erosion model KINEROS2, which is based on first principals (i.e., physics based), is suitable for evaluating the effect of events on runoff and erosion in small watersheds (Smith et al. 1995). In spite of its limitations, successful applications of this model to gaged watersheds has been reported in the literature (Osborn and Simanton 1990, Goodrich et al. 1994, Smith et al. 1999, Ziegler et al. 2001, Kalin et al. 2003, Kalin and Hantush 2003). This paper presents an application to the event-based model KINEROS2 to simulate runoff and sediment discharge in a USDA experimental watershed. The objectives are: 1) to identify model parameters that contribute mostly to model output uncertainty by performing Monte Carlo simulations; and 2) estimating uncertainties in model predictions.

Background Theory

KINEROS2 is a distributed, event-oriented, physically based model describing the processes of surface runoff and erosion from small agricultural and urban watersheds (Woolhiser et al. 1990). The watershed is represented by cascade of planes and channels, in which flow and sediments are routed from one plane to the other and, ultimately, to the channels. The cascades allow rainfall, infiltration, runoff, and erosion parameters to vary spatially. This model may be used to determine the effects of various artificial features such as urban developments, small detention reservoirs, or lined channels on flood hydrographs and sediment yield.

When rainfall rate approaches the infiltration capacity, Hortonian overland flow begins. KINEROS2 assumes one-dimensional flow in each plane and solves the kinematic wave approximation of the overland and channel flow equations using finite differences. The flow rate is related to the channel flow cross-sectional area or overland flow depth through Chezy and Manning flow resistance relationships. In these relationships the channel or bed slope approximates the friction slope.

Sediment transport equation is described by the following mass balance equation:

$$\frac{\partial}{\partial t}(AC)+\frac{\partial}{\partial x}(QC)-e(x,t)=q_s(x,t) \qquad (1)$$

in which C is the volumetric sediment concentration $[L^3/L^3]$; A is the channel cross section area $[L^2]$; for overland flow it is equal to the flow depth h for a unit flow width [L]; Q is the is the channel discharge $[L^3/T]$; for overland flow it is equal to the discharge per unit width $[L^2/T]$; e is sediment erosion rate $[L^2/T]$ given below; and q_s is the rate of lateral sediment inflow for channels $[L^3/T/L]$. In KINEROS2 Sediment erosion/deposition rate e is composed of rainfall splash erosion rate g_s and hydraulic erosion rate g_h:

$$e=g_s+g_h \qquad (2)$$

Rainfall splash erosion is given by (Woolhiser et al., 1990):

$$\begin{aligned} g_s &= c_f\, e^{-c_h h}\, r\, q; && q>0 \\ &= 0; && q<0 \end{aligned} \qquad (3)$$

in which c_f is a positive constant [T]; h is flow depth [L]; c_h is damping coefficient for splash erosion $[L^{-1}]$; r is rainfall rate [L/T]; q is excess rainfall (rainfall rate minus interception minus infiltration) [L/T]. The exponential term represents the reduction in splash erosion caused by increasing depth of water (Smith et al. 1995). In channel flow, this term is usually equal to zero: the accumulating water depth absorbs nearly all the imparted energy by the raindrops. The hydraulic erosion represents the rate of exchange of sediment between the flowing water and the soil over which it flows. Such interplay between shear force of water on the loose soil or channel bed and the tendency of the soil particles to settle under the force of gravity may be described by this first-order rate expression:

$$g_h=c_g(C^*-C)A \qquad (4)$$

in which C^* is the volumetric concentration at equilibrium transport capacity $[L^3/L^3]$; c_g is a transfer rate coefficient $[T^{-1}]$. For sheet flow A = h. This relationship assumes that if C exceeds equilibrium saturation, C^*, deposition occurs. c_g is usually very high for fine, noncohesive material, and very low for cohesive material. Several expressions for C^* are available in the literature (e.g. Woolhiser et al. 1990). In our analysis, we used the formula by Engelund and Hansen (1967).

Net Capillary Drive Parameter

At the beginning of a storm and prior to ponding, the infiltration rate is rain limited and equal to the rate of precipitation. At the onset of ponding, the infiltration rate is equal to the infiltration capacity, provided that it is greater than the saturated hydraulic conductivity of the soil, and is given by (Parlange et al. 1982):

$$f(t)=K_s\left[1+\frac{\alpha}{e^{\alpha F(t)/[(G+h)(\theta_s-\theta_i)]}-1}\right] \qquad (5)$$

in which f(t) is the infiltration capacity [L/T]; F(t) is the cumulative depth of the water infiltrated into the soil [L]; θ_s is the soil porosity $[L^3/L^3]$; θ_i is the initial soil moisture content prior to the storm; α is a parameter generally between 0 and 1; and K_s is the soil saturated hydraulic conductivity [L/T]. $\alpha = 0$ corresponds to the familiar Green Ampt infiltration method. For most soils, $\alpha = 0.85$ has been recommended (Parlange et al. 1982). G is the net capillary drive parameter:

$$G = \int_0^{\psi_i} [K(\psi)/K_s] d\psi \quad (6)$$

ψ here is defined as the negative of the capillary pressure [L]. If we substitute the Brooks and Corey soil characteristic relation for unsaturated conductivity:

$$K(\psi) = K_s \, (\psi_b / \psi)^{2+3\lambda} \quad (7)$$

and integrate from 0 to ψ_b, with $K(\psi) = K_s$ and from ψ_b to ψ_i, with $K(\psi)$ given by (7)we obtain:

$$G = \psi_b \left\{ 1 + \frac{1 - (\psi_b / \psi_i)^{1+3\lambda}}{1 + 3\,\lambda} \right\} \quad (8)$$

in which ψ_b is the bubbling pressure [L]; ψ_i is the negative of the soil initial capillary head [L]; and λ is the pore-size distribution index. The specific case of $\psi_I = \infty$ produces the commonly used expression for the net capillary drive function $G = \psi_b [(2+3\lambda)/(1+3\lambda)]$.

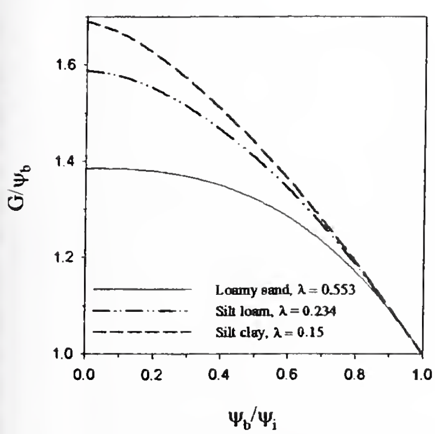

Figure 1. Scaled net capillary drive vs. scaled initial capillary pressure for three different soil textures.

This expression is used in KINEROS2, which may be valid after prolonged rainfall hiatus. For relatively wet soil conditions or short hiatus periods, the capillary pressure may attain a finite, but not a large absolute value. The effect of this on runoff and sediment yield hitherto is unknown. Figure 1 shows variation of the scaled net capillary drive G/ψ_b with scaled initial capillary pressure head ψ_b/ψ_i for soil textures loamy sand ($\lambda = 0.553$), silt loam ($\lambda = 0.234$), and silt clay ($\lambda = 0.15$). Values for λ shown in the figure are the arithmetic means (Rawls et al. 1982). Note that for sufficiently wet silty clay, arithmetic mean $\psi_b = 76.54$ cm (Rawls et al. 1982), G would be quite different, say, at $\psi_b/\psi_i = 0.7$ than at 0. For example, at $\psi_i = \infty$, $G = 1.69 \times 76.54 \approx 129$ cm; at $\psi_i = 109$ cm, $G = 1.28 \times 76.54 \approx 98$ cm. The use of G value based on $\psi_i = \infty$ may lead to over predicting infiltration and under predicting runoff, consequently, sediment yield.

To achieve the above two objectives, we begin by developing the probability density function for the G parameter, using the commonly used expression $G = \psi_b[(2+3\lambda)/(1+3\lambda)]$, while bearing in mind that in general G may also depend on the initial capillary pressure ψ_i through Equation (8) (refer to Figure 1). Taking the natural logarithm of both sides of the above expression (G at $\psi_i = \infty$) leads to:

$$\ln G = \ln \psi_b + \ln\left(\frac{2+3\lambda}{1+3\lambda}\right) \quad (9)$$

Rawls et al. (1982) indicated that ψ_b and λ are log-normally distributed; they provided the arithmetic and geometric mean values with the corresponding standard deviations for both parameters, for different texture class. Over the reported range of values for λ, we have this approximation:

$\ln G \sim N(\mu_{\ln G}, \sigma^2_{\ln \psi_b})$,

$$\mu_{\ln G} \approx \mu_{\ln \psi_b} + \ln\left[(2+3\bar{\lambda})/(1+3\bar{\lambda})\right] \quad (10)$$

That is; G is **lognormally** distributed, with the mean of lnG (i.e., geometric mean) given by (10) and variance of $\ln G \approx \sigma^2_{\ln \psi_b}$, which is the variance of $\ln\psi_b$. $\bar{\lambda}$ is the geometric mean of λ. Rawls et al. (1982) provide values of $\sigma^2_{\ln \psi_b}$ and $\bar{\lambda}$ for different soil textures. Table 1 provides the arithmetic mean and standard deviations of G for different soil textures obtained from the lognormal approximation and by performing 10000 Monte Carlo simulations, using the statistics of the lognormally distributed ψ_b and λ (Rawls et al. 1982).

Figure 2 plots the theoretical arithmetic mean (analytical) and standard deviation versus those obtained by MC simulations. The comparison shows that the **lognormal** approximation of G is valid over different soil textures. We note that the results in

Table 1 are based on $\psi_i = \infty$, which in light of Equation 8 varies with ψ_i.

Table 1. Summary statistics of G (cm) parameter for various soil types.

	Arithmetic				Geometric	
Soil	mean		std		(MC)	
texture	theo.	MC	theo.	MC	mean	std
Sand	39	40	118	156	9.9	5.3
Loamy sand	41	44	131	156	12.3	4.8
Sandy loam	64	62	186	153	22.1	4.3
Loam	105	112	475	493	17.9	6.9
Silt loam	158	156	563	544	33.5	5.8
Sandy clay loam	181	180	864	800	44.1	5.0
Clay loam	129	129	364	309	42.3	4.5
Silty clay loam	195	183	601	561	55.0	4.9
Sandy clay	219	224	909	937	48.6	5.9
Silty clay	209	204	666	583	59.0	4.9
Clay	242	232	770	689	64.1	5.0

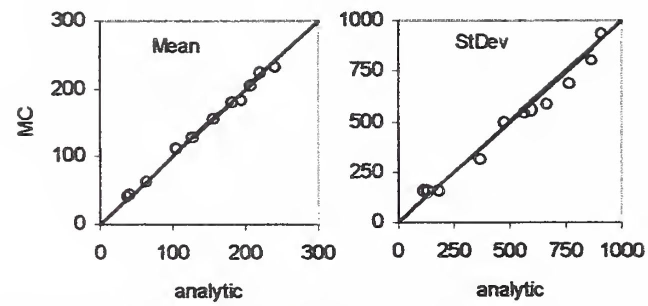

Figure 2. MC versus theoretical mean and std of G.

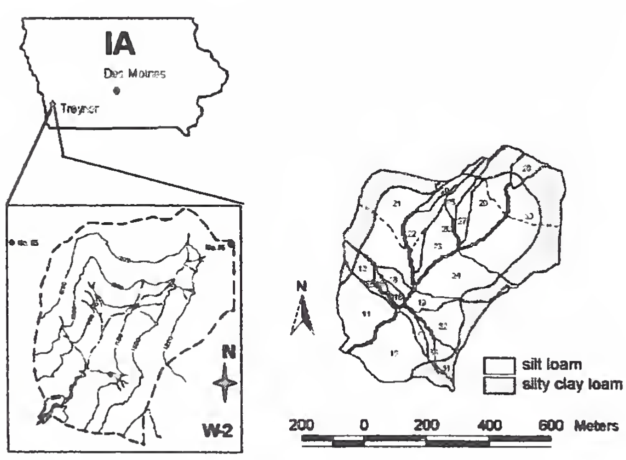

Figure 3. W-2 study watershed (left) and cascade planes, channels, and soil texture class (right).

Monte Carlo Simulations

The objective here is to identify model parameters that have the greatest impact on model output uncertainty, and compare MC results with observations in a small USDA experimental watershed (W-2). The watershed is located near Treynor, Iowa, and has an area of 83 acres (Figure 3).

Two rain gauges (115 and 116) measure rainfall intensity in the watershed. The watershed has a rolling topography defined by gently sloping ridges, steep side slopes, and alluvial valleys with incised channels that normally end at an active gully head (Kalin et al. 2003, and references therein). Slopes vary from 2 to 4 percent on the ridges and valleys and 12 to 16 percent on the side slopes. The major soil types are well drained and consist of silt loam (SL) and silty clay loam (SCL) textures that are very prone to erosion, requiring suitable conservation practices to prevent soil loss. Corn has been grown continuously on W-2 since 1964. Figure 3 (to the right) shows spatial extent of soil texture in the W-2 watershed and the overland flow planes, marked by solid line boundaries, used in the simulations. To assess model output response to model input parameters' uncertainty, we performed Monte Carlo simulations using KINEROS2. This is accomplished by generating one thousand set of independently distributed random values of the parameters λ, ψ_b, K_s, S_i, n_c, and n_p, ϕ, c_g, c_f, I, and $D_{50}$from their respective probability distributions, then performing 1000 model runs, one for each randomly generated parameter set. Above, n_c, and n_p, respectively, are the channel and plane Manning's roughness; S_i is the initial soil saturation; ϕ is the soil porosity; I is interception depth; and D_{50} is the median particle size.

Kalin and Hantush (2003, and references therein), provide the range and distributions of key soil and model parameters. We generated the distribution of G parameters using equation (9) and lognormally distributed λ and ψ_b (Rawls et al. 1982). Figure 4 shows probability of exceeding peak sediment discharge rate Q_s (Kg/s), sediment yield (tons), and time to peak sediment discharge (min) for each of the above randomly generated parameters λ, G, K_s. S_i, n_c, and n_p. For example, the curves corresponding to K_s are obtained by sampling its values from its lognormal distribution while fixing all other parameters at their mean values. Plane and channel Manning roughness coefficients n_p and n_c are assumed to be uniformly distributed.

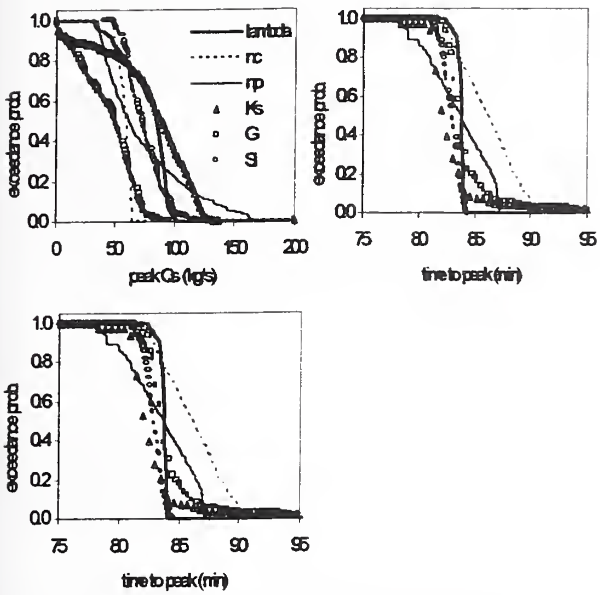

Figure 4. Probability of exceedance of peak sediment discharge rate (Kg/s), sediment yield (tons), and time to peak sediment discharge (min) for different randomly distributed parameters.

A sudden drop from 1 to 0 in the exceedance probability implies no variation of the model output with respect to the particular parameter uncertainty, whereas the more gradual the transition from 1 to 0, the more sensitive the model output to the parameter. Both the peak sediment discharge and sediment yield are highly sensitive to K_s, G, S_i, and n_p, and to a lesser extent to λ (with ψ_b fixed at its geometric mean). Uncertainty in the channel Manning roughness n_c has almost no impact on the model output (a sharp transition from 1 to 0 as shown in Figure 4). Parameter λ mostly affects Q_s and to a lesser extent the sediment yield. We note that sensitivity with respect to λ is more pronounced than what is reported by Kalin and Hantush (2003), which is rather expected, since this parameter generally affects the net capillary drive G parameter. Although λ affects model output only through the G parameter, allowing ψ_b to vary randomly, but independently, with λ explains the more gradual transition from 1 to zero of the probability exceedance curve for G than that for λ, indicating a greater uncertainty of the model output with respect to the former. The Manning roughness coefficients n_p and n_c have the greatest impact on exceedance probability of time to peak sediment discharge, with K_s and G having rather a moderate effect, as the last of Figure 4 shows. S_i and λ have the least effect on the distribution of time to peak sediment discharge rate. Using MC simulations, Kalin and Hantush (2003) showed that for large rainfall events, peak sediment discharge and sediment yield is highly influenced by uncertainty in the hydraulic erosion parameter c_g and much less sensitive to the rain splash erosion parameter c_f. What was interesting, however, is that this mode of sensitivity is reversed for smaller events, where rain splash erosion dominates model output uncertainty.

Figure 5 shows the median, 25% quartile values, 5% quartile values, and observations of runoff Q_f and sediment discharge Q_s for the large and small rainfall events shown in Figure 6. The 75% and 95% quartile values, which along with the 25% and 5% quartile values bracket the 50% and 90% confidence intervals, respectively, are not shown for the purpose of clarity. The results are obtained by performing MC simulations with the above model parameters, including c_g (0.01-1 s^{-1}) and c_f (100-1000 s), randomly generated. The two extreme values of Parlange et al. infiltration parameter α are used; i.e., $\alpha = 0$ and 1. Although $\alpha = 0.85$ is recommended (Parlange et al., 1982), the results in Figure 5 shows that this parameter has almost no impact on the median of both runoff and sediment discharge except in the vicinity of peak values. For both events, the median significantly over estimated the observed Q_f and Q_s during the rising parts of the hydrograph and sedimentograph and early portions of the recession curve. It is remarkable that the median, and without calibration, simulated fairly well the observed values of the flow and sediment discharge rates during later portions of the recession; roughly, during the time period from 90th to the 120th minutes for the larger of the two events in Figure 7, and from 70th to 100th minute for the smaller event. Overall, the median of both runoff and sediment discharge rates are within order of magnitudes of the observed values for the larger event. More than 50% of the observations are within the median (± 25% quartile values). Within this confidence interval, the probability is 50% that flow or sediment discharge would be observed, provided that the model approximates the underlying physical processes reasonably well. All measurements fall within the 90% confidence interval, corresponding to the median ±45% quartile values.

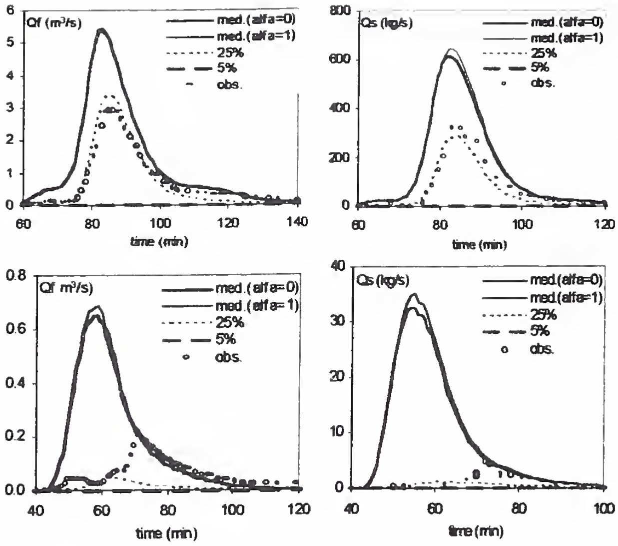

Figure 5. MC simulations for large (top two) and small (bottom two) rainfall events: median and quartiles.

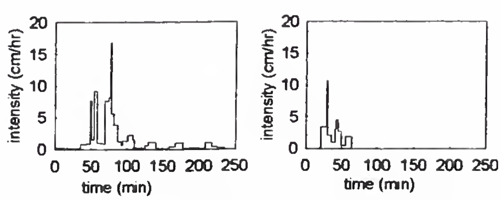

Figure 6. Rainfall events on 6/13/83 (left) and 8/26/81 (right).

Conclusions

This paper applied the event-, physically-based, and distributed runoff/erosion model KINEROS2 to an experimental watershed in Iowa. The net capillary drive, G, parameter is a key parameter to this model, which affects both runoff and sediment transport. Statistics of the net capillary drive parameter were obtained and tabulated for all soil texture classes. These values can be used in future applications of KINEROS2. Monte Carlo simulations were conducted to assess the impact of uncertainty in model parameters on the variation of sediment discharge rate, sediment yield, and time to peak sediment discharge rate. Comparison with observations of runoff and sediment discharge rates of the median, median ± 25% and ± 45% quartile values, obtained by performing Monte Carlo simulations indicated that KINEROS2 performed well given the uncertainties in model parameters as reported in the literature. The model can be calibrated successfully without fear of producing artifact model parameters.

Acknowledgments

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Relations between Hydrology and Solute Fluxes at the Five Water, Energy, and Biogeochemical Budget (WEBB) Watersheds of the United States Geological Survey

Richard M.T. Webb, Norman J. Peters, Brent T. Aulenbach, James B. Shanley

Abstract

A clear understanding of how natural watersheds receive, transform, and export solutes can lead to improved environmental management. With this background, decision makers can better anticipate how natural and anthropogenic stressors will alter the existing equilibrium.

A principal component analysis was used to identify statistical relations between hydrologic conditions and net exports of cations (H^+, Ca^{2+}, Mg^{2+}, Na^+, K^+, NH_4^+), anions (HCO_3^-, Cl^-, SO_4^{2-}, NO_3^-) and silica (H_4SiO_4) for the five U.S. Geological Survey (USGS) Water Energy and Biogeochemical Budget (WEBB) sites. The input data consisted of six years of monthly averages of daily simulated precipitation, snow melt, evapotranspiration, saturated overland flow, infiltration excess overland flow, macropore flow, recharge of the root zone from the saturated zone, soil moisture content, groundwater discharge, and net solute fluxes.

Five principal components account for 83 percent of the variance in observed in net fluxes of water and solutes at the WEBB watersheds. The components appear related to basic hydrologic controls on the net exports of major ions common to these five hydroclimatologically distinct sites. Each component describes two diametrically opposed conditions: (1) Wet (storm or melt)/Dry (drought or freeze) (50 percent of variance). All solutes, with the exception of ammonia, were correlated with this component; (2) Dry periods (with cool, wet soils) /Wet periods (with warm soils with available root zone storage) (14 percent of variance). Nutrients and sulfate were correlated with this component; (3) Dry soils (during warm, dry periods)/Wet soils (during cool, wet periods) (8 percent of variance). Nitrate and chloride were weakly correlated with this component; (4) Low base flows with limited recharge /Moderate baseflows with some recharge (7 percent of variance). Weathering products were positively correlated and nutrients and sulfate negatively correlated with this component; and (5) Spring melts or rains on dry soils/late summer rains on wet soils (4 percent of variance). Ammonium was negatively correlated with this component.

These results describe generic relations observed at all sites. Statistical models exist that better describe the variance at any single site.

Keywords:

watersheds, geochemistry, hydrology, surface water model, principal component analysis, elemental fluxes

Introduction

To clarify first-order processes governing the quantity and quality of water flowing from the headlands of small, forested watersheds (less than 100 km²), the U.S. Geological Survey initiated the Water, Energy, and Biogeochemical Budget (WEBB) project in 1992

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(Baedecker and Friedman 2000). Five forested upland sites were chosen for long term study: Loch Vale, Colorado; Trout Lake, Wisconsin; Sleepers River, Vermont; Panola Mountain, Georgia; and Luquillo Forest, Puerto Rico. Each site is unique in its climate, geology, soils, and vegetation (Table 1).

A key objective of the WEBB program is to understand how basic hydrologic and biogeochemical processes active at each site may respond to changes in precipitation chemistry (acid rain) and temperature (global warming). The objective of this study was to identify the dominant linkages between the hydrology and biogeochemistry for developing physical and geochemical models. Principal component analysis (PCA) of variables describing the hydrology and net solute exports was used to identify common linkages across the diverse hydroclimatic regimes occupied by the WEBB sites.

Methods

The PCA variables consisted of six years of monthly averages of daily precipitation, snow melt, evapotranspiration, saturated overland flow, infiltration excess overland flow, macropore flow, recharge of the root zone from the saturated zone, soil moisture content, and groundwater discharge, each derived from a hydrologic simulation, and net solute fluxes derived from a mass balance of precipitation inputs and streamwater outputs from each watershed.

XTOP_PRMS

Hydrologic fluxes and average soil moisture content deficit were simulated using XTOP_PRMS, a hydrologic model built in the USGS Modular Modeling System (MMS)(Leavesley et al. 1998) using modules built from the Precipitation Runoff Modeling System (PRMS) (Leavesley et al. 1983), the National Weather Service's HYDRO-17 snowpack model (Anderson 1973); and a multi-catchment version of TOPMODEL (Beven and Kirkby 1979, Beven 1997) modified and coupled with the other MMS modules. The TOPMODEL module in XTOP_PRMS differs from the original TOPMODEL in three ways: A fraction of recharge can be routed directly to the stream to simulate interception by macropores such as worm borrows or roots (Piñol and others, 1997); negative saturation deficits (artesian conditions) are used to replenish root zone deficits with any excess accounted for as exfiltration; and vertical hydraulic conductivities are modeled as having a log-normal distribution described by a median and a coefficient of variation.

Spatial parameters were derived using the GIS Weasel (Viger, 1998). The Weasel is a GIS interface used to delineate, characterize, and parameterize basin features. It is composed of ArcInfo* (ESRI, 1992) GIS software, C language programs, and shell scripts. Spatial parameters needed by XTOP_PRMS include elevation, slope, aspect, topographic index, soil type, available water-holding capacity of the soil, vegetation type and cover density, the dominant radiation planes (zone receiving similar radiation loadings), interception-storage capacity, and stream topology.

Daily precipitation totals, and minimum and maximum temperatures provide the temporal data needed to drive the model. Intrinsic parameters affecting processes such as snowmelt and fluxes within the hillslope were manually calibrated to obtain a good match between the observed hydrograph and the simulated one. The Nash-
Sutcliffe efficiency (Nash and Sutcliffe 1970) values for the manually-calibrated models were 0.71, 0.24, 0.79, 0.68, and 0.60 for Loch Vale, Allequash, Sleepers River, Panola Mountain, and Icacos respectively. The poor calibration for the Allequash is probably because beavers dam the stream at times, resulting in extreme fluctuations unrelated to precipitation or snowmelt. The overall affect of the beavers on the hydrologic model results is mitigated by monthly averaging of daily flows.

Hydrologic variables

Fluxes between hydrologic compartments and the soil moisture in the root zone were normalized by the basin areas to obtain depths per unit area. Monthly total (or average in the case of soil moisture content) fluxes of the daily values were then standardized to yield a distribution with a mean of zero and unit variance. This step was necessary to avoid principal components from

* Use of trade names is for identification purposes only and does not imply endorsement by the U.S. Government.

Table 1. Characteristics of the five U.S. Geological Survey Water, Energy, and Biogeochemical Budget sites.

WEBB site	Loch Vale[1]	Trout Lake[2]	Sleepers River[3]	Panola Mountain[4]	Luquillo[4]
Geographic Province	Southern Rocky Mountains	Northern Highland	New England piedmont	Southern piedmont	Caribbean island arc
Catchment used for water quality model	Andrews Brook	Allequash	W-9	Lower Gage	Icacos
National Acid Deposition Site used for solute inputs	Loch Vale - CO98	Trout Lake – WI36	none	none	El Verde - PR20
Catchment area for hydrologic model (ha)	690	4000	41	41	326
Outlet elevation (m)	3215	494	524	222	616
Highest elevation (m)	3850	555	679	279	844
Climate Type	Cold continental	Humid continental	Humid continental	Humid continental / subtropical	Humid tropical
Mean Annual Temperature (°C)	0	4.5	6	16	21
Mean Annual Precipitation (mm)	1230	760	1100	1245	4320
Percentage of Mean Annual Precipitation as snow (percent)	85	15	25	<1	0
Ecosystem type	Taiga - boreal forest / alpine tundra	Northern lakes and forests	Northern hardwood forest	Southern hardwoods	Subtropical lower montane wet forest
Percent forest cover	2	84	100	91	99
Surficial geology	Thin soil / talus	Glacial drift	Calcareous silty glacial till	Weathered colluvium / alluvium	Colluvium, frequent landslides
Average depth to bedrock (m)	0 to 5	30 to 50	1 to 4	0 to 5	4 to 15
Bedrock Type(s) and areal percentage	Biotite schist	Granite/ amphibolite	Phyllite / calcareous granulite	Granodiorite/ amphibolite	Quartz diorite

1 (Baron 1992, Campbell et al. 1995, Clow et al. 2000)
2 (Walker and Bullen 2000)
3 (Shanley 2000)
4 (Peters et al. 2000)
5 (Larsen and Stallard, 2000)

identifying individual site processes rather than common intersite processes. Eight hydrologic flux variables and one state variable (root-zone soil moisture content) were evaluated with respect to 10 solute fluxes using principal component analysis. The hydrologic variables were selected because of their anticipated explanatory power for describing biogeochemical fluxes.

- Net precipitation (NP) – is the amount of precipitation reaching the land surface after subtracting the amount that is intercepted by the forest canopy and evaporated back to the atmosphere.
- Snowmelt (Melt) – is calculated using a temperature index method during days with no rain (Anderson 1973). On days with rain, a more complete energy balance is computed. Energy transfers are scaled according to the estimated percent snow-covered area.
- Actual evapotranspiration (ET) – is equal to 100 percent of the potential evapotranspiration (Hamon 1961) during periods of active transpiration when the root zone is at field capacity; ET reduces linearly to zero at the wilting point.
- Overland flow from infiltration-excess (OFInfex) – occurs when the precipitation rate exceeds the capacity of the soils to absorb it (Green and Ampt 1911, Horton 1939). This results in a perched water table.

- Saturated overland flow (OFSat) – occurs when precipitation falls on areas where the water table is at the surface (Dunne and Black 1970).
- Root zone moisture (SMCont) – increases with recharge from rain or melt and decreases during active evapotranspiration or by recharge by exfiltrated waters.
- Flux of water from the saturated zone to the root zone (RZWet) – occurs during dry periods in areas of the catchment where exfiltration is predicted to occur.
- Macropore flow (MF) – Recharge in excess of field capacity infiltrates through the unsaturated zone if storage is available. A fraction of the recharge is routed directly to the stream as macropore flow.
- Baseflow and exfiltration (BF) – Following the standard TOPMODEL concept, baseflow increases exponentially as the water table nears the surface.

Where the model predicts zero flux for the entire time period, such as melt at Luquillo or infiltration excess at Allequash, the time series of standardized variables was set to zero.

Deposition and export of solutes

The deposition and export of major cations (H^+, Ca^{2+}, Mg^{2+}, Na^+, K^+, NH_4^+), anions (HCO_3^-, Cl^-, SO_4^{2-}, NO_3^-) and silica (H_4SiO_4) are routinely measured for each WEBB site. In the absence of any internal chloride sources, any chloride leaving a watershed must have arrived through atmospheric deposition. At the other extreme is silica, the most common element, other than oxygen, in the crystal lattices of the bedrock minerals. The other species vary in that they are found both in the atmosphere and in bedrock minerals or vegetation.

Input chemical fluxes were computed using methods developed for the National Acid Deposition Program (NADP) (Dossett and Bowersox 1999). NADP maintains collection sties at or near Loch Vale (CO98), Allequash (WI36), and Icacos (PR20). At Sleepers River and Panola Mountain precipitation samples were collected and analyzed using NADP protocols. Concentrations of major solutes were multiplied by the volume of precipitation to arrive at spatially averaged input fluxes.

Output fluxes were estimated using charge-balanced flux models created for each solute. Terms include a hyperbolic fit of the concentration and discharge relations and annual and semiannual sine and cosine terms. Fluxes were normalized by basin area to describe output fluxes in units of millimoles or milliequivalents per square meter. Modeled concentrations were adjusted to observations through linear interpolation of residuals between observations (Aulenbach and Hooper 2001).

Monthly input fluxes were subtracted from monthly output fluxes to obtain net fluxes. Wherever output chemistry was unavailable for a site during water years 1992 through 1997, the mean monthly export from the remainder of the series was inserted. Ammonia concentrations never exceeded detection levels for stream samples collected at Loch Vale, Trout Lake, and Sleepers River; net exports were equal to the negative of the input flux at these sites. The net values for each solute flux for each site were then standardized to mean zero and unit variance.

Principal Component Analysis

Principal Component Analysis (PCA) is a linear dimensionality reduction technique, which identifies orthogonal directions of maximum variance in the original data, and projects the data into a lower-dimensionality space formed of a subset of the highest-variance components (Bishop 1995).

Input for the all-site PCA described here consists of 360 observations of 19 variables (9 hydrologic and 10 chemical variables); The 360 observations consist of 72 months (October 1991 through September 1997 - water years 1992-97) of observations for each of the five WEBB sites. Individual PCA models were also run for each site to compare the complexity of the individual hydrobiochemical systems. The data for each of these was 72 observations by 19 variables for Loch Vale, Sleepers River, and Panola Mountain. The Icacos PCA included 18 variables (no snow melt), and the Allequash PCA included 17 variables (highly permeable soils results in no infiltration excess or root zone wetting).

Results

The number of components needed to explain the variance in all observations in a dataset reflects the complexity of the overall system. In general, strong seasonality of hydrologic fluxes and associated solute fluxes corresponds to simpler systems that can be described with fewer components. Only two components account for more than 80 percent of the variance for the Loch Vale PCA model. Three components are required to account for a similar amount of variance at Panola Mountain and Sleepers River, and four for Allequash and Icacos. To explain 80 percent of the variance for the combined data set of all sites, five components are needed (Table 2). Each of these five can be related to general processes present to some degree at all sites, but would not necessarily be an optimal descriptor at any given site.

Table 2. Component loads for the all-site PCA. Loads less than 0.1 are not shown. [Abbreviations: Comp; component; BF, baseflow and exfiltration; ET, evapotranspiration; Melt, snowmelt; MF, macropore flow; NP, net precipitation; OFInfex, overland flow from infiltration excess; OFSat, saturated overland flow; RZwet, flux from the saturated zone to the root zone; SoilM, root-zone soil moisture; ANC, acid-neutralizing capacity; Ca, calcium; Cl, chloride; H2SiO4, silicic acid; K, potassium; Mg, magnesium; Na, sodium; NH4, ammonium; NO3, nitrate; SO4, sulfate]

	Comp.1	Comp.2	Comp.3	Comp.4	Comp.5
Proportion of Variance (in percent)	50	14	8	7	4
BF	0.26			-0.12	0.25
ET		-0.35	0.53		-0.11
Melt	0.23				0.59
MF	0.25	-0.16	-0.19	-0.23	0.26
NP		-0.41	-0.40		-0.36
OFInfex	0.17	-0.33		-0.19	-0.40
OFSat	0.24	-0.23		-0.35	
RZwet	0.23	-0.24	0.20	-0.24	
SoilM	0.10	0.14	-0.63		
ANC	0.28			0.30	
Ca	0.30			0.20	
Cl	0.26	0.13	0.15	0.18	-0.13
H2SiO4	0.29			0.13	
K	0.29				-0.10
Mg	0.29			0.31	-0.11
Na	0.29			0.27	-0.13
NH4		0.45		-0.37	-0.38
NO3	0.15	0.37	0.20	-0.39	
SO4	0.25	0.25		-0.25	

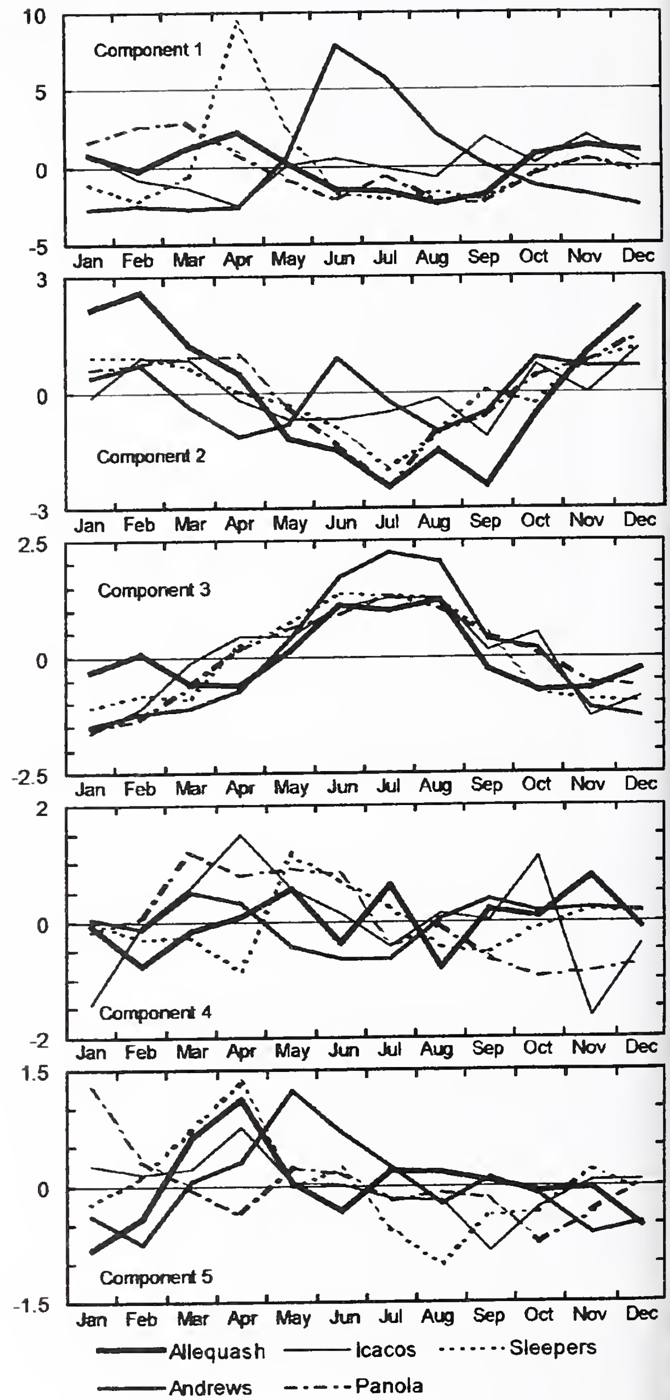

Figure 1. Mean monthly scores for the first five principal components of the combined hydrologic- geochemical WEBB data set.

Hydrologic and biogeochemical significance of the first five principal components

A component consists of a combination of the original variables. The loading of each variable in Table 2 describes how heavily weighted that variable is on the component. The loadings are the coefficients of the principal component transformation. For each observation (month) in the original data set, a score for each component was also computed. A score indicates the relative presence and sign for each component for each observation. The sign indicates which of the two diametrically-opposite hydrologic conditions is present during a given month. Seasonal patterns can be discerned by plotting mean monthly scores (Figure 1).

Component 1 - Flushing/freezing (50 percent of variance): Accumulated solutes are flushed from the watershed by intense rain or snow melt or alternately retained when precipitation and solutes are locked up when the basins freeze or enter a drought.

Component 2 – Dry periods (with cool, wet soils) /Wet periods (with warm soils with available root zone storage) (14 percent of variance): Retention of ammonia, nitrate, and sulfate is less during dry and cool periods with saturated soils than it is during wet warm periods with available root zone storage.

Component 3 - Dry soils (during warm, dry periods)/Wet soils (during cool, wet periods) (8 percent of variance): This component describes the upward flux of water from the saturated zone into drying riparian soils during periods of high evapotranspiration. Exfiltration through desiccating surfaces increases the net export of nitrate and chloride; during wet and cool periods, the nitrate and chloride in the precipitation may move from the base of wet soils down to mix with ground water as might occur during ground water ridging.

Component 4 - Low base flows (with limited recharge) /Moderate baseflows (with some recharge) (7 percent of variance): During very low flows, ions from deep in the soil profile are released; nutrients and sulfate are tightly retained near the surface. During moderate recharge events the nutrients and sulfate exports are rinsed into a more saturated soil profile to be released in the base flow as the contribution of base cations diminishes.

Component 5 - Spring melts or rains on dry soils/late summer rains on wet soils (4 percent of variance): Ammonia is taken up by growing vegetation in the spring. Mineralization of organic debris reintroduces the ammonia into the system to be released during late summer rains when transpiration begins shutting down.

Conclusions

The route that water from rain and snowmelt takes on its way to the stream or back to the atmosphere affects the stream chemistry. Upcoming modeling efforts directed at simulating the hydrologic and biogeochemical processes active in this diverse set of watersheds should be able to incorporate the following:

- Precipitation chemistry and dry deposition
- Weathering of bedrock and subsequent release of cations and alkalinity
- Soil processes including ion exchange and redox variations in the root zone resulting from different ET rates.
- Energy balances for accurate estimates of evapotranspiration and rates of chemical and biological reactions in the soils.
- Biologic uptake

The water coursing through the nation's rivers carries with it a myriad of constituents, some innocuous, some toxic. Truly pristine waters are scarcer every day as development continues into once remote headwaters. By evaluating the present hydrologic state of a watershed and the forecasted climatology, the fate of solutes deposited in precipitation and those applied to or naturally existing in the watershed can be reasonably predicted.

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Operational Modeling of Soil Moisture at Local and Regional Scales

Christa Peters-Lidard, Yihua Wu, Mike Tischler, Peggy O'Neill

Abstract

Soil moisture is one of the basic links between the water and energy cycles of land surfaces through its regulation of infiltration, runoff, transpiration and thermal capacity. In this study, an operational modeling system of soil moisture at hillslope (local) to watershed (regional) scales has been developed using the community Noah land-surface model (LSM). This system simulates profiles of soil moisture (both liquid and frozen) and soil temperature, skin temperature, snowpack depth, snowpack water equivalent (and hence snowpack density), canopy water content, and surface water and energy fluxes, including runoff, infiltration, and evapotranspiration. The system was tested using soil moisture data from the Monsoon '90 experiment, carried out at the Walnut Gulch Experimental Watershed (WGEW), near Tombstone, Arizona. The results show that the system has the potential for operational soil moisture modeling.

Keywords: soil moisture, land-surface model, Monsoon '90, soil-vegetation-atmosphere transfer

Introduction

Routine, or operational, estimates of hillslope-to-watershed scale soil moisture have potential applications in regional resource management, including flood and water resource forecasting, irrigation scheduling and determining mobility with lightweight vehicles. Hillslope scales may be defined as 10 to 100 m and watershed scales range from 1,000 to 25,000 km^2. Watershed management applications generally require daily soil moisture information to depths ranging from the sub-surface (15 cm) to the entire root zone (>1 m), while remote sensing-based products provide only surface soil moisture at depths ranging from 1-5 cm over bi-weekly to monthly intervals. Therefore, we are developing a combined approach using Soil Vegetation Atmosphere Transfer (SVAT) models and remotely sensed observations to provide routine daily estimates of profile soil moisture.

SVAT models have generally been developed for weather and climate modeling applications, and typically include solution of a form of the Richards' equation, including representation of parameters and processes controlling the evolution of soil moisture such as infiltration, evapotranspiration, percolation and drainage (see also Moran et al., in review). In this work, we apply a publicly available SVAT model known as the community Noah LSM (Chen et al. 1996), which has been validated at the point and watershed scale with respect to its water and energy balance predictions. In the following sections, we describe the application of the model to the Monsoon '90 field program conducted in WGEW, and assess the suitability of the SVAT model for operationally predicting hillslope-to-watershed scale soil moisture.

Monsoon '90

In the summer of 1990, the Monsoon'90 large-scale interdisciplinary field experiment was conducted in the 148 km^2 WGEW (Figure 1). During Monsoon'90, daily gravimetric soil moisture data were collected at eight micrometeorological-energy flux (Metflux) sites, in addition to standard

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meteorological variables and surface fluxes. In addition, an airborne L-band Push Broom Microwave Radiometer (PBMR) mounted on a National Aeronautics and Space Administration (NASA) C-130 aircraft was flown at an altitude of 600m above the ground to yield soil moisture products derived from measured microwave brightness temperature Tb (Schmugge et al. 1994). Tb data were collected over an approximately 8 x 20 km area with a 40 m horizontal resolution for six days: 212 (Jul. 31), 214 (Aug. 2), 216 (Aug. 4), 217 (Aug. 5), 220 (Aug. 8), and 221 (Aug. 9), as shown in Figure 2.

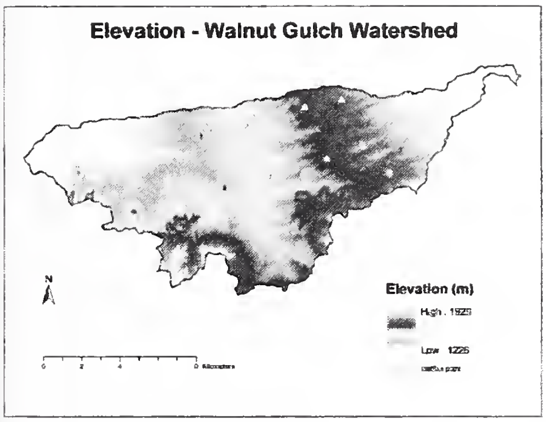

Figure 1. Walnut Gulch Experimental Watershed, showing location of 8 Metflux sites (Kustas and Goodrich, 1994).

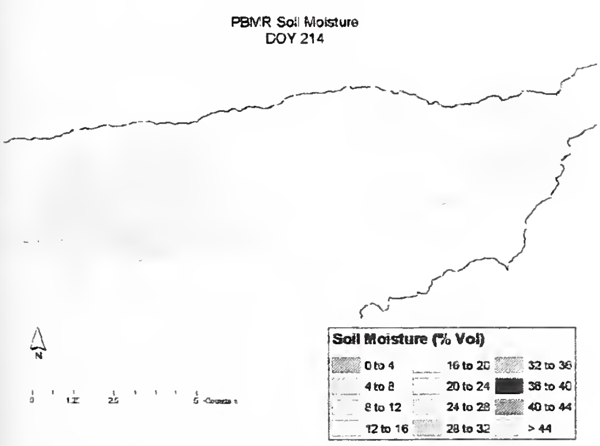

Figure 2. PBMR soil moisture product over WGEW for DOY 214 (Aug. 2) 1990.

We have assessed the overall error of the PBMR product versus the gravimetric observations collected to a depth of 5 cm, and found the expected compound error to be 4.5% $\pm$ 1.9%. Figure 3 illustrates the ability of the PBMR data to capture the hillslope-scale 5-cm soil moisture over time during Monsoon '90, as measured by gravimetric methods and converted to volumetric units. As the figure shows, the PBMR data is able to capture the dynamic range of the soil moisture observations.

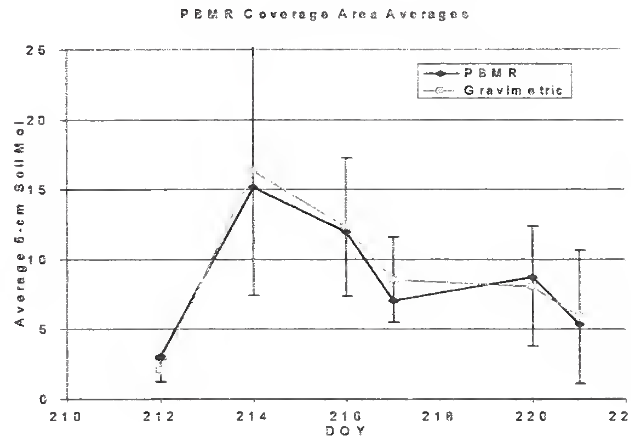

Figure 3. Time series of 5-cm soil moisture measured at the 8 Metflux sites by PBMR and gravimetric sampling. Error bars indicate the standard deviations of gravimetric observations, based on 3 samples per metflux site, or 24 samples total.

Modeling Approach

The publicly available community Noah LSM (Chen et al. 1996), is applied to the WGEW for Monsoon '90 by configuring the model to execute over a 660 by 333 grid domain with a horizontal resolution of 40m. There are four vertical soil layers, with thicknesses of 5, 25 60 and 75 cm. To provide time for "spin-up", simulations commenced at 0000 GMT, DOY 204 and ended at 2300 GMT, DOY 227. Initial soil moisture and temperature at the four depths are interpolated from observations at the eight Metflux sites. Input parameters and near-surface atmospheric forcing data are required as described below.

Forcing data

The NOAH LSM requires input forcing data that includes precipitation, solar radiation, long wave radiation, temperature, relative humidity, wind speed and surface pressure. The dynamic meteorological forcing data were obtained from Houser (1996), who applied a state-of-the-science interpolation algorithm to precipitation data collected at 88 rain gauges deployed over the watershed. All other meteorological forcings were assumed to be spatially constant, given that they were available only at some of the Metflux sites.

Parameters

Soil texture and land cover data are required to specify hydraulic, thermal and radiative parameters required by the LSM. Soil texture data sets considered for operational use include, from finest to coarsest resolution, Soil Survey Geographic (SSURGO), Soil Dataset, State Soil Geographic (STATSGO) Soil Dataset, and Soil texture from Food and Agricultural Organization of United Nations (FAO). Interestingly, there is only one soil type in the Walnut Gulch watershed in both STATSGO and FAO soil classifications: loamy sand for STATSGO and Sandy loam for FAO. All soil texture data were mapped to the texture classes of Cosby et al. (1984), as shown in Figure 4 for SSURGO data, since Noah uses their lookup tables to determine soil hydraulic parameters. In addition to the soil texture data sets, Saturated Hydraulic Conductivity and Porosity were derived directly from the SSURGO data as an optional input to the LSM.

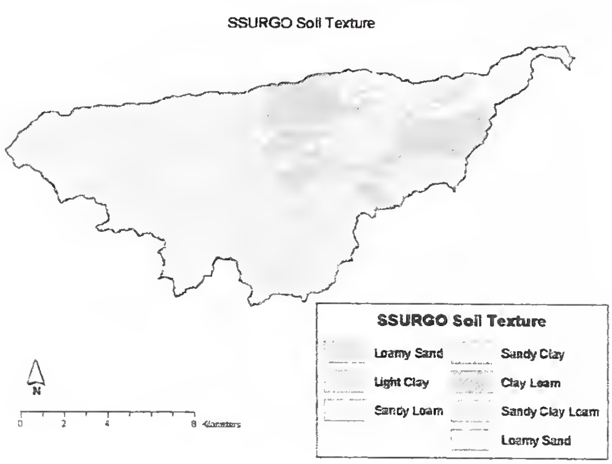

Figure 4. SSURGO soil texture data reclassified according to Cosby et al., 1984.

Three land cover data sets were also considered for use in operational soil moisture modeling. They include the 1992 NALC land cover data set (NALC92), the Environmental Protection Agency (EPA) and United States Geological Survey (USGS) land cover data set (GIRAS), and Moderate Resolution Imaging Spectroradiometer (MODIS) land cover. Given that the Noah model has adopted parameter lookup tables according to the classification of Dorman and Sellers (1989), these land cover data sets were remapped to their 13 land cover types, as shown in Figure 5 for NALC, in order to utilize the parameter lookup tables within Noah. In addition, LAI, greenness fraction and albedo data were obtained from Houser (1996) as optional inputs to the model.

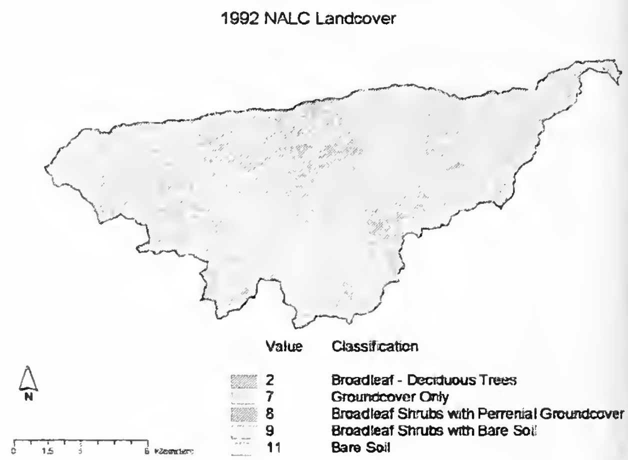

Figure 5. NALC land cover data reclassified according to Dorman and Sellers (1989).

Input degradation experiments

In addition to a control run, in which the highest resolution soil texture and land cover data sets were used along with image inputs of key soils and vegetation parameters, three sets of parameter degradation experiments were conducted. This first set of experiments explores the impact of soil parameter inputs (lookup table vs. images) as well as the impact of coarser resolution soils data on operational soil moisture prediction. The second set of degradation experiments focuses on degrading land cover inputs (tables vs. images) as well as land cover source data sets from finest to coarsest. The third set of experiments explores the impact of degrading precipitation data, by applying averaging windows to the data to simulate rainfall radar inputs as well as watershed average rainfall. The results of these experiments will be compared to that from the control in the next section.

Results

Overall, the results from the control run suggest that the Noah model is skillful in predicting watershed-scale soil moisture, as shown in Table 1. Figures 6 and 7 further illustrate the time series of watershed-scale RMS error and bias, respectively. These results suggest that the model error is close to the observed error, but that there is a persistent high bias in the model. The cause for the high bias has been further explored at the hillslope scale, as discussed below.

Table 1. Results of the control run.

RUN	MODEL	PBMR	RMS	BIAS
SSURGO + Ksat +Porosity Input	0.14	0.09	0.08	0.05

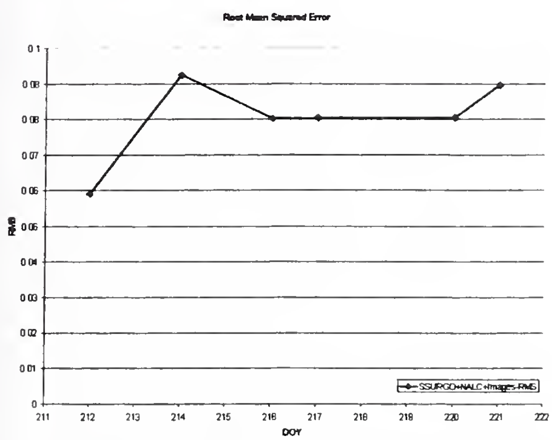

Figure 6. Time series of watershed-scale RMS error for the control run.

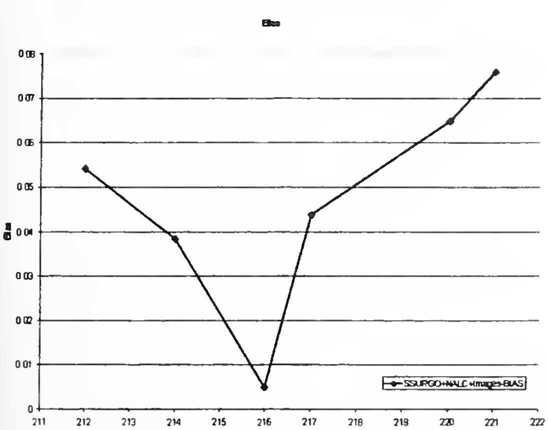

Figure 7. Time series of watershed-scale bias for the control run.

Comparisons between model predictions at the hillslope scale are made by extracting the model and PBMR 40 x 40 m pixels in the vicinity of the gravimetric sampling sites (as shown in Figure 1). This analysis suggests that the model physics is generally consistent with the observed, but that there is a persistent high bias at certain Metflux sites, as shown in Figure 8 for Metflux site 7.

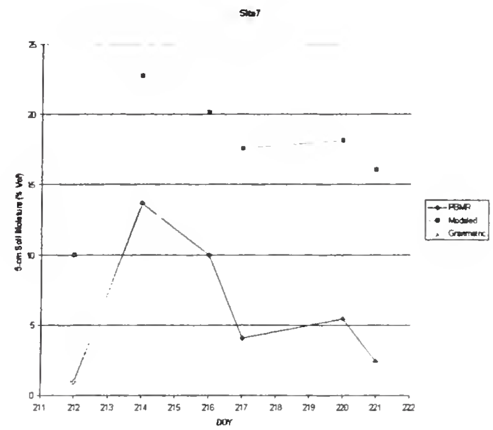

Figure 8. Time series of modeled and measured soil moisture for Metflux site 7 illustrating high bias in the initial conditions for the model.

This bias is likely the result of a poor initial condition, and the fact that the model was initialized only 8 days prior to the experiment. Further analysis suggests that bias is a strong function of soils and land cover parameters (not shown), with sites classified as "bare soil" exhibiting high bias during the wettest period, and sites classified as "loam" or "sandy clay loam" exhibiting the highest bias throughout the period.

Following the assessment of the model at the watershed and hillslope scales, an assessment of the models sensitivity to data sets was conducted in order to determine the level of effort required for operational implementation.

Soils degradation experiments

Soil data sets were degraded from SSURGO to STATSGO to FAO, in addition to replacing the SSURGO-derived images of hydraulic parameters with those using Cosby et al. (1984) lookup tables. The results of these experiments are summarized in Table 2.

Table 2. Results of the soils degradation experiments. First experiment denotes the control run, with the finest available parameter inputs.

RUN	MODEL	PBMR	RMS	BIAS
SSURGO + Ksat +Porosity Input	0.14	0.09	0.08	0.05
SSURGO + Lookup Table	0.18	0.09	0.10	0.10
STATSGO + Lookup Table	0.13	0.09	0.06	0.04
FAO + Lookup Table	0.15	0.09	0.07	0.06

As illustrated by the table, RMS errors are lower for STATSGO and FAO soils than for SSURGO, although this is likely due to the fact that the STATSGO and FAO soil classifications reclassify soils with the highest errors to those with the lowest errors. An important result is that the soil parameter lookup table clearly degrades results, as shown by comparing the control run in line 1 with the lookup table run in line 2. Given the PBMR product errors of approximately 4.5% ± 1.9%, the differences among SSURGO, STATSGO and FAO may not be statistically significant.

Land cover degradation experiments

Land cover data sets were degraded from NALC, to EPA/USGS to MODIS, in addition to replacing the Houser (1996) images of land cover parameters with those using default Noah lookup tables. The results of these experiments are summarized in Table 3.

Table 3. Results of the land cover degradation experiments. First experiment denotes the control run, with the finest available parameter inputs.

RUN	MODEL	PBMR	RMS	BIAS
NALC+Greenness+Albedo Images	0.14	0.09	0.08	0.05
NALC+Lookup Table	0.13	0.09	0.08	0.04
EPA+Lookup Table	0.18	0.09	0.10	0.09
MODIS+Lookup Table	0.17	0.09	0.09	0.08

As illustrated by the table, the soil moisture bias is slightly lower for lookup tables as compared to the control run image input, although as with the soils degradation experiments, the differences are likely not statistically significant. However, the soil moisture bias is clearly increased with EPA and MODIS land cover, suggesting that the higher resolution NALC land cover is important for operational soil moisture modeling.

Rainfall degradation experiments

Finally, a set of rainfall degradation experiments was conducted to assess the importance of high resolution rainfall input for soil moisture modeling. These experiments consisted of degrading the original interpolated rain gauge data by taking first 10x9 pixel averages, then 30x37 pixel averages, then watershed average rainfall, as illustrated in Figure 9.

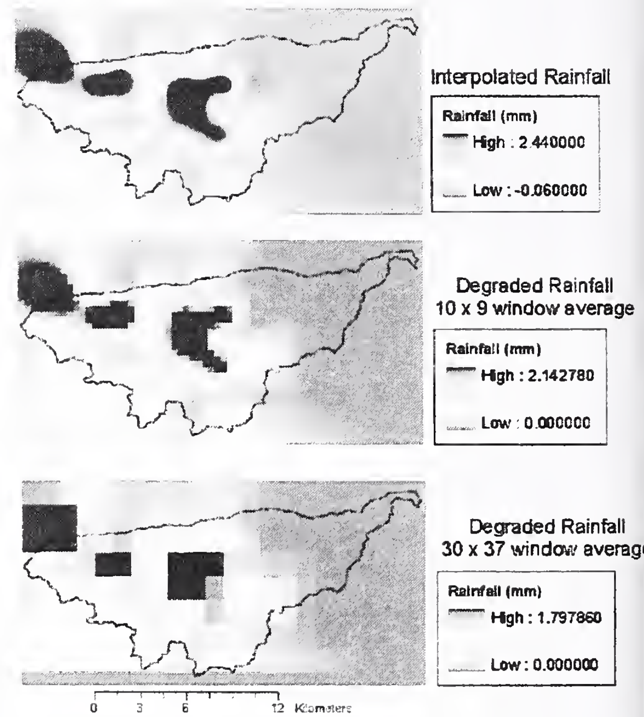

Figure 9. Original and degraded rainfall for DOY 204.

The results of these degradation experiments are summarized in Table 4, which indicates the counterintuitive result that the domain average rainfall produces the best results. This can be explained by noting the consistent wet bias caused by improper spin-up, which is counteracted by the domain average rainfall, as it tends to underestimate local rainfall rates. This leads to an overall reduction in RMS as well, since most of the error in the model is bias.

Table 4. Results of the rainfall degradation experiments. First experiment denotes the control run, with the finest available parameter inputs.

RUN	MODEL	PBMR	RMS	BIAS
Interpolated Gauge Rainfall	0.14	0.09	0.08	0.05
400m Average	0.12	0.09	0.07	0.03
1200m Average	0.12	0.09	0.07	0.03
Domain Average Rainfall	0.08	0.09	0.05	-0.01

Conclusions

In summary, a publicly available LSM has been applied to the problem of operational soil moisture prediction using data sets collected during the Monsoon '90 field program. The PBMR soil moisture products for Monsoon '90 were derived from regressions at 8 gravimetric sampling locations, with an expected compound error of 4.5%±1.9%V/V, although locally the error may be in excess of 8%.

The Noah LSM has a positive (wet) bias, which persists throughout the Monsoon '90 period. Causes of error include a poor initial condition, with high bias caused by inadequate spinup, an underestimation of losses such as evapotranspiration, likely caused by underestimates of greenness and high stomatal resistance parameters. Overall, the model is shown to adequately predict soil moisture at the watershed scale, with errors in predicted soil moisture at about 8%V/V for the "best" input data. The parameter degradation experiments indicate that using lookup tables for soil hydraulic parameters clearly degrades results, and that the results are highly sensitive to rainfall data input. Given that degraded rainfall offsets the high bias in the initial conditions, it is likely that given proper initial conditions, degraded rainfall would result in a dry bias. This suggests that spatially distributed rainfall input is critical to accurate operational soil moisture prediction.

Current and future work with this modeling system is focusing on integrating the LSM with a geographic information systems (GIS) interface in order to facilitate use of the system for applications, including regional resource management, flood and water resource forecasting, irrigation scheduling and determining mobility with lightweight vehicles. As part of this process, we are developing tools to facilitate the substantial data preprocessing activities involved, including interpolation and downscaling of coarse resolution or point-scale data sets to the hillslope scales required for these applications.

Acknowledgments

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Modeling the Potential Effects of Forest Management and Climate Change on Water Yield Across the Southeastern U.S.

Ge Sun, Steven G. McNulty, Jennifer Moore

Abstract

Forest ecosystems are expected to undergo dramatic structural and area changes in response to projected pressures of timber demands and global climate change in the 21st century. Associated with changes in forest structure and area distribution is forest water yield, which affects water availability for both humans and aquatic ecosystems needs. Diverse forest hydrology conditions exist in the southeastern U.S. due to great differences in climate, vegetation cover types, and land topography/elevation across the region. The objective of this study is to project the potential effects of forest management and climate change on annual water yield at a regional scale. A generalized annual actual evapotranspiration (AET) model was first calibrated with a watershed-scale database derived from 39 forest-dominated watersheds representing the diverse eco-regions of the Southeast. Then, the model was validated at a 0.5*0.5 degree spatial resolution (about 50 *75 km^2) across the region using historic hydrology, climate, and landcover data. Finally, the model was applied to the region using the Hadley Centre Had2CMSul climate change scenario for the region. The simple annual time step hydrologic model adequately explained the spatial variability of water yield across the southeastern region. Modeling results suggest that water yield responses to forest removal are diverse across the physiographic gradients with the highest values occurring in the conifer-dominated regions. The model predicted water yield increases as high as 450 mm/yr as a result of complete forest removal. The Had2CMSul predicts the southeastern region will become warmer and wetter during this century. As a result, this study projects that the majority of the region will experience a decrease (<170 mm/yr) in water yield during the first half of this century but an increase (<300 mm/yr) in the second half of this century.

Keywords: forest hydrology, climate change, Southeastern U.S., modeling, forest management

Introduction

Forest ecosystems are expected to undergo dramatic changes in response to projected pressures from timber demands, landuse change resulting from urbanization, and global climate change in the 21st century. The recently released Southern Forest Resource Assessment (http://www.srs.fs.fed.us/sustain) concludes that although overall the total area of forest lands did not change greatly, large areas of land have been lost to urban uses, in Florida among other states, while agricultural areas in the lower Gulf Coastal Plains, for example, have been reforested (Wear and Greis 2002). Thus, forest cover distribution patterns are projected to shift dramatically in the next 50 year due to population growth and timber price changes (Wear and Greis 2002). This landcover transformation may have significant effects on water quantity and water quality across the region. Intensive forest management practices that employ modern agriculture-type technology (i.e., short crop rotation, fertilization, ditching, irrigation) have been commonly used in the southeastern U.S. to increase timber production in a unit land area.

Water yield responses to forest management have been well studied in the southeastern U.S. and elsewhere by employing the paired small watershed approach. Studies on water-vegetation relations clearly show that forests use large amounts of water, and water yield

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from small watersheds is strongly correlated to the vegetation cover type and amount and climatic conditions (Bosh and Hewlett 1982, Stednick 1996). The most obvious and immediate watershed response to forest harvesting is increased water yield due to reduction in total ecosystem evapotranspiration, an increase in runoff, and elevated nutrient and sediment loading to streams (Swank et al. 2001). With the recovery of vegetation and ecosystem reestablishment, water quality effects diminish. Recent studies suggest that the magnitude of hydrologic response and the time required to recover to pre-disturbance levels differs greatly for various forest ecosystems (i.e., forested wetlands vs. upland mountains). The magnitude of hydrologic response is relatively higher and recovery time is longer for hilly, upland systems (Sun et al. 2001).

The recent National Assessment on Climate Change suggests that climate change and variability will have dramatic effects on both water and forests in the Southern U.S. (U.S. Global Change Program 2000). The Southern U.S. is becoming wetter (greater streamflow) due to increased precipitation) and water quality is degrading due to intensive agricultural practices, urban development, coastal processes, and mining activities. A generalized regional scale hydrologic model is needed to address the effects of climate change on water availability in forest ecosystems (U.S. Global Change Program 2000).

Watershed scale forest hydrology models have been used to simulate and explain water yield response to forest management (Swift et al. 1975, Sun et al. 1998). However, there have been few attempts to model or use historic data to explain the hydrologic responses to forest management at broad scale or to examine all the factors and interactions that affect watershed responses. Trimble and Weirich's (1987) study on large basins is the one exception. The authors found that forest recovery in the piedmont region of the southeastern U.S. has resulted in streamflow decrease from the 1920s to the 1960s. Douglass (1983) derived a general empirical equation to estimate water yield increase for the Appalachian hardwoods. The empirical model suggests that the first year hydrologic response is controlled mainly by the forest basal area removed and solar radiation received at the site. Unfortunately, the model does not include precipitation as an independent variable, thus it has limited use for similar mountain regions. The empirical WRENSS (U.S. Forest Service 1980, Huff et al. 1999) water yield methodology derived from hydrologic models and experimental data is the first effort to model hydrologic response at a regional scale. However, this approach has not been programmed and implemented for the humid southeastern region that is dominated by high rainfall, complex climatic and topographic variability. No single conceptual or computer model can describe the hydrological processes in southeastern forest ecosystems. McNulty et al. (1996) examined the potential climate change impacts on regional forest water yield with a monthly time step, stand level forest ecosystem model PnET-IIS. This model linked forest growth and productivity and water use (evapotranspiration), and proved applicable to a variety of mature forests, but it could not simulate non-forest lands. Hence, PnET-IIS has limitations for examining the effects of forest conversions and climate change impacts on landscapes with mixed land use. Existing regional scale hydrological models for global change studies are developed on watershed hydrologic principles such as HUMUS (Brown et al. 1999) or simplified water balances (Vörösmarty et al. 1998, Hay and McCabe 2002). HUMUS, modified from the Soil Water Assessment Tool (SWAT) model, was applied to the continental U.S. (Brown et al. 1999). However, the simulation results for averaged annual water yield in large basins were not satisfactory, especially for the southeastern U.S. (Brown et al. 1999).

This paper reports applications of a generalized simple water balance model to examine annual water yield response after forest harvesting, landuse change, and climatic change across a climatic and topographic gradient in the southeastern U.S. Our objectives were to assess the potential regional impacts of silvicultural practices, projected climate change, and their combined effects on evapotranspiration and water quantity and quality across the southeastern U.S. We hypothesize that the magnitude of water yield response to disturbance from management and climate change varies across the southern U.S. We also hypothesize that the impact magnitudes from landuse change and climate change are significantly different.

Methods

Databases and spatial scales

We acquired long-term (1961-1990), fine scale (4 x 4 km^2) gridded climate data for the continental U.S. (Daly 2000). We used this database for hydrologic model validation and projection of forest management effects across the southeastern U.S.

For climate change studies, we used a spatial scale of 0.5° by 0.5° (about 50 km x 75 km) that corresponds to the grid size of the VEMAP climate database (Kittel and others 1997) (Figure 1). The historic (1895-1993) and Had2CMSul climate change scenarios (1994-2099) were used to drive the validated water balance model as described in the next paragraph. Our modeling region encompasses 13 southern states from Virginia to Texas (Figure 1). The 1992 MRLC remote sensing land-cover dataset (http://edc.usgs.gov/glis/hyper/guide/mrlc#mrlc4) was used as a base regional map to display predicted hydrologic variables at a finer spatial scale, 30m, than the climate datasets (4 x 4 km^2 and 0.5° x 0.5°). Land cover was regrouped into five classes including evergreen forest, deciduous forest, crop, urban, and water body. In this study we examined one simple but extreme forest management scenario: clear-cut harvesting--representing the conversion of forests to urban land use.

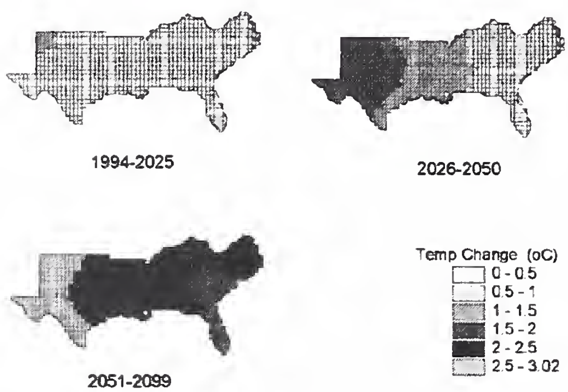

Figure 1. Had2CMSul GCM projected air temperature change (1994-2099) compared to historic climate (1895-1993) for the southeastern U.S.

Climate change scenarios

When compared to the average historical climate (1895-1993), the Had2CMSul general circulation model (GCM) suggests that the entire southeastern U.S. will have an increase in average air temperature in the range of 0.5-1.0 °C, 0.5-2.1 °C, and 0.5-1.5 °C for the time periods of 1994-2025, 2026-2050 and 2051-2099, respectively. Most of the region is projected to see an increase in annual precipitation up to 25% in the next 100 years, except southern Texas, where precipitation is predicted to decrease up to 10% (Figures 1 and 2).

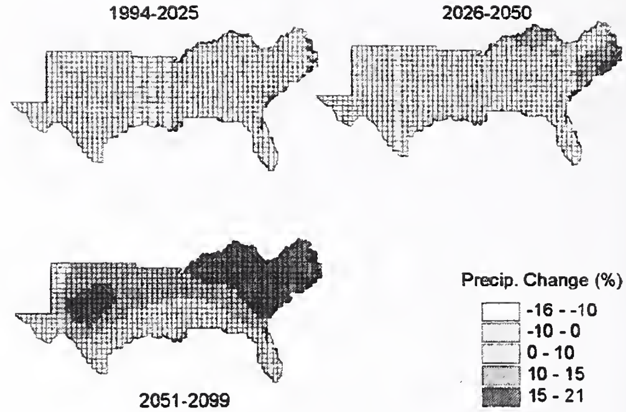

Figure 2. Had2CMSul projected annual precipitation change (1994-2099) compared to historic climate (1885-1993) for the southeastern U.S.

Hydrologic models

Annual water yield at the meso-scale (4–75 km) in this study can be estimated as the difference between precipitation received and evapotranspiration lost to the atmosphere:

$$S = P - AET \quad (1)$$

where, S = annual water yield (mm/yr), P = annual precipitation (mm/yr), AET= annual water loss from evapotranspiration (mm/yr).

The variable AET is estimated using an empirical formula (Equation 2) derived from long-term hydrologic water budget data from over 250 watersheds around the world (Zhang et al. 2001).

$$AET = \left(\frac{1 + w\frac{PET}{P}}{1 + w\frac{PET}{P} + \frac{P}{PET}}\right) \times P \quad (2)$$

where, PET is potential evapotranspiration that can be calculated from solar ration and air temperature data; w is the plant-available water coefficient and reflects the relative differences of water use for transpiration. For a grid cell with mixed land uses:

$$AET = \Sigma(AET_i * f_i) \quad (3)$$

where f_i is the percentage of landuse i.

Table 1. Calibrated model parameters and modeling schemes.

Simulation Scenario		w or AET	P (mm/yr)	PET (mm/yr)
Case 1: Current climate and land cover	Evergreen	w= 2.8	Historic Data long-term average during 1961-1990; spatial resolution 4 km^2	Computed from historic air temperature databases, long-term average 1961-1990; spatial resolution 4 km^2
	Deciduous	AET reduced by 20% from evergreen AET for the upland region		
	Crop	w=0.5		
	Urban	w=0.0		
	Water	N/A AET=P		
Case 2: Forest removal under historic climate	Hilly upland region: AET reduced 20% from evergreen forest AET with w=0.0			
	Lowland region: AET computed as w=0.0 for all species			
Case 3: Climate change, current land cover		Same as Case 1	Historic climate: 1895-1993 Future climate: Projected for the period 1994-2099	

In lieu of net solar radiation data, this study calculated PET for each watershed using Hamon's temperature based method, a simple approach but comparable in prediction accuracy to more sophisticated approaches (Vörösmarty et al. 1998, Lu et al. 2003).

Since Equation 2 is sensitive to land-cover type through the w parameter and climate change through the P and AET variables, we can use Equation 1 to project the potential hydrologic responses to forest cover manipulation and climate change.

Model calibration was first conducted at the watershed scale to obtain a w parameter dataset for different land-cover types (Table 1). The database for model calibration was derived from 39 forest-dominated watersheds with long-term (6-30 years) forest hydrology data located across the southeastern U.S. (Lu et al. 2003). Model validation was carried out at the regional scale by comparing long-term (1951-1980) average regional USGS runoff values, scaled up from a 5 km resolution (Gerbert et al. 1987), and hydrology model results computed on the 0.5° by 0.5° cell by cell basis. Model validation results were considered superior when compared to results from the continental sale hydrologic model HUMUS (Brown et al.1999). Detailed procedures and results for both model calibration and validation were reported in Sun et al. (2002).

As a result, a modeling scheme (Table 1) was developed to extrapolate Equation 1 to examine the spatial variability of water yield, and its response to forest removal and climate change at 4 x 4 km^2 and 0.5° x 0.5° spatial resolution respectively.

Results and Discussion

Spatial distribution of water yield across southeastern forests

We first examined the spatial distribution of water yield under current landcover and historical conditions (Case 1 in Table 1). Long term average Annual precipitation in the southeastern U.S. varies from over 1600 mm in the gulf coastal region to less 600 mm in western Texas, showing a strong south to north precipitation gradient. The highest precipitation area was found in the Appalachian Mountains near the border of NC, GA, and AL. Air temperature, and thus PET, generally follows the decreasing south-north gradient, but the pattern is modified by the land topography, such as the Appalachian Mountains. The climate of the coastal plain regions along the Atlantic Ocean and the Gulf of Mexico is characterized as hot and humid.

Affected by the combination of precipitation and evapotranspiration, the annual water yield from forest ecosystems varies greatly across the southeastern region (Figure 3). The hilly

Appalachian Mountain region produces the highest water yield, compared to the coastal plain and other low land in the piedmont region, due to lower temperature but higher precipitation. Other high water yield regions are coastal Louisiana and Alabama and the Ouachita Mountains.

Figure 3. Simulated average annual water yield across the southeastern U.S. with a spatial resolution of 30 m scaled down from a 4 x 4 km^2 resolution.

Deciduous hardwoods, the dominant forest ecosystem in the hilly regions with lower AET than the conifer forests mostly covering the lowlands may also explain the high water yield in the southeastern region to some extent. As expected, the lowest water yield (0-200 mm) was found in the central Texas region which is the hottest area receiving the least precipitation. We found the runoff/precipitation ranged from less than 30% in the majority of Texas and Okalahoma and the coastal plains of Florida, Georgia, and South Carolina, to over 50% in the Appalachians and in the Gulf of Mexico coastal plain.

Spatial distribution of potential water yield response to forest removal across the region

Using the parameters and databases defined in Case 2 in Table 1, we examined the potential effects of forest removal on water yield. The magnitude of water yield response to forest removal, clear-cutting in this case, is influenced by precipitation received (Harr 1983), the forest type (Swank and Douglas 1974), the amount of solar radiation received as represented by the PET variable (Douglas 1983), and land form (upland vs. lowland) (Sun et al. 2001). Combining these factors in one single formula (Equation 1) and using the modeling scheme described in Table 1, we predicted that water yield increase due to forest removal varies greatly across the region, ranging from 50 mm/yr to 440 mm/yr (Figure 4). This magnitude is reasonable when compared to the literature of watershed-scale experimental data (Bosch and Hewlett 1982, Douglas 1983). The high response regions to forest clear-cut were found in the in the Atlantic lower coastal plains, the high runoff region of the Louisiana and Alabama coasts, and the conifer-dominated region bordering Louisiana and Arkansas.

However, the model predicted about twice as high (>300 mm/yr) as reported value (0-150 mm/yr) in the literature for the groundwater-dominated coastal plain regions (Fisher 1981, Riekerk 1989, Wynn et al. 2000). This discrepancy may be explained by three arguments: 1) the definition of 'water yield' in Equation 1 is not consistent with the reported streamflow values since water yield can be recharged by groundwater storage, 2) setting the parameter w at 0.0 may not be appropriate for a forest harvest site, since the harvested site may experience large amounts of soil evaporation, especially when the ground water table is close to the surface. Under these conditions, AET would not change much from pre-harvest to post-harvest. Unfortunately, there are not enough data to estimate the w parameter for clear-cut conditions. The value of 0.0 in this study may represent the condition of an urbanized or bared landscape that was originally forested, and 3) PET values estimated for post clear-cut forests should adjusted from PET calculated for pre-harvest conditions.

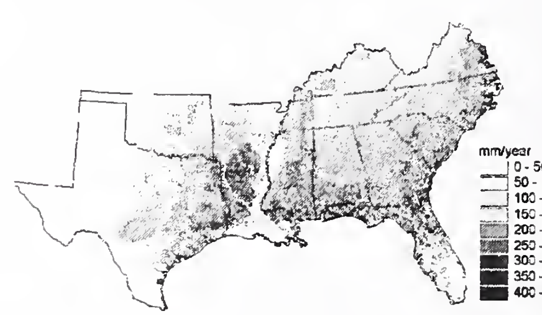

Figure 4. Simulated potential annual water yield response across the southeastern U.S. with spatial a resolution of 30 m, scaled down from a 4 km resolution.

Spatial distribution of water yield response to projected climate change

Using parameters and climate database defined in Case 3 (Table 1), we examined the effects of climate change on water yield in the next 100 years. Based on the projected GCM scenario described earlier, our simulation suggested that PET would increase dramatically, especially during the last 50 years of this century and in the state of Texas (Figure 5). The variable AET, affected by the change in both PET and precipitation, is expected to increase also for the majority of the southeaster U.S., ranging from 50 to170 mm/yr during the 2051 to 2099 period. AET is expected to decrease in southern and western portions of Texas (0-50 mm/yr).

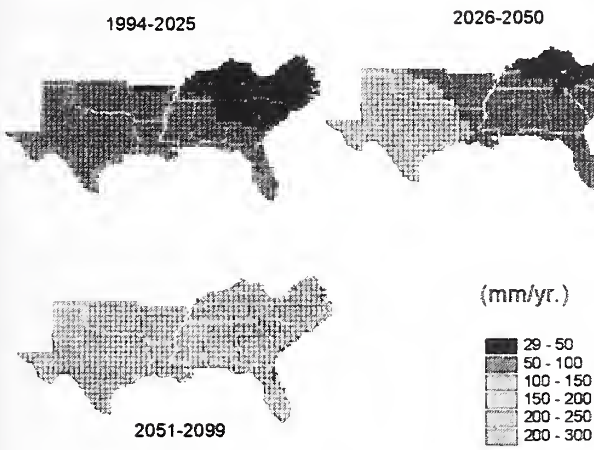

Figure 5. Predicted changes in PET across the southeastern U.S. Climate change scenarios were derived from Had2CMSul.

As a combined result from increased AET and precipitation, water yield was projected to have a rather complex pattern in space and time: an increasing trend (0-100 mm/yr) in the eastern part of the region, but a decreasing trend (0-90 mm/yr) in Texas, northwestern Oklahoma, Louisiana, and southern Mississippi during the period 1994 to 2050 (Figure 6). Water yield is predicted to increase (0-300 mm/yr) in most of the region, except the Gulf coast region and southern Florida during 2051 2099 (Figure 6).

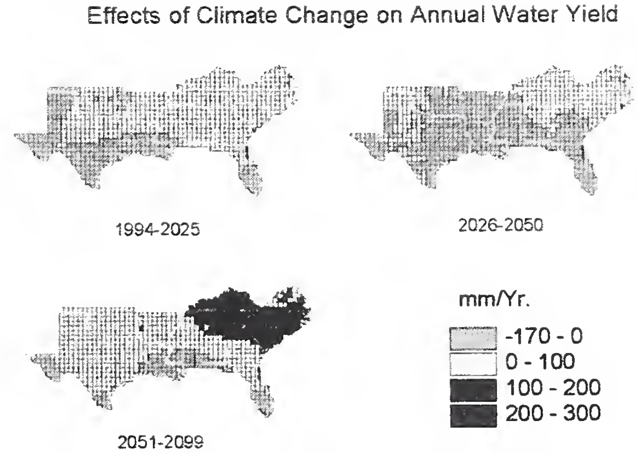

Figure 6. Predicted water yield response to climate change to across the southeastern U.S. Climate change scenarios were derived from Had2CMSul.

Summary and Conclusions

We developed a simple GIS-based modeling system to predict long-term average hydrologic response to forest harvesting and climate change at the regional scale. Regional maps of historic long-term water yield from different forest types, and water yield responses to forest harvesting and future climate change were produced. Using this model, we conclude that clear-cutting or converting forest to urban land use will increase streamflow with a large variation in magnitude across the region. The coastal plains and other conifer-dominated region that have high background runoff were identified as the regions most sensitive to forest removal. However, research is needed to reduce the uncertainty of plant water use parameters at the regional scale. Additional generalized parameters are needed to reflect the differences of hydrologic processes (changes in PET under different forest management scenarios) observed across the region.

Across the Southeast, air temperature is projected to increase up to 3.0°C while precipitation is expected to increase up to 20% for the majority of the region (Texas is exception). Consequently, annual water yield response is projected to vary both in space and time, a decrease trend in the first half the century but an increasing trend for the region.

The major concerns of climate change are decreases in local water yield for the already water-stressed region including Texas and south Florida where surface runoff is already low at present. For other regions, increases in water yield or rainfall intensity may have negative impacts such as increased soil erosion and flooding. Forest removal or reforestation may cause water yield changes (increase or decrease) with a magnitude similar to that of climate change, given the limitations of climate change scenario accuracy. Thus, forest management can play a role in mitigating the effects of climate change on water yield across the region.

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Water Quality and Quantity I

Quantification of Urbanization in Experimental Watersheds

James Bonta, William Shuster, Elizabeth Warnemuende, Hale Thurston, Doug Smith, Michael Goss, Heriberto Cabezas

Abstract

Although urbanization has a major impact on watershed hydrology, there have not been many studies to quantify how basic hydrological relationships are altered by the addition of impervious surface under controlled conditions. The USDA-ARS and U.S. EPA have jointly initiated a pilot program to study the impacts of simulated impervious surfaces on hydrology, sediment, and water quality in small experimental watersheds located at the North Appalachian Experimental Watershed, Coshocton, OH. This paper outlines the approach and rationale for using rainfall simulation, experimental watersheds subjected to natural precipitation and weather, and modeling for a multiyear project. Percent imperviousness is planned from 0% to 40% under two spatial arrangements of imperviousness - stream-channel-connected and stream-channel-disconnected imperviousness. The results from laboratory rainfall simulation will help guide the implementation of impervious surfaces in the watersheds. Preliminary evaluation of the Coshocton baseline runoff data shows that, during the time of constant land-use since 1975, annual runoff depths are similar and runoff regimes have been constant. Results from this study are applicable to the development of urban hydrology analyses, hydrology and water-quality models, design and testing of urban best-management practices, and environmental management.

Keywords: curve number, water quality, erosion, urban runoff, imperviousness

Introduction

Agricultural land is increasingly becoming urbanized due to residential and industrial development. These developments replace relatively pervious soils with impervious roofs, roads, streets, parking lots, driveways, etc. Rain falling on these impervious areas no longer infiltrates into the soil for later slow release to stream channels, but runs off quickly to adjacent pervious areas or to gutters and sewers, and eventually to stream channels. Furthermore, pesticide and nutrient use increase for lawns and landscaped areas, erosion increases, and other constituents are added to the developed watershed from deicers, leaked chemicals from vehicles on roads, etc., that under undisturbed conditions, would not be available for transport to the stream. The net effect of increasing imperviousness is to increase runoff volumes and peak-flow rates in stream channels, the

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frequency of downstream flooding, the likelihood of property damage and larger concentrations of sediment and chemical constituents, and generally to degrade aquatic ecosystems.

The U.S. EPA is interested in developing policy tools for watershed management that use market mechanisms and incentives to reduce ecological impacts and improve water quality in a cost effective manner. An example of such a mechanism is the trading of runoff-control responsibility within a watershed between those for whom it is expensive to abate with those for whom it is relatively cheap. To be ecologically effective, quantification of impervious-surface-caused runoff and efficacy of control efforts is crucial.

Many studies have been conducted using *available* stream-gauge and other data to quantify the effects of urbanization factors on runoff, sediment, water quality, and aquatic ecosystems (Schueler and Holland 2000, Shuster et al. 2003). However, results are only applicable to the generalized uncontrolled watershed conditions during the time of data collection. Research is lacking that isolates factors important to understand runoff generation and chemical transport under urbanized conditions. Investigations are needed to understand and quantify how runoff-forming and chemical-transport processes change during urbanization under controlled conditions. This will allow isolation of important factors to improve runoff and water-quality estimating procedures, provide results for science-based watershed management decisions, and to test innovative best-management practices (BMPs) that mitigate the hydrologic impacts of urbanization.

To address this issue, the U.S. EPA and the USDA-ARS jointly initiated a pilot project to investigate urbanization by utilizing experimental watersheds at the North Appalachian Experimental Watershed (NAEW) near Coshocton, Ohio. The purpose of this paper is to describe the approach being pursued and the rationale, and to present some preliminary background data on the hydrology of the experimental watersheds.

Project Objectives and Approach

Objectives for the urbanization project

The overall objective for the urbanization project is to explore and quantify the effects of land disturbances by urban development on watershed hydrology and water quality using a combination of rainfall simulation and experimental watersheds. Specific objectives are:

1. to conduct rainfall simulation experiments to evaluate general differences between connected and unconnected runoff paths to a stream channel;
2. to develop an understanding of changes in runoff-formation processes under controlled watershed conditions due to land alteration caused by urban development (i.e., increasing imperviousness);
3. to improve hydrologic modeling of urbanizing watersheds; and
4. to develop innovative urban best-management practices (BMPs) for controlling water and chemicals.

Approach

The approach to be used is empirical, exploratory, and statistical. Use of existing experimental watersheds having historical data is planned. This is because baseline data have been collected; except for additional parameters of interest for which there are no data. This also reduces the time to begin installation of imperviousness. Computer modeling will be used to extrapolate data beyond the experimental watersheds. Four experimental watersheds were selected as test beds to evaluate different levels of imperviousness over time.

Watersheds and Data

Description of watersheds

The four small experimental Coshocton watersheds chosen for study (Figure 1) are WS106, WS121, WS185, and WS192, and are located on a hilltop. Watershed areas for each are 0.63 ha for WS106, 0.57 ha for WS121, 2.99 ha for WS185, and 3.07 ha for WS192. Soils are primarily Rayne silt loams and Berks shaly silt loam. Slopes are typically of the order of 18-25%.

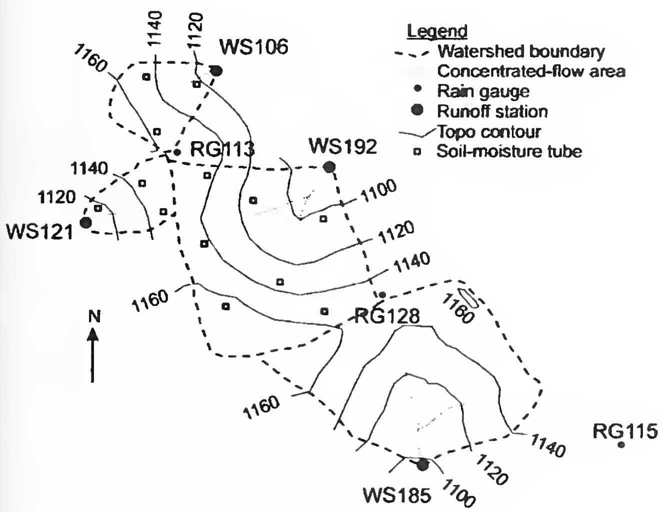

Figure 1. Experimental watersheds, soil-moisture tubes, and rain gauges at Coshocton, OH for the urbanization project.

Data available

Runoff records begin about 1940 at all sites; however, the record ended in 1972 at WS185, and in 1994 at WS192. Records for WS106 and WS121 continue to the present. Records prior to about 1975 were collected under a rotational cropping system and constant land-use records begin about 1975. Runoff data (breakpoint) were collected with H flumes.

Water quality data are available since about 1974 at the three sites with the longest records. Constituents measured include TOC, NO_3-N, NH_4-N, organic N, PO_4-P, K, SO_4, Ca, Mg, Na, Cl, Br, pH and sediment for varying record durations. Samples were obtained on a runoff-event basis by using a Coshocton wheel water sampler, which automatically samples the flow during runoff events.

Recording rain gauge RG113 is located the western part of the hilltop, and RG115 is located to the east (Figure 1). The rainfall record (breakpoint) begins in 1940 and continues to the present. Rain gauge 128 has a record from about 1942 to 1970. A 60+ year record of weather elements is also available at Coshocton.

The soil-moisture record began about 1975 at the three watersheds with long runoff records (Figure 1). Three sites in WS106 and WS121were measured through 1978, after which only one site was measured. Seven sites were measured in WS192 through 1978 and only at one site after this time. Frequency of measurement was every two weeks and then decreased to monthly measurements in the 1990s. Soil moisture was measured with a neutron probe in depth increments of 6 in to a depth of about 1 m, and gravimetrically in the top 7-9 in.

Detailed soil and topographic maps are available, and a first order soil survey is being conducted as well. Some soil physical characterization data are also available.

Land-use at the three sites with the longest records since 1975 was mostly hay meadow and pasture. The land-use since 1975 at WS185 has been no-till corn through 1986, and hay meadow since then. The period of record of most interest for the present study is from about 1975 to the present. This is because the land-use has been relatively constant and water quality data began at about this same time.

Methods

The project calls for installation of increasing percentages (0% to 40%) of imperviousness on the experimental watersheds over time. However, challenges arise regarding the actual size, shape, materials, distribution of impervious elements on the watersheds, and representativeness of the impervious elements in terms of simulating of actual urban conditions, as well as the nature of land disturbance surrounding each impervious element. Initially, the urban condition to be approximated will be a residential neighborhood, without streets, and no house gutters. Because results from watershed studies depend upon the weather and the urbanization project will take at least five years to complete, rainfall simulation will be used to help guide the size and spacing of impervious elements to be placed on the watersheds.

Rainfall simulation

Warnemuende et al. (2003) describe the rainfall-simulator component of this research and present some initial results. The role of the simulator in the overall project is discussed below.

There are two primary spatial configurations that have been found to be important in urban watersheds - "connected" and "unconnected" impervious elements (Figure 2), also cited in the literature as "effective" and "ineffective" imperviousness. At the extremes, an element that is "connected" to the stream channel will shed water immediately to the channel. An "unconnected" element will shed water to pervious areas, allowing infiltration while traveling to the channel. The purpose of the initial rainfall simulations is to provide guidance on arrangement of impervious elements in the experimental watersheds (connected and unconnected).

The rainfall-simulator research activity addresses the impact of different levels and arrangements of impervious surface on runoff production and soil water dynamics at a spatial scale of the order of square meters. Experiments have initially been conducted using constant-intensity artificial rainfall applied to a 16-m^2 square soil bed (4 m x 4 m). This soil bed had a concave shape that simulates a concentrated-flow area in a watershed. The overland-flow and channel slopes in the soil bed were set to small slopes, and data were collected at the lower edge of the box. Runoff hydrograph data were collected and erosion samples obtained. Results suggested that travel times were too short and treatment differences were not detectable.

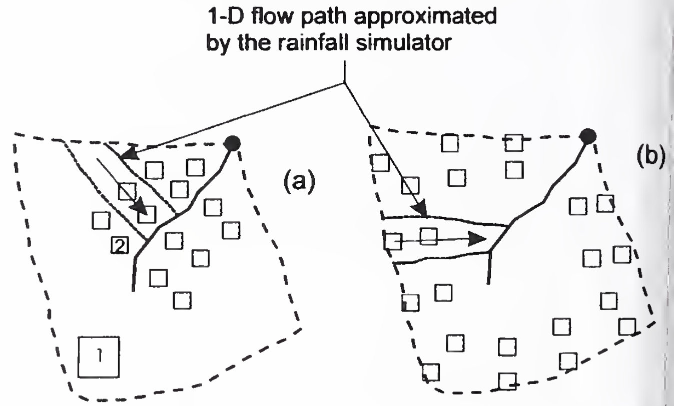

Figure 2. Spatial arrangement of impervious elements on the experimental watersheds for: (a) elements "connected" to the stream channel, and (b) elements "unconnected" to the stream channel.

As an alternative, a 1-dimensional approach approximates a 1-dimensional strip of watershed (Figure 2) by a uniformly sloped, rectangular surface on which impervious elements are installed and artificial rainfall applied. Runoff and sediment are collected at the bottom edge of the plot.

Preliminary plans call for the use of one rainfall simulation (~25 mm hr^{-1}), for zero and 100 percent impervious cover; and one run for each of two different arrangements of 5, 10, 25, and 40 percent impervious cover. The 100 percent impervious cover is a precipitation calibration run to confirm rainfall intensity and to develop corrections as necessary. All simulations are performed with a triple layer of screen between the simulator and soil surface to moderate rain-drop impact and to minimize erosion of the bare soil surface. Impervious elements will be installed and a rainfall-simulation run conducted for different imperviousness configurations.

The data will be analyzed to evaluate the effectiveness of unconnected impervious elements as a potential best-management practice (BMP) for practical landscaping and construction, for evaluating the utility of rainfall simulation for urbanization objectives, and for guiding the field watershed experiments of this project. Furthermore, the rainfall simulator will be used as a screening tool for a variety of BMPs such as vegetated areas, pavers, etc.

Watershed studies

Watershed studies require many years to conduct because of varying weather. In the present study, sufficient data must be collected for an adequately-sized data set for each level of imperviousness.

Impervious elements will be installed on four watersheds as described below, and the percent imperviousness will be increased through the years as sufficient data are collected. Initially, 5% imperviousness is planned. By the end of the multi-year project, total imperviousness will have been increased to 40%.

Using the four watersheds, there will be two treatments, based on preliminary rainfall simulator analysis (simulator tests are ongoing). One treatment will consist of impervious elements placed at the periphery of one of the 0.6- and 3-ha watersheds (unconnected; Figure 2b). The other treatment will consist of watershed impervious elements placed closer to the concentrated flow areas on the other 0.6- and 3-ha areas (connected; Figure 2a). On the smaller watersheds the watershed impervious elements will be smaller than realistic in order to attain 40% imperviousness. The larger watersheds will enable the watershed elements to be more realistic in size. The use of small and large watersheds will also allow a measure of scale of imperviousness and natural spatial variability in watershed response to precipitation to be addressed. Along with rainfall simulator trials, the size of the impervious elements will be determined by consulting a landscape architect, and by using digitized topographic maps and GIS to determine "reasonable" sizes for the impervious elements.

The 3-ha sites have not been consistently monitored over recent years. That is, one watershed has a recent record, while the other has not been measured for many years. This may require the installation of impervious elements later in the project. One approach to monitoring is to have the different percentage imperviousness treatment levels within the same year. This would allow the evaluation of the watershed treatments under a wider variety of precipitation inputs.

The aggregate of impervious elements will simulate a residential neighborhood. Each element will be made of landfill liner that will be in gable-roof form suspended about 1 m above the ground. Water will not be allowed to flow under the impervious element. To simulate disturbance due to excavation for foundations, a moldboard plow will be used to disturb the area surrounding each impervious element (a residential lot) and then disked. It has been shown that land disturbance increases runoff due to destruction of soil structure. Turf will be installed around each impervious element to simulate the turf that would be installed by a homeowner. Turf will be maintained by application of pesticides and nutrients, and by mowing. In each of the following years the percent imperviousness will be increased the same amount on the watersheds.

Infiltrometer measurements are planned using the truck-mounted NAEW infiltrometer. Infiltrometer measurements will be taken in areas scheduled for impervious-element installation. This will quantify both the potential loss of infiltration after the area becomes impervious and determine the extent of perviousness for sod or seeded lawns under different antecedent soil water conditions.

Soil-water potential and water-content measurements will be made across the watershed to quantify changes in soil-water regimes under varying precipitation events and weather conditions.

Methods of analysis

Data collected will be runoff hydrographs and a variety of water-quality constituents. Data collected under imperviousness will be compared with historic NAEW data. Baseline data will be analyzed to determine differences between the watersheds, and the degree to which they respond similarly (preliminary results in "Baseline Runoff Data" section).

Data analysis for each level of imperviousness will include an evaluation of curve number, runoff rate increase, runoff volume increase, soil moisture changes, water quality changes, etc. Nutrient, sediment, and major ion concentrations will be determined. If enough funding is available, pesticide concentrations will be determined as well.

The initial approach to analyzing the data will involve plotting a measure of percent imperviousness on the abscissa against a response variable on the ordinate (Figure 3). A response variable is the ratio of a variable to its pretreatment average or median value (or another representative pretreatment value). For example, the ratio of the average or median total runoff under a given percent imperviousness to the corresponding pretreatment value will be computed and plotted against percent imperviousness. Response variables to be initially investigated include peak runoff, total runoff volume, unit-hydrograph parameters, chemical constituent and sediment concentrations and loads, etc. In addition, duration curves of runoff, concentrations, and loads, and NRCS curve numbers will be compared among treatments. The method of analysis may be changed after preliminary exploratory analysis of the data.

Figure 3. Initial analysis of the urbanization data using a response variable.

The literature has many examples of the use of percent imperviousness as a measure of urbanization. This is because *available* data were analyzed. However, percent imperviousness in not necessarily the best measure of imperviousness to use. This is because of the infinite combinations of width of impervious element, spacing between them, and position with respect to the stream channel. In the present study, other measures of imperviousness can be investigated in both the rainfall simulator and watershed studies because of the controlled nature of the study.

Watershed models (e.g., TR-20/55; HSPF; SWMM) for urban areas will be reviewed and one or more selected for further study. Using data collected from project experiments and other data sources, the model(s) will be verified. Model components will be modified as needed to improve modeling. The advantages, disadvantages, and limitations of event-based and continuous simulation approaches using different models for a runoff-credit-trading system will be explored. This will include evaluating the importance of connectedness of impervious areas to the stream channel, use of design storms, use of weather and precipitation generators, and utility of flow-duration curves. It is also of practical interest that effectiveness and performance measures for BMPs be assessed.

Best Management Practices (BMPs)

BMPs for controlling water, sediment, and chemicals will be developed and evaluated. This includes evaluation of the spatial arrangement of impervious elements in a watershed, impervious pavers, vegetative control of water and chemicals, and other innovative BMPs that will be developed as data collection proceeds. A rainfall simulator will be used to conduct some of these studies, including indoor and outdoor simulators.

At the end of monitoring when all watersheds are at the maximum percent imperviousness, additional monitoring will allow for evaluation of BMPs and investigation of other factors involved in urbanization. Examples include installing a road with runoff ditches in the watersheds; gutters to cause flow to bypass pervious areas for the unconnected treatment, forcing flow directly to the stream channel; use of porous pavers; and installing "green roofs" on the impervious elements to store water for later slow release.

Baseline Runoff Data

Baseline annual runoff data were examined on three of the experimental watersheds for consistency and for potential changing hydrologic conditions. The data for WS106, WS121, and WS192 since 1975 (the start of constant land-use) were plotted against one another (Figures 4 and 5). The data show generally good agreement between WS106 and WS121, and WS106 and WS192. The annual runoff comparison between WS121 and WS192 agree well also (not shown) as inferred from Figures 4 and 5. Average annual runoff for the available record from 1975 through 2002 for WS106 is 4.96 cm, WS121 is 5.83 cm, and WS192 is 6.42 cm (through 1994). The corresponding average annual precipitation from RG113 is 95.15 cm. On an annual basis, 5.2% of the precipitation runs off at WS106, 6.1% at WS121, and 10.5% at WS 192. It is apparent that runoff occurs only during heavy snowmelt and rainfall events. Differences in runoff due to urbanization should be apparent as imperviousness increases during the experiment.

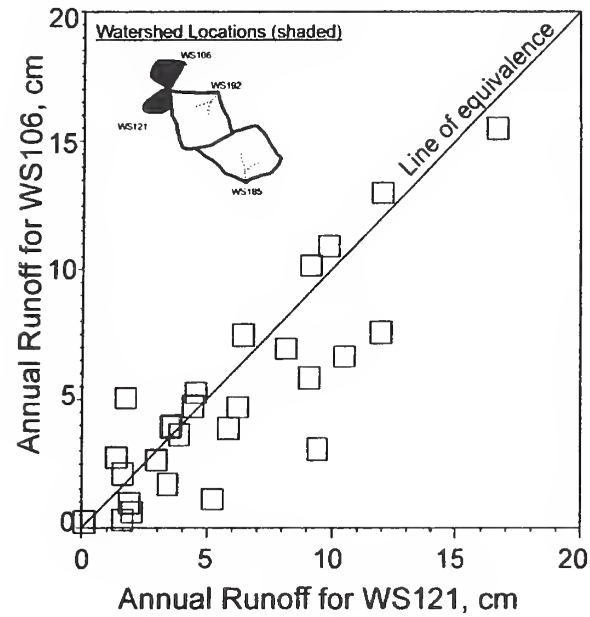

Figure 4. Annual runoff comparison between WS106 and WS121.

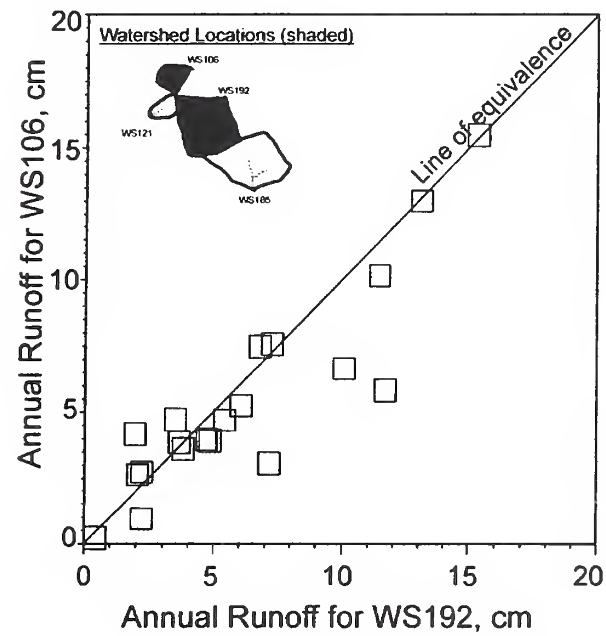

Figure 5. Annual runoff comparison between WS106 and WS192.

The plot of monthly total runoff at WS106 and WS121 (Figure 6) is typical of similar comparisons between the other watersheds (graphs not shown). Monthly runoff shows more scatter than the similar annual-runoff comparisons (Figure 4). Differences are attributed to different slope aspects among these two watersheds (Figure 1); differences in monthly snow accumulation and subsequent delayed snowmelt runoff; and physical characteristics affecting runoff generation in the watersheds though particularly for small storm events.

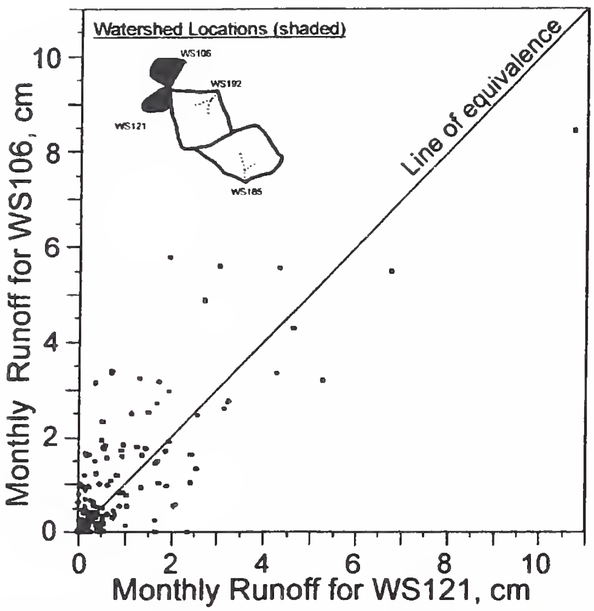

Figure 6. Monthly runoff comparison between WS106 and WS121.

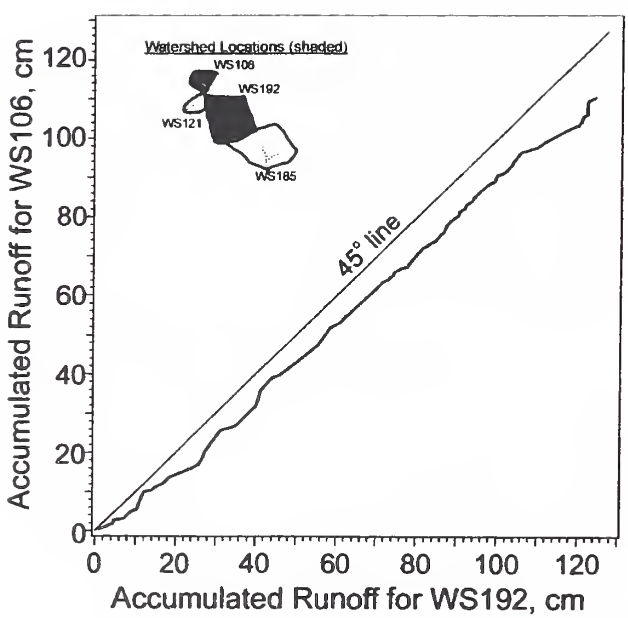

Figure 7. Double-mass plot of monthly runoff between WS106 and WS192.

The double-mass curve of monthly runoff for WS106 and WS192 (Figure 7) since 1975 show a general constant slope (nearly a 45° line). This also confirms the good agreement between annual totals at these two sites in Figure 5. This is because the accumulated annual totals will plot on the monthly double-mass-curve graph. and nearly a 45 ° line (Figure 8).

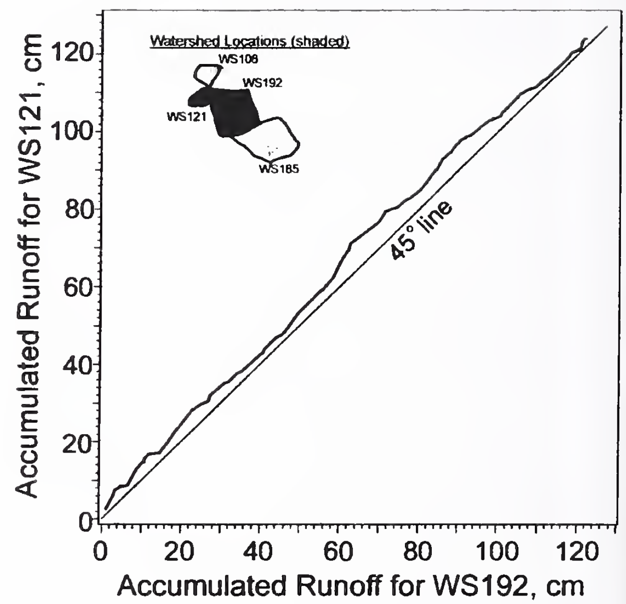

Figure 8. Double-mass plot of monthly runoff between WS121 and WS192.

The double-mass plot between WS 121 and WS192 shows similar constant slope. This graph also shows that annual totals agree well. A similar plot for WS106 and WS121 shows a similar linear trend. The constant slopes at all sites show that there are no major changes that have occurred during this period of record to affect hydrology.

Summary

This paper outlines the approach and rationale for a project investigating the effects of increasing urbanization (0% to 40%) on runoff, sediment, and water quality by using artificial rainfall simulation, modeling, and experimental watersheds. The project was jointly initiated by the U.S. EPA and the ARS at the North

Appalachian Experimental Watershed at Coshocton, OH. Rainfall simulation initially will help guide placement of impervious elements on the watersheds as percentage imperviousness increases. Two spatial arrangements of imperviousness will be explored - impervious elements connected to the stream channel and elements unconnected, wherein water flows over pervious areas en route to stream channels. The Coshocton watershed study will increase general understanding of runoff-production processes under urbanizing conditions, and will document the effects of land disturbances due to urbanization on hydrology and water quality under natural precipitation and weather. Modeling efforts will aid in the extrapolation of experimental watershed results to ungaged areas. Rainfall simulation and experimental watershed studies will also be used to investigate potential best-management practices.

During this project, urbanization on the experimental watersheds will be approximated by installing impervious surfaces on the surface of the watersheds. A small area surrounding each impervious element will be disturbed to simulate excavations during construction. This will simulate an increase in runoff production on pervious areas as would occur during construction in a residential development. A lawn will be installed and maintained on the disturbed area and will receive nutrient and pesticide inputs.

The experimental watersheds have long records (since 1975) of runoff, precipitation, and water quality under constant land-use, providing baseline conditions with which to compare urbanizing effects. Preliminary exploration of the baseline runoff data revealed that the three watersheds, for which there are runoff records, behaved similarly. Annual runoff totals agree well, and no apparent changes are occurring in runoff regime during the baseline period.

The data and results of this project will be used for improved modeling, BMP development and evaluation, and as data bases for a national tradable runoff credits program. Users of results from this project include U.S. EPA, ARS, and university scientists, environmental regulators, engineers, hydrologists, landscape architects, and governmental officials involved in establishing guidelines for minimizing the impacts of urbanization.

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Water Supply and the U.S. Army Corps of Engineers

James J. Comiskey

Abstract

The purpose of this paper is to present the role of the Corps of Engineers in supplying water to urban and agricultural areas of the United States. The following aspects will be discussed: (1) overview of water supply storage in Corps reservoirs; (2) increasing demand for municipal and industrial water in the U.S.; (3) authorities available to Corps for reallocation of existing storage for additional water supply purposes; (4) Corps assistance during droughts; (5) water supply concerns and the environment, and (6) security issues in managing Corps reservoirs.

Keywords: water supply, Corps reservoir, reallocation

Introduction

Water is important for the well-being of both humans and ecological systems and in many of our recreational and economic activities. Neither man nor plant nor animals could survive without water and water is employed in virtually everything that we do. It is use in industry, used to produce electricity (hydroelectric power); it is very important in both our inland and maritime transportation industry; it is used in the disposal of our waste products. This precious resource also provides cultural and amenity values. Both our way and quality of life depends, to a great extent, on an adequate supply of fresh water.

Our planet earth is blessed with and abundance of this essential liquid, water. It covers about two-thirds of the surface of our planet. On the other hand, the actual supply of fresh water is fairly small. Nearly 97 percent of the water is found in the oceans. Of the remaining 3 percent, about 2 percent is contained in ice caps in the Arctic and Antarctic regions of the world. Much of the rest is underground, temporarily stored in the atmosphere or of such poor quality that it cannot be effectively utilized. Thus, the amount of water that we can actually use from rivers, lakes, streams and underground sources amounts only approximately 0.3 percent of the world's supply.

On the whole, the United States has an ample water supply. While about 1400 billion gallons per day are available, we actually only withdraw about 380 billion gallons on an average day. These numbers to some extent do not reveal the entire story about water shortages in the United States. There are seasonal variations in supply and some parts of the country are mostly dry (the West) while the East has a more dependable water supply.

Role of the U.S. Army Corps of Engineers in Managing Our Nation's Water Supply

The U.S. Army Corps of Engineers, as one of the Nation's largest water management agencies, has been given a major role by the U.S. Congress in assuring our supply of water. Water under Corps management is utilized for our homes, our industries, recreation, agricultural production, generation of electricity,

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flow augmentation for enhanced fish habitat and for water quality.

Although municipalities, in general, have been given the responsibility for municipal and industrial water supplies, the Corps of Engineers has developed and maintained distribution systems for the Washington, DC area and some surrounding cities since the 1860s. The Corps through its reservoirs continues to supply water to municipalities and industries.

With respect to Corps of Engineers facilities that contain water supply storage, this agency, through 235 agreements operates 117 such reservoir projects in 24 states and Puerto Rico having about 9.5 million acre-feet of storage for water supply along with other project purposes such as flood damage reduction, navigation, recreation etc. In total the Corps has constructed over 600 reservoirs with 200 million acre-feet of storage. (An acre-foot is equivalent to one area acre, covered by one foot of water that equals 43,560 cubic feet or 325,851 gallons). It is estimated that if all the water currently stored in Corps reservoirs for water supply needs were utilized, about 85 million Americans could be served for a year. This amounts to about 3 trillion gallons of water for use by communities and businesses.

Agricultural Needs for Water

About 80 percent of the water utilized in the United States is for agricultural purposes, including the irrigation of crops and the feeding of livestock. The U.S. Army Corps of Engineers is a key player in providing irrigation water to farmers, primarily in the Western states along with the Bureau of Reclamation. Farmers in these states are responsible for the cultivation of approximately 10 million acres of land, resulting in the production. Presently, about 57 million acre-feet of storage has been included in 50 Corps projects for irrigation and other joint use purposes. Most of this water storage (84 percent) is located in Corps reservoirs in the states of Montana, North Dakota, and South Dakota, in the Missouri River Basin.

Reallocation Authority

The Corps of Engineers can reallocate existing reservoir storage for water supply and to include such uses along with other project purposes. This type of authority has great potential to supply additional sources of water due to the fact that few new projects are being built. Reallocation of water storage includes:

- Dedication of space in a reservoir, presently not being utilized.
- Transfer of space from some existing uses to water supply purposes.
- Physical modification of dam (raising its height etc.) to increase the storage capacity.
- Changes in reservoir operating patterns to better optimize project benefits.

About 50 such allocations have been completed at Corps facilities, resulting in the reallocation of over 400,000 acre-feet of storage.

Corps Assistance during Droughts

A drought may be defined as a time of below normal precipitation, resulting in water supply shortages. Such droughts may result from normal variations in the weather or possibly be a by-product of global warming, El Niño, etc. In times of drought, the Corps of Engineers may assist in the reduction in some of the effects of these water-scarce periods. This assistance may take on several forms:

- Contingency water--The Corps can coordinate with state or municipalities to temporarily withdraw water from existing Corps projects to augment normal needs. These efforts may also involve meetings or other coordination sessions with water agencies to maximize regional or local water needs.
- Planning assistance to states--States can receive Corps water resources planning expertise on cost-shared studies.
- Emergency supplies of water--Water can be provided from Corps reservoirs to a municipality if water is sub-quality or

contaminated and may pose a public health threat.

- Construction of emergency wells and water transportation--The Corps can construct wells under its emergency power authorities or transport water after a determination is made that farms and/or communities etc. are likely to be impacted by a drought.

Corps of Engineers Water Management and the Environment

The Corps performs many studies involving water supplies every year to protect and restore the environment. Many of these studies and projects involve improving fish and wildlife habitats by reallocating supplies or revising reservoir operating procedures.

On rivers or streams that may have periodic low flow conditions, the Corps can temporarily store water in its reservoirs and periodically release it to increase flow, and thus improve habitats and enhance overall water quality in the stream. The Corps has currently 89 environmental storage projects underway to supplement low stream flows.

A number of other studies are presently being conducted by the Corps whose purpose is to improve environmental conditions for selected fish species. An example of this is a water supply analysis being done on the Snake River in Idaho. The goal of the study is to identify actions to improve Chinook and sockeye salmon runs.

Environmental considerations are a top priority in Corps water supply projects. The Corps carefully evaluates methods to minimize negative environmental impacts and to enhance the environmental benefits of each project.

Population Growth Equal Greater Water Demand

The population of many urban areas in the United States is increasing at a rate that is compelling these areas to develop new sources of water to meet growing needs. Even though water today is being more efficiently managed, our population growth is exceeding these efficiencies. Many of these growing regions are located in the Mountain and Southwest states. The percent of growth in some of these areas during the past ten years demonstrates the need to increase availability of fresh water supplies:

- Nevada, 66 percent
- Arizona, 40 percent
- Colorado, 31 percent
- Utah, 30 percent
- Idaho, 29 percent

In the Western states, the development of water resources for irrigation has often been the highest water priority. However the total withdrawal of water for all purposes in the U.S. has declined principally due to reductions in water usage for irrigation and mining. In a few areas, this allows the Corps or other water agencies to reallocate water previously reserved for irrigation to municipal and industrial water supply.

Our Aging Water Infrastructure

Federal, state and municipal water supply reservoirs, diversion structures, pipelines, wells, water treatment plants distribution lines are called collectively our water infrastructure. In several areas of the U.S., major parts of this infrastructure are from 50 to 100 years old or older and should be replaced or upgraded. Such rehabilitation on a massive scale offers several challenges and can be quite expensive. However, the rehabilitation of these aging structures can present several opportunities, including the following:

- Conservation of water supplies by better transport systems and water treatment plants.
- Use of a variety of incentives to conserve water.
- Use of economies of scale through combining numerous small water systems into a larger regional facility.

- Upgraded security of water supply systems against chemical and biological threats.

The Corps of Engineers may do water storage reallocation studies that sometimes includes the planning of the design level and estimates of cost and associated expenses to other portions of the distribution system. Reallocation is very relevant if a potential exists to combine municipal or regional approaches.

Economic Benefits of Corps Water Supply

Economic benefits associated with Corps water supplies have been estimated to include the following:

- About $770 million in annual benefits attributable to 3 trillion gallons of water stored in 117 Corps projects.
- At 750 gallons per capita per day, these 3 trillion gallons would satisfy water requirements of every American for approximately 17 days.
- The Corps supplies 18.5 trillion gallons of water for agricultural purposes and other joint use water needs.

Security Issues

The September 2001 terrorist attacks generated concern at all levels of water resources management about the security and safety of our water supply facilities. Perhaps on a more positive note, it should be borne in mind that our country's water supply is safe and not as nearly vulnerable to a terrorist attack or contamination as it first may appear.

This is true in part because security has been a high priority in the design, management and operation of our water supply systems. Most of these systems are subject to stringent testing and monitoring. Secondarily, the quantity of water treated at most municipal facilities is so large and that it would potentially take many tanker loads of chemical and biological contaminants to circumvent the powerful dilution and treatment processes employed in the purification of water. In addition, physical barriers such as fences, monitoring systems and frequent water quality testing make it highly unlikely that contaminants could be added to the system without immediate detection.

Conclusions

As one of the Nation's largest water management agency, the U.S. Army Corps of engineers has been given a major role by Congress in assuring adequate supplies of municipal, industrial and agricultural water supplies from its reservoirs. In addition, water from these same facilities is used for recreation, generation of electricity, enhancement of water quality and instream flows for fish and wildlife needs. The Corps can reallocate existing storage to augment water supply uses. Additionally, in times of drought, this same agency may assist states and municipalities by providing emergency water and other forms of drought assistance.

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Spatiotemporal Assessment of Long-Term Stream Water Chemistry Across Louisiana

Y. Jun Xu

Abstract

Louisiana uses a set of 475 state-defined sub-segment watersheds as the spatial framework for its watershed assessment. The main goal of this assessment program is to develop Total Maximum Daily Loads (TMDLs) to meet water quality standards that designate the beneficial uses of surface waters. However, continuous long-term water quality data are lacking for most of the sub-segment watersheds. This study employed most recent and long-term data (1978-2001) selected from 25 monitoring stations to identify spatial and temporal patterns in water quality changes across the state's five major landforms from the coastal march to the coastal plain, Mississippi alluvial, upland terrace, and upland hills. Temporal trends of stream water quality including dissolved oxygen (DO), nutrients (total N, P, OC), total suspended solid (TSS), fecal coliform (FC), and toxic elements (copper, lead, and arsenic) were analyzed. The study shows that historically the streams in the coastal march and coastal plain regions contained higher nutrient loads and lower DO than those in the upland regions. The DO levels and nutrient concentrations remained relatively consistent over the past 24 years, whereas the concentrations of the toxic elements in the streams largely decreased during the late 1980s and early 1990s. Monthly TSS varied largely in all 25 streams and across seasons. These results indicate that the development of water quality standards needs to be site specific, and that more intensive, storm-based sampling is necessary to adequately characterize TSS in Louisiana's watersheds.

Keywords: stream water quality, spatiotemporal assessment, watersheds, Louisiana lowland

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Seasonal Variations in River Flow and Nutrient Concentrations in a Northwestern USA Watershed

Anne Sigleo, Walter Frick

Abstract

Dissolved nutrient concentrations were measured in the Yaquina River, Oregon from 1998 through 2001 to determine nutrient loading from the watershed as part of a larger agency program for evaluating nutrient sources. The effects of storms on dissolved nutrient transport were investigated relative to stream discharge for three storm events, including one in a high rainfall-discharge year, and two in average years, one of which followed a drought year. During the drought year (no flows >25 m^3s^{-1}), total dissolved nitrate input was considerably less than in wetter years. However, dissolved nitrate concentrations were unusually high in the first winter storm runoff after the drought. In the November 2001 storm, dissolved nitrate increased rapidly (to nearly 200 □M) but decreased by 20 to 30 percent as the storm progressed. The dissolved nitrate nitrogen loads varied from 17,400 kg day^{-1} during high-flow storm events to less than 2.25 kg day^{-1} during late summer, low flow conditions. Dissolved silica dynamics were quite different and during storm events silica concentrations in the Yaquina River decreased to near zero at the storm height, probably due to dilution by rapid, shallow flow, and then recovered after 48 hours. During the time interval studied, over 94% of the dissolved nitrate and silica were transported during the winter months of greater rainfall indicating that seasonality and river flow are important determinants when considering nutrient loadings.

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Keywords: nitrogen, silica, orthophosphate, river flow, Oregon

Introduction

The capability of rivers to export nutrients is controlled by water discharge, which in turn is a function of climate, topographic relief, water retention properties of the soils, and geologic structure of the basin. The hydrologic cycle controls the timing and volume of freshwater delivery to coastal ecosystems. Hydrographic input to Pacific Northwest rivers is related to seasonal variations in precipitation with mild wet winters and cool dry summers (Peterson et al. 1984). To estimate the effects of storms on nutrient mobility, the dissolved nutrients and suspended sediment loads were measured and compared with stream discharge in the Yaquina River, OR. This study comprises the watershed portion of a larger nutrient budget project for Yaquina estuary.

The specific objectives of this work were to determine the watershed input of dissolved nitrate, ammonium, phosphate, and silica, along with suspended and resuspended sediments in the Yaquina River. We were specifically interested in the variation of nutrient fluxes with respect to inter-annual variations in precipitation, along with the variation of dissolved nitrate, ammonium, phosphate and silica fluxes with water discharge. We were further interested in distinguishing between seasonal effects as compared to the effects due to variations in annual discharge.

Methods

Yaquina River and watershed

The Yaquina watershed (Figure 1) rises from sea level at Newport, Oregon to an elevation of 1249 m. The watershed has a surface area of 655 km^2 and an average elevation of 166 m. The primary land use is silviculture. The forests are dominated by conifers, although disturbed sites are frequently occupied by pioneer broad-leaved trees (Ohmann and Gregory 2002). Broad-leaved trees, predominantly red alder and broad-leaf maple, also occur in riparian areas along streams. The Yaquina River flow at Chitwood gage averages 6.2 $m^3\ sec^{-1}$, although it can vary from 0.28 $m^3 sec^{-1}$ during late summer low flow conditions to >78 $m^3\ sec^{-1}$ during storm events. Typically discharge from rainfall is high in late fall and winter (November to March) and low to moderate the remainder of the year. The Yaquina River flows through the Tyee, Yamhill, Yaquina and Alsea Formations, as well as Quaternary alluvial deposits. The rocks of these formations contain massive to thinly bedded tuffaceous siltstones, sandstones, basalt breccias, and augite-rich tuff that are sources for silica and suspended sediment (Snavely and Wagner 1963). The soils generally are well drained with poorly developed horizons (Ohmann and Gregory 2002).

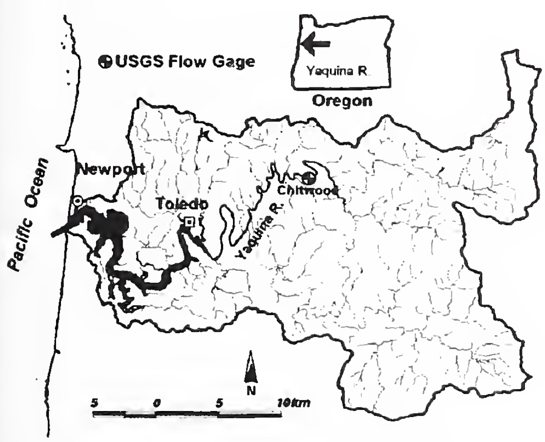

Figure 9. The Yaquina River watershed.

Sampling Methods

Water samples were collected weekly at USGS stream gage 14306030 (presently operated by Oregon Water Resources Department) near Chitwood, OR (Lat 44 39 29 N, Long 123 50 15 S) from September 1998 through December 2001. There were no high flows >25 m^3s^{-1} in the winter of 2000-2001 during a drought. In 2001, the first flood event of the fall-winter season was intensely sampled November 21-28. For suspended sediment determinations water samples (250 to 100 ml) were filtered through precombusted, preweighed glass fiber filters. Duplicate samples were analyzed gravimetrically for suspended sediment concentration and the results mathematically averaged. Filtered (Whatman 25 mm GF/F filters in a Gelman nylon filter holder) water samples (20 ml) were collected for dissolved nutrient analysis and frozen within an hour. Dissolved nutrients were analyzed by a contract laboratory (MSI Analytical Laboratory, U.C. Santa Barbara, CA) using a Latchet Autosampler for simultaneous determination of nitrite, nitrate + nitrite, ammonium, phosphate and silicate. Because nitrite comprised less than 1% of the nitrogen species at this site, it is not discussed further, and nitrate + nitrite will be treated as nitrate.

Specific conductance and temperature were measured with a YSI 30 conductivity temperature probe suspended in the water at the time the samples were collected. Dissolved organic carbon (DOC) was analyzed with an OI Model 700 TOC analyzer. Chlorophyll *a* was measured on a Turner 10-AU fluorometer.

Nutrient load calculations

The nutrient data were combined with hydrographic data to calculate suspended sediment and nutrient loadings from the surrounding watershed. Nutrient loads are a product of a nutrient concentration and the mass water flow. To calculate the amount of nutrient transported by the Yaquina River in a specified time period the following formula was used:

$$M = \sum_{i=\text{begin hour}}^{i=\text{end hour}} [\mathrm{x}]_i\, f_i\, \Delta t \qquad (1)$$

where M is the total mass of nutrient or suspended sediment transported at Chitwood in some specified time period beginning at begin hour and ending at end hour, the term X in brackets is the nutrient or suspended sediment concentration at hour i, f_i is the corresponding river flow rate, and □t is the time interval (e.g., 3600 s) under consideration.

Results

River discharge during flood events of WY99 (water year October 1, 1998 to September 31, 1999) and WY02 (water year October 1, 2001 to September 31, 2002) peaked sharply followed by a gradual recession for seven to 10 days (Figure 2). The December 1998 and November 1999 storms were the largest storms for their respective water years. The second major storm of WY00 (December 1999) followed a major 50-year storm by approximately three weeks. In addition, this

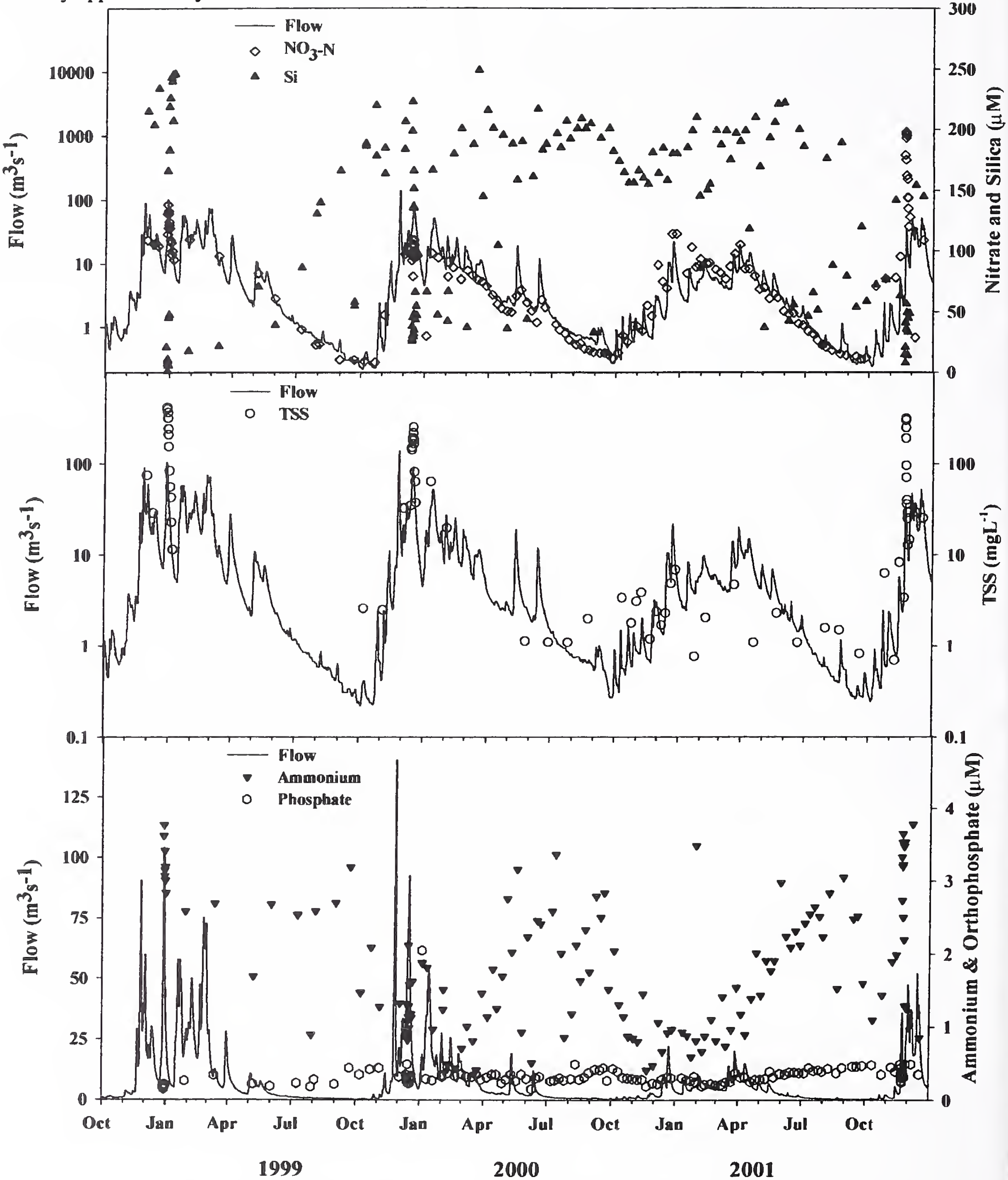

Figure 10. Nitrate, silica, TSS, ammonium, orthophosphate, and streamflow as functions of time, October 1999 – December 2001. Nitrate and TSS, on linear scales, may be compared to streamflow plotted logarithmically (top and middle panels) to emphasize potential functional relationships. In the bottom panel streamflow is plotted on a linear scale to illustrate the range in water flow.

storm contained two episodes of peak flow, the second being slightly larger than the first. There were no flood events in WY01.

Total suspended solids (TSS) ranged from 425 mg l^{-1} during the December 1998 storm to less than 1 mg l^{-1} during low flow summer conditions (Figure 2). The water temperature varied from 7.5 C in December to 20.3 C in July and August. Specific conductance varied from 69 to 114 □S, a range indicative of freshwater.

In the Yaquina River, dissolved silica concentrations increased from 120 □M (pre-storm) to over 270 □M in freshwater portions of the river during the rising hydrograph, decreased to near zero at the storm height and then recovered (Figure 2). During the late summers of 1999 and 2001, silica decreased by a factor of 4 suggesting biological utilization of that nutrient. During the summer of 2000, there was no apparent decrease in silica concentrations suggesting a surplus of dissolved silica relative to nitrate.

Overall the concentration of nitrate nitrogen varied by a factor of 20, with the lowest values coinciding with low river flow during September and early October (Figure 2). Dissolved nitrate increased rapidly during high flow events and then decreased by 20 to 30 percent as the storm progressed. During the drought year WY2001, total dissolved nitrate input was considerably less than in wetter years. Dissolved nitrate concentrations were unusually high (up to 195 □M) in the first winter storm runoff after the drought. After winter high flow events, nitrate decreased smoothly indicating biological utilization until the concentrations were reduced to 10□M in late summer and early October.

Ammonium concentrations varied from 0.8 to 3.8 □M with the higher values occurring during winter storm events and late summer when heterotrophic activity is at its maximum. Phosphate ranged from 0.16 to 0.49 □M with the higher values tending to occur during winter, although high values also occurred in August and September of 2000 and 2001. The low phosphate variation suggests that phosphate is tightly cycled within the system (Molinero and Burke 2003). Dissolved organic carbon (DOC) values were generally between 1.5 and 3 mg l^{-1} with the highest values occurring in September and October during the fall phytoplankton bloom. Chlorophyll *a* varied from below the detection limit during December and January to over 6 □g l^{-1} in August.

Discussion

Silica

Dissolved silica concentrations were greatly decreased during storm events (Figure 3). Examination of Figure 3 indicates that the silica concentration decreased with the first rainfall, and decreased further still with the first storm of WY 02. When the storm effect subsided, silica increased to pre-storm concentrations. The apparent decline in silica concentrations is caused by dilution from rapid, shallow flow (Kennedy 1971). Generally rapid flow is minimal in forested systems. However, if surface soils are saturated, during intense rainfall near surface flow will become important. It is the rapid, shallow flow that is thought to dilute the older, base flow concentrations of silica (Kennedy 1971). This dilution effect at maximum flood also is observed in the California, Sacramento-San Joaquin system (Smith et al. 1985, Schemel and Hager 1986). Careful inspection of Figure 2 suggests that there was more silica draw down during the late summer after the drought.

Nitrate

Nitrate is highly soluble relative to silica, for example. As a result, the concentration of nitrate simply increases as water flow increases, and there is no apparent dilution (Hill et al. 1999). The source of nitrate appears to be from the surrounding watershed as suggested by the low conductivity water (average < 70 □S). Within the watershed, over 20% of the regional vegetation is classified as broad-leaf hardwood, and of that 20%, 90% consists of alder (Ohmann and Gregory 2002). Alder (*Alnus* spp.) forms a symbiotic relationship with the actinomycete *Frankia* spp. and together they are able to fix nitrogen from the atmosphere (Vogel and Gower 1998). In comparative studies of conifer forests with and without alder, the C and N content of the over story biomass, total vegetation, and forest soil were greater in conifer forests with alder than those without alder (Vogel and Gower 1998). In other words, the added nitrate from the presence of alder increased the overall productivity of the system. In addition, ^{15}N natural abundance studies further supported the premise that alder had increased the nitrogen budget by symbiotic diazotrophic nitrogen fixation (Vogel and Gower 1998). Other scientists also reported greater litter fall

N inputs in conifer stands with alder relative to those without alder (Binkley et al. 1992). Because there is a substantial amount of alder in the Yaquina watershed (Ohmann and Gregory 2002), it seems reasonable that decaying alder leaf litter provides a source of nitrate in this forested watershed.

Nutrient load calculations indicate that the annual nitrate load is directly related to river flow. Thus in high rainfall years, more nitrate is transported into the river from the watershed than in low rainfall years. During dry years, nitrate tends to build up in soils, largely as a result of reduced plant uptake, and is washed into streams at larger rates during subsequent wet years (Goolsby et al. 1999). This relationship is clearly seen for the WY02 flood in November 2001 (Figures 2 and 3).

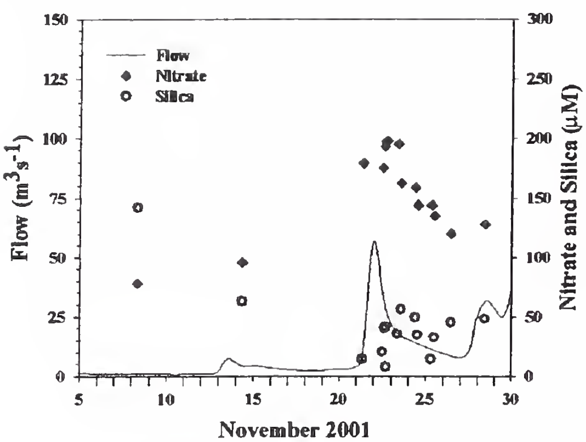

Figure 11. November 2001 storm nitrate and silica.

An empirical nitrate model

The semi-logarithmic relationship suggested between nitrate concentrations and stream flow shown in Figure 2 may be modeled by the following equation:

$$\text{nitrate} = 19.79 \ln(\text{stream flow}) + 35 \quad (2)$$

A comparison of the nitrate model predictions and observed nitrate values is shown in Figure 4. The bulk of the three-year samples are represented by open triangles, labeled "Normal," whereas the solid triangles correspond to the transition period ending the drought year 2001, beginning with the first significant, but comparatively minor, fall storm on October 11th and followed by the major storm beginning on November 21st and peaking two days later. The cluster of three points near 75 □M observed corresponds to data collected on October 11th and 25th and November 8th. The thirteen highest observed nitrate values in this group correspond to storm samples taken between November 21st and 28th. The highest outlying value above the line of agreement (106.7 □M) corresponds to the first sampling event, December 6th, after the storm. The point immediately below it is an unrelated outlying point corresponding to the sample taken on January 4, 2000. The r-square value for all points is 0.65 whereas the r-square value excluding the events represented by the solid triangles is 0.86.

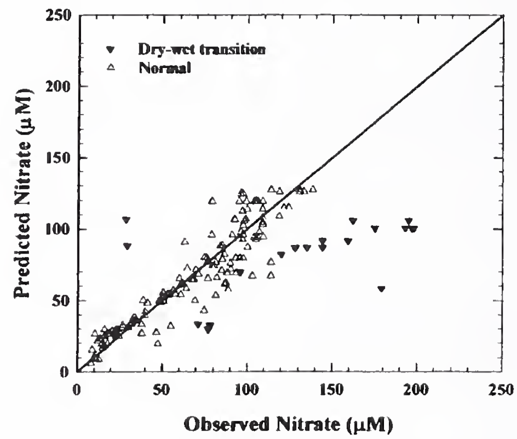

Figure 12. Nitrate model: Predicted versus observed concentrations. Conversion factor: 72 μM nitrate N equals 1 mg l^{-1} nitrate nitrogen.

It is clear from Figure 4 that the highest nitrate values are significantly underpredicted. Manipulation of the logarithmic model expressed by the equation above shows that, if the model were accurate at all times, the highest nitrate value would correspond to a stream flow of almost 3000 m^3s^{-1}, more than an order of magnitude greater than the actual maximum daily stream flow at the time. However these instances are associated with highly transient conditions as indicated by Figure 2. In other words, they are associated with rapid changes in the hydrograph and almost hourly nitrate samples.

Based on the November 2001 storm, it seems clear that long dry spells lead to a buildup of nitrate in soils that contributes to subsequent high nitrate concentrations in the Yaquina River associated with the first significant storms. The model appears to apply to conditions not affected by major transitions from dry to wet weather.

According to the empirical nitrate model, the dissolved nitrate nitrogen loads varied from 22586 kg day^{-1} during a high-flow storm event (11/26/99) to 1.34 kg day^{-1} during late summer. For comparison, the computed maximum and minimum based on high (12/28/99) and low flows and corresponding discrete nitrate samples yield approximately 17440 and 2.25 kg day^{-1} respectively. The daily maximum for the drought was 2588 kg day^{-1}, almost an order of magnitude smaller than that of the previous year (22586 kg day^{-1}). The annual dissolved nitrogen loads varied from 480 t yr^{-1} to 112 t yr^{-1} during the drought year. For the period of November 1, 1999 to April 30, 2000, the nitrate-nitrogen load was 459 t, whereas for the following summer-fall period of May 1 to October 31, 2000, 23.6 t is predicted to have been carried down the Yaquina River. The results indicate that for the time interval studied, about 94% of the annual dissolved nitrate was transported during the winter months of higher rainfall during the wet years compared to about 84.5% during the winter months of the drought year.

Conclusion

The results indicate that winters of higher rainfall will increase the annual nutrient load to the river relative to low rainfall winters. Silica is carried downstream from the upland watershed. During periods of high river discharge, the dissolved silica load tends to increase until "flood stage." During highest flows, dilution from rapid, near-surface flow becomes important. Ammonium and phosphorus concentrations increase only slightly during storm events indicating that the total load is tied directly to river discharge. Increased dissolved nitrate concentrations during storm events supports the hypothesis that the majority of nitrate-N may enter the estuary during storm pulses. A simple model relating nitrate concentration to the natural logarithm of stream flow predicts reasonably well, attaining an r-square value of 0.86 for events not preceded by drought.

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Modulating Storm Drain Flows to Reduce Stream Pollutant Concentrations

Walter Frick, Debra Denton

Abstract

Pathogen and toxic chemical concentrations above the chemical and toxicity water quality standards in creeks and rivers pose risks to human health and aquatic ecosystems. Storm drains discharging into these watercourses often contribute significantly to elevating pollutant concentrations during wet weather, especially following extended periods of dry weather over which pollutants accumulate, or after seasonal pesticides applications that cause high concentrations in retention structures and flood control basins drained by the storm drains. In many instances the discharges from the storm drains are controlled by pumps that run intermittently in response to water level elevations in the retention basins. These pumps usually run at full volume, modulated only in stepwise fashion when more than one pump serves the overflow structure. The on-off mode of operation is insensitive to conditions in the ambient flow or the effluent. Modulating storm drain flows can ameliorate the impact of pathogens or toxic residues found in the storm drain effluent by controlled and optimum mixing of the effluent and ambient streams. Plume models simulating the mixing process in real time based on continuously measured stream levels and storm drain volumes, together with variable flow pumps, could be used to blend the effluent with the receiving stream in a way that mitigates the impact of the storm drain on the environment. The Visual Plumes model is used on a storm drain discharging to urban Arcade Creek in Sacramento, California to demonstrate the potential benefits that may be realized by implementing this strategy.

Keywords: storm drain, diazinon, Visual Plumes, pollutant concentration, flow rate

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Introduction

The development of Total Maximum Daily Loads (TMDLs) for a particular water body depend on the location of point sources, available dilution, water quality standards, non-point source contributions, background conditions, and in-stream pollutant reactions and pollutant toxicity (USEPA 1991). To establish a TMDL, both the upstream and downstream boundaries of the water body and the critical flow conditions must be defined. Critical conditions are stream flow, pollutant loading, and water quality conditions that result in no acute or chronic concentrations exceeding the chemical-specific water quality standard or the toxicity water quality standard.

Thus development of TMDLs implies that the effects of merging streams, including storm drains bearing their respective pollutant loads, can be adequately characterized. It goes without saying that pollutant concentrations undergo changes at stream confluences, adjusting in conformance with mass balance that, given sufficient fetch, approaches a new fully mixed state defined by

$$C_f = \frac{C_a Q_a + C_{SD} Q_{SD}}{Q_a + Q_{SD}} \qquad (1)$$

where C is concentration, Q is flow, and the subscripts a, f, and SD refer to the ambient receiving stream upstream of the confluence, the confluent stream, and the storm drain respectively.

The storm drain could be a combined sewer overflow, or CSO, an "event during which excess combined sewage flow caused by inflow is discharged from a combined sewer, rather than conveyed to the sewage treatment plant because either the capacity of the treatment plant or the combined sewer is exceeded" (Washington State Department of Ecology 1994). The content of the water and intermediate storage, if any, remain undefined. CSOs imply loss of control, and, therefore, this work addresses a broader range of outfalls, including storm drains.

Addressed herein are structures found in urbanized areas that discharge accumulated wastewater from storm water retention basins to streams or channels by means of a pump intermittently pumping at a steady rate, or several pumps operating in stepwise fashion.

The model storm drain described herein discharges to a small watershed (88 km^2) in Sacramento, California. Arcade Creek is an urban perennial stream with a mean dry weather flow of 0.50 cfs (0.014 m^3s^{-1}) with storm event flows reaching 500 – 700 cfs (14.2 – 19.8 m^3s^{-1}) within a few hours following rain because of limited opportunities for watershed infiltration. The watershed drainage area comprises 79% urban, 19% agricultural, and 2% barren lands according to the land-use classification of Anderson et al. (1976). Approximately ten miles of Arcade Creek are 303(d) listed for chlorpyrifos, two organophoshates, and diazinon, an organophosphorus insecticide.

During wet weather events there are times when Arcade Creek is flowing with pollutant concentrations that do not exceed the diazinon water quality criteria upstream of the storm drain discharge but do exceed the criteria downstream of it. In such cases the mass balance is such that the higher concentration of the storm drain effluent produces plumes of elevated concentrations in the receiving stream. If the storm drain water exceeds a threshold concentration, the receiving stream will not have sufficient water to dilute the plume below the criterion concentration even after full mixing. In such an event the stream will exceed criteria concentrations between that point and the next confluence, and even farther, unless decay, absorption or other process or strategy reduces concentrations below criterion level.

One strategy to meet water quality criteria in the receiving stream would be to moderate the flow rate serving the retention basin and storm drain. If conditions allow, the storm drain flow rate would be reduced to prevent exceeding the criterion concentration on an average basis. There will be a plume of elevated concentrations but the criterion pollutant isopleth will form a closed contour relatively close to the storm drain source instead of growing until it exceeds the criterion across the entire cross-section of the stream. Examples based on conditions generally describing a stream such as Arcade Creek are presented below to demonstrate the overall concept. Both storm drain and ambient variables are changed slightly for illustrative purposes.

Site Description

In connection to previous work (Denton 2001), three creek sampling sites were established. Downstream Site A is located adjacent to the existing USGS gaging station (No. 11447360), near Interstate 80 and Watt Avenue in Sacramento, and is 7.72 km upstream of the Natomas East Main Drain. Site B is located 1.84 km upstream of Site A. Site C is located 2.96 km upstream of Site A. The storm drain (SD) is located 0.50 km upstream of Site B. The USGS gaging station located on the creek provides an historic hydrologic context for the data collected during the study.

Methods

Diazinon and water chemistry

Water samples were collected from Sites A (1.8 km downstream of Site B), B (in between Sites A and C and slightly below the storm drain), and C (2.9 km upstream of Site C) in Arcade Creek during a twelve-month period (August 2000 through July 2001) to characterize the spatial and temporal concentrations of diazinon. Sampling events consisted of (bi-weekly) dry weather baseline values and storm events (n = 4). The storm events included both fall and winter events. Stream flow and stage data and general water quality measurements were collected in the field at each sampling event. Rainfall data were obtained from Gage #16 at American River College within the Arcade Creek watershed and used to determine precipitation amounts for the storm events. Such information was used to establish stream flows or other information indirectly.

Diazinon concentration reduction strategy

An examination of Eq. 1 may be used to explain the simple concentration reduction strategy. If C_{SD} is greater than the criterion concentration but C_a is below it, then there will be a threshold storm drain flow rate, $Q_{critical}$, at which the fully mixed concentration will be equal to the criterion concentration.

$$Q_{Critical} = Q_a \frac{C_a - C_{criterion}}{C_{criterion} - C_{SD}} \qquad (2)$$

where $C_{criterion}$ is the diazinon water quality criterion.

If this flow rate is exceeded by the storm drain pump then, downstream of the confluence, where the contributions of both streams are fully mixed, the diazinon criterion will be exceeded. However, if the pump is metered to reduce the flow below the critical flow rate, the receiving stream will exceed the criterion concentration only in a relatively small mixing zone. That is the basis for the simple strategy outlined herein. It can be achieved without using the Visual Plumes model. However, the model can be used to further analyze plume concentration patterns to not only simply meet the fully mixed criterion concentration but, for example, to provide a passage at the confluence in which the criterion concentration is met. The approach applies to suitably designed retention basins.

Estimating concentration contours beyond the stream dilution limit

When using plume models such as Visual Plumes UM3 it is important to understand that the streamflow constrains the amount of dilution that can occur. This is effectively what Eq. 2 states. When this limit is reached UM3 issues a stream-limit statement informing the user that beyond that point additional dilution is not supported by actual events. However, it continues the simulation as if additional diluting water were available. This feature of the model can be exploited to give further estimates of how the concentrated portion of the plume might continue to become more uniform downstream. This approach is similar to the reflection technique used in the PDS surface plume model (Davis 1999).

Unless the concentration isopleth to be contoured is equal to the ambient concentration, the concentration isopleth will be narrower than the width of the turbulent plume itself. Thus, when the stream limit is reached, the concentrated core of the plume will typically occupy only a fraction of the stream width. In the plume fringes, in other words along the banks, the concentrations often will be less than the contoured criterion concentration. As the plume material continues to flow downstream a mixing process similar to the one used in the model will result in the centerline concentration decreasing as the bank concentrations increase. Eventually, if sufficient fetch is available and there are not radical changes in stream conditions, a uniform concentration is reached. If the ultimate uniform concentration is higher than the contoured concentration, the ultimate contours will conform to the banks. Otherwise the contour may broaden briefly before ultimately closing as the width of the concentrated core reaches zero.
Visual Plumes UM3 may be used to estimate the contours of the criterion concentration beyond the stream dilution limit by assuring that the mass of pollutant in the plume element remains constant beyond that point. The model entrainment equation then may simulate the continued carrier fluid mixing process on which the appropriate concentration profile is simply superimposed so that the in-stream portion of the pollutant mass remains constant. The carrier mass, i.e., the water in the in-bank portion of the model plume element, is also constant beyond this point, as it must be. It is then simply a matter of solving for the width of the contour to continue to plot the stream core beyond the point of the stream dilution limit being reached. This procedure depends on being able to integrate the concentration profile across the stream, or, across the plume, if it is narrower than the stream.

Plume profiles

It is convenient to express concentration profiles in terms of the relative radius, f,

$$f = \frac{r}{b} \tag{3}$$

where b is the radius of the plume. Upon growing to fill the water column depth the plume becomes a two-dimensional problem, in which case r and b become the relative and full width of the plume.

There are several profiles in common use, including the Gaussian and the 3/2 power profiles (Kannberg and Davis 1976). The former distribution extends to infinity, making it necessary to associate a statistic with the plume boundary. The latter is associated with flux-averaged concentrations. For the material element used in the Visual Plumes UM3 model a third profile, $g(f)$, is adopted, where

$$g(f) = 1 - f^2 \tag{4}$$

The peak-to-mean ratio for this profile, k, for radial symmetry is 2.0. For a reflecting one-dimensional distribution $k = 1.5$. These are consistent with peak-to-mean ratios calculated from experimental data (Tian 2002).

Several related equations are used to apply this profile to the problem of determining specific concentration isopleths in the Visual Plumes UM3 model.

Results

Diazinon

Throughout the study period, all 137 samples collected were above draft USEPA diazinon acute and chronic criteria of 100 ng/L (USEPA 2000). The 10th percentile values of diazinon concentration for Site A, B, C and storm drain (SD) were 201, 216, 204 and 150 ng/L, respectively. The 90th percentile values of diazinon concentration for Site A, B, C and storm drain were 830, 836, 773 and 2415 ng/L, respectively.

The first winter storm event (Table 1) yielded the highest concentrations of diazinon from the storm drain (maximum = 4,800 ng/L), and concentrations remained high in consecutive sampling days. The second winter storm event yielded lower diazinon concentrations with a maximum of 1,300 ng/L, and concentrations decreased more quickly compared to the first sampling event.

The highest diazinon concentrations occurred during the rising limb of the hydrograph (Fig. 1) and in the first few days following the first fall and winter storm events. Diazinon concentrations were generally high during storm events (Fig. 2). However, there are a few diazinon concentration peaks during low or base flows (e.g., 11/17/2000 and 04/02/2001). Typically, higher diazinon concentrations occurred during winter storm events, which may have included both agricultural (i.e., offsite-movement from agricultural fields) and urban use components (Table 1).

Diazinon concentrations were converted to mass loads to express a load to the creek in kg/day. Not surprisingly, streamflow (cfs) and mass loading (kg/day) of diazinon in the creek yielded a good correlation (Fig. 3, $R^2 = 0.71$). Data point (a) was the first winter storm event with a high flow (651 cfs) and a high diazinon concentration (1200 ng/L), whereas the data point (b) was the second winter storm event with a high flow (561 cfs) and a lower diazinon concentration (240 ng/L).

At the storm drain flow rates were estimated from the sump pump rating curve, pipe discharge assembly, and time period during which each of the two sump pumps was operating. Each pump delivered approximately 2000 gpm (0.126 m^3s^{-1}) to Arcade Creek when running. Storm drain mass load ratios were calculated for three different storm dates (February 11, 12, and 19, 2001) when the concurrent flow rates and diazinon concentrations were known for both the storm drain and the creek. The contribution of diazinon from the storm drain ranged from 0.67 to 17.8% of the diazinon load in Arcade Creek.

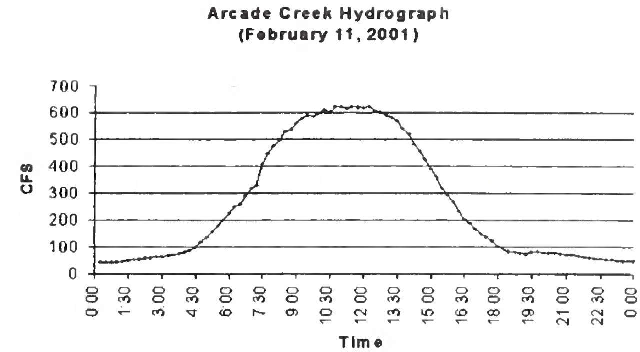

Figure 1. Arcade Creek hydrograph for 2/11/01.

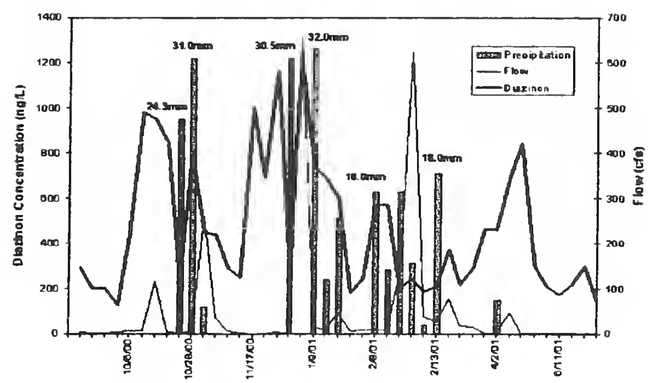

Figure 2. Diazinon concentration, flow, and precipitation at Site A. Note: Site A is the location of a USGS gaging station No. 11447360.

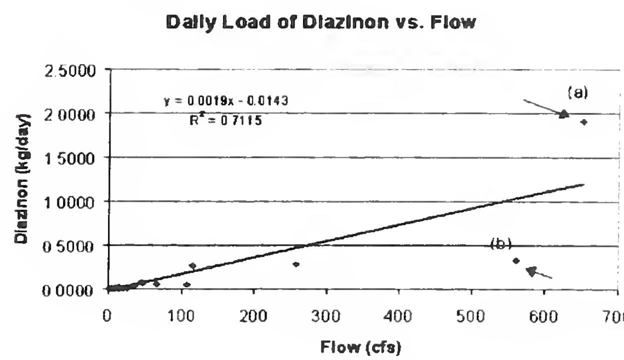

Figure 3. Mass loading of diazinon vs. flow (August 2000 - July 2001).

Model sensitivity run

Several realistic hypothetical cases are developed to show the sensitivity of plume area exceeding the criterion value to variations in flow. The results are shown in Fig. 4. Plumes (a) through (d) correspond to storm drain flows of 0.233, 0.243, 0.253, and 0.116 m^3s^{-1}, approximately one (0.116) or two pumps pumping at once.

Other model input conditions
Effective creek width, 10 m
Average flow depth, 3 m
Arcade Creek flow, 17 ms^{-1}
Arcade Creek diazinon concentration, 90 ng/L
Storm drain diazinon concentration, 800 ng/L
Depth to storm drain centerline, 2.7 m
Effective port diameter, 0.4 m
Isopleth (solid contours) concentration, 100 ng/L
Isopleth (dashed contour) concentration, 95 ng/L

Table 1. Winter storm event diazinon concentrations for storm drain

Storm date	Time interval	Diazinon ng/L)
01/09/2001	0-6 h	3 000
(1st winter event)	7-12 h	2 300
	13-18 h	4 600
	19-24 h	4 800
01/10/2001	0-6 h	2 700
	7-12 h	3 100
	13-18 h	4 500
	19-24 h	2 600
01/11/2001	0-6 h	3 100
	7-12 h	2 100
	13-18 h	1 600
	19-24 h	2 000
02/10/2001	0-6 h	1 300
(2nd winter event)	7-12 h	1 100
	13-18 h	800
	19-24 h	740
	24h composite	690
02/11/2001	0-6 h	360
	7-12 h	390
	13-18 h	320
	19-24 h	280
	24h composite	400
02/12/2001	0-6 h	580
	7-12 h	370
	13-18 h	400
	19-24 h	570
	24h composite	630
02/13/2001	0-12 h	440
	13-18 h	410
	19-24 h	450
	24h composite	450
02/19/2001	0-6 h	1 200
(3rd winter event)	7-12 h	1 200
	13-18 h	1 100
	19-24 h	1 200
	24h composite	1 100
02/25/2001	0-6 h	1 200
	7-12 h	1 900
	13-18 h	1 600
	19-24 h	1 700
	24h composite	1 600

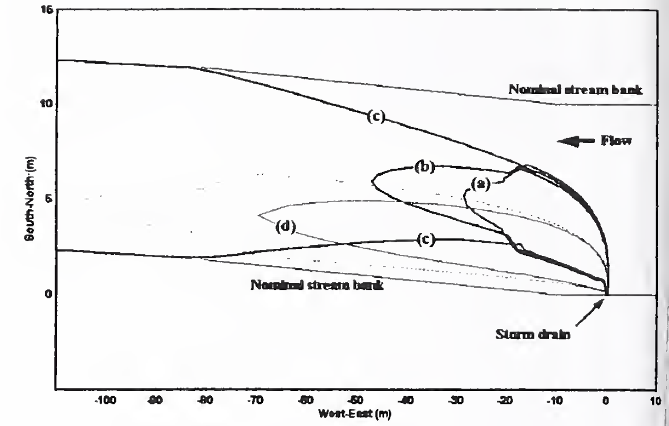

Figure 4. Simulated plumes--solid isopleths define the 100 ng/L ambient diazinon criterion concentration. Storm drain flows are (a) 0.233, (b) 0.243, (c) 0.253, and (d) 0.116 m^3s^{-1}, respectively. The light dashed isopleth, 95 ng/L concentration, part of Plume (d), shows how concentrations further decrease under these conditions.

Based on flow of 17 m^3s^{-1} (600 cfs) in Arcade Creek (near the upper range of wet weather flow rates) and using Eq. 2, the critical storm drain flow is computed to be approximately 0.243 m^3s^{-1}.

Plumes (a) through (c) illustrate the extreme sensitivity of the plume area in excess of the diazinon criterion to small fluctuations in storm drain flow as the flow approaches and then exceeds the critical storm drain flow. In theory even smaller increments in flow would illustrate the observed sensitivity to flow greater than the critical flow, however, the time step used UM3 is too large to show it.

By reducing the flow rate by a factor of two, perhaps by shutting off one of two pumps, the plume area in Plume (d) (Fig. 4) is produced. While it is somewhat larger in area than Plume (a), it is nearer to one bank and rapidly dilutes further as shown by the dashed 95 ng/L concentration isopleth. Plume (d) bends into the current more rapidly due to a reduction in discharge velocity. In this case the criterion can be safely met outside the plume's mixing zone.

Discussion

Plume sensitivity to critical flow fluctuations

The plume area is extremely sensitive to storm drain flow near critical flow due to the fact that center of the plume typically has concentrations well in excess of the 100 ppb criterion concentration when the outer edges of the plume, reach the banks of the creek and available diluting water is fully incorporated into the plume. After this point the plume is no longer diluted further but continues to mix laterally, a process that raises concentrations at the banks to the benefit of decreasing concentration at the plume centerline. In effect, the banks act as reflecting surfaces, reflecting back to the center of the plume material that would otherwise spread laterally in width-unconstrained receiving water. At significantly lower flow, the mixing zone area of the plume (d) is comparatively small and could be easily avoided by sensitive species capable of avoiding such concentrated regions.

Conclusion

As watershed hydrologic processes and conditions such as chemical processes (dry vs. wet weather inputs), background receiving water chemical concentrations, plume morphology, and dynamics of the storm hydrograph are better understood, water quality managers will be able to control and reduce the receiving stream chemical concentrations to specified criteria by varying storm drain flows (thereby controlling the mass of chemical inputs) until the criteria are met. In addition, managers could detour peak flows to treat chemical contaminants via best management practices (BMP), such as wetlands, until the chemical of concern is below the levels that cause acute and chronic toxicity to aquatic organisms. This is important because, as the TMDL is developed for diazinon and limits on its use become necessary, there will be alternative pesticides that will replace diazinon and it will be paramount to have the ability to reduce drain stormflow (thereby reducing mass loading) and to have BMPs to reduce pesticide concentrations before discharge into the aquatic environment.

This analysis demonstrates the utility of the Visual Plumes UM3 model in characterizing the appropriate mixing zone size for urban storm drains and diagramming the chemical concentration isopleths to estimate the location and size of stream plumes that may be exceeding criteria. If the size of the isopleth of concern is limited in space and time, then aquatic organisms may still pass upstream as necessary. For example, referring to Fig. 4, they would find the low concentration region in the vicinity of the flow arrow. Managers must check with the appropriate regulatory authority regarding their state's mixing zone allocation, if allowed and not restricted due to the presence of sensitive spawning grounds or threatened and endangered test species.

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Nutrient and Herbicide Movement in Streamflow from Two Midwestern Watersheds

Gene Alberts, Dan Jaynes, Robert Lerch

Abstract

Watersheds in different geographic areas offer many opportunities to study how differences in climate, crops, soils, and landscapes affect the off-site movement of runoff, sediment, nutrients, herbicides, and other environmental contaminants. The objective of this research was to compare differences in nitrate-nitrogen (NO_3-N) and atrazine concentrations and discharges in streamflow from two watersheds in the Midwest for a 10-year period, 1992 through 2001. One watershed was the 7,280-ha Goodwater Creek watershed located in north-central Missouri, the other, the 5,130-ha Walnut Creek watershed in central Iowa. The Goodwater Creek watershed is located in the Central Claypan Major Land Resource Area (MLRA 113). Soils within this watershed are poorly drained because of a subsurface horizon 150 to 300 mm deep that has a clay content greater than 50%. The Walnut Creek watershed is located in the Central Iowa and Minnesota Till Prairies Major Land Resource Area (MLRA 103). Many of the soils within this watershed are tile drained to increase productivity. Over 80% of the area within each watershed is cropped, but the fraction of corn cropping is higher in the Walnut Creek watershed. The drainage outlet of each watershed is instrumented with a concrete v-notch weir, water stage recorder, and refrigerated automatic pumping sampler. During low stages of streamflow, grab samples of baseflow were collected at the weir for chemical analyses. Mean annual precipitation values (1992-2001) for the Goodwater and Walnut Creek watersheds were 997 and 785 mm, with mean annual streamflow discharges of 392 and 223 mm, respectively. Mean annual NO_3-N discharges from the Goodwater and Walnut Creek watersheds were 6.9 and 20.4 kg ha^{-1}. Flow-weighted NO_3-N concentrations (10 years) from the Goodwater and Walnut Creek watersheds were 1.8 and 9.2 mg L^{-1}, with flow-weighted annual concentrations ranging from 1.2 to 2.7 mg L^{-1} for Goodwater Creek and 6.3 to 13.1 mg L^{-1} for Walnut Creek. The highest NO_3-N concentrations from both watersheds occurred during the April through June seasonal period, with 10-year flow-weighted concentrations of 2.1 and 11.2 mg L^{-1} for the Goodwater and Walnut Creek watersheds. Mean annual atrazine discharges from the Goodwater and Walnut Creek watersheds were 18.1 and 1.7 g ha^{-1}. Flow-weighted atrazine concentrations from the Goodwater and Walnut Creek watersheds for the 10-year period were 4.6 and 0.8 ug L^{-1}, with flow-weighted annual concentrations ranging from 0.8 to 19.3 ug L^{-1} for Goodwater Creek and 0.1 to 1.9 ug L^{-1} for Walnut Creek. The highest atrazine concentrations occurred during the April through June seasonal period, with 10-year flow-weighted concentrations of 9.3 and 1.3 ug L^{-1} for the Goodwater and Walnut Creek watersheds. Our results show that water quality problems associated with crop production are dependent on how water moves within and through the watershed topography because of soil and anthropogenic factors. In the Goodwater Creek watershed, streamflow is comprised of about 90% surface runoff and 10% ground water recharge, and in such a hydrologic environment, atrazine is much more susceptible to movement. The soils within the Walnut Creek watershed are more permeable and the pot-hole topography limits the amount of natural surface runoff. About 74% of the streamflow discharge from Walnut Creek over the 10-year period was tile drainage. NO_3-N, being mobile in the soil, readily entered the tile drain system and moved rapidly from upland areas of the watershed to Walnut Creek.

Keywords: atrazine, NO_3-N, water quality

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Introduction

The Clean Water Act of 1972 has improved the quality of our nation's water resources primarily by targeting and solving point-source pollution problems associated with discharge of contaminated industrial effluent. However, pollution from nonpoint sources is currently perceived as one of the nation's most serious water quality problems. The U.S. Environmental Protection Agency (EPA) has estimated that 70 percent of impaired rivers and streams and 49 percent of impaired lake areas are impacted adversely by agricultural activities (U.S. EPA 1998).

The most important nonpoint-source water quality problem in the Midwest is stream water contamination resulting from agricultural practices. This research focuses on contamination associated with nutrients and pesticides, specifically NO_3-N and atrazine. Discharges of NO_3-N from the Mississippi River Basin have been linked to the increased spread and severity of hypoxia within the Gulf of Mexico (Rabalais et al. 1996, CAST 1999). About 55% of the nitrogen released from the Basin has been attributed to nitrogen fertilizer used to supplement natural soil fertility in crop production. The leaching of NO_3-N and the interception and release from tile-drained fields have been identified as important causal factors (Tomer et al. 2003).

Atrazine has been used by farmers for over 40 years as a low-cost, broad spectrum, and season-long herbicide. In recent years, it has been linked to hormonal disruptions, both in humans and animals, and an increased risk of cancer. Currently, atrazine is applied to about two-thirds of the corn and sorghum acreage in the U.S. Atrazine from farm fields and watersheds has been found to be the major contaminant of stream water throughout the Midwestern U.S., being detected in 90% or more of the stream water samples. (Thurman et al. 1992, Lerch et al. 1998, Blanchard and Lerch 2000). Concentrations often exceed the EPA maximum contaminant level (MCL) of 3 ug L^{-1} in the post-plant period, following atrazine application. Because seasonal rainfall amounts and distribution cannot be accurately predicted, herbicide losses in runoff are largely unpredictable. To reduce atrazine losses in surface runoff, research has focused on approaches to reduce application rates, particularly on soils with high runoff potential (Donald et al. 2001).

Watersheds can vary greatly in their hydrologic and water quality responses to management inputs, even for similar cropping systems, because of differences in climate, soils, management inputs, and landscape topography. Research results from multiple watersheds located in different areas of the U.S. are needed to help assess regional and national impacts of agricultural practices on water quality.

The objective of this research was to compare differences in NO_3-N and atrazine concentrations and discharges in streamflow from two watersheds in the Midwest. The study spans a 10-year period, 1992 through 2001, and represents normal variations in precipitation characteristics; such as amount, intensity, and timing relative to cultural operations and chemical applications.

Methods

Goodwater Creek watershed

This 7,260-ha watershed is within the Central Claypan Soils Major Land Resource Area (MLRA 113), an area of about 4 M ha. About 90% of the watershed is cropped following the ranking of soybeans >> wheat > corn > sorghum > hay. Typical crop rotations are corn/soybeans, corn/soybeans/wheat, soybean/soybeans/wheat, and sorghum/soybean/wheat.

The topography is characterized by broad, nearly flat divides, gentle side slopes, and broad alluvial valleys often dissected with small streams. The soils generally have a silt-loam surface texture underlain by an argillic horizon 150 to 300 mm below the soil surface. The clay content of the argillic horizon is generally > 50% and is comprised primarily of montmorillonite. Principle soils are the Udollic Ochraqualfs, Albaquic Hapludalfs, and Vertic Ochraqualfs. Many soils that occupy moderately to severely eroded side slope positions have a silty clay texture due to the incorporation of the claypan into the surface horizon by plowing and disking operations.

The long-term mean annual precipitation is 929 mm. Spring rainfall often occurs as intense, short duration thunderstorms on recently tilled and planted soils. Fall rainfall often occurs as low intensity, long duration events associated with slow moving cold fronts. Water content of the soil root zone decreases during the summer because high evapotranspiration from rapidly growing and maturing crops exceeds rainfall.

Walnut Creek watershed

This 5,130-ha watershed is within the Central Iowa and Minnesota Till Prairies Major Land Resource Area (MLRA 103), an area of about 7.2 M ha. About 92% of the watershed is in crop production, primarily a corn-soybean rotation. About 15% of the corn acreage is

in continuous cropping. Topography is undulating with poorly developed surface drainage. Low relief swells and swales create closed drainage depressional areas called potholes. The western portion of the watershed has an extensive tile drainage network that connects the potholes throughout the landscape. Many potholes have risers above the soil surface to capture surface runoff. Tile drainage feeds into a network of large subsurface drains maintained by county drainage districts. Almost 75% of the watershed is tile drained, and it is this underground delivery system of drainage water to Walnut Creek that is its major defining feature (Jaynes et al., 1999). There is more natural surface drainage in the eastern portion of the watershed near the outlet.

Principle soils within the watershed are Typic Hapludolls, Typic Haplaquolls, and Aquic Hapludolls. These upland soils generally have a loam texture and moderate permeability. Soils in, and adjacent to, the pot holes typically have mucky silt loam and silty clay loam textures and are poorly to very poorly drained.

The long-term mean annual precipitation is 818 mm. Rainfall during the spring and summer often occurs as brief, but intense, thunderstorms.

Instrumentation and laboratory procedures

Similar procedures were used for field data and sample collection as well as laboratory preparation and analyses of streamflow samples. Streamflow was measured continuously at the outlet of both watersheds using a data logger to measure stream height (stage). Measured and/or theoretical relationships between discharge rate and stage were used to compute streamflow discharge. When stage height reached a predetermined level, the pumping sampler at each location was activated. Stringent quality assurance/quality control procedures were used in the field collection, transport, storage, processing, and analyses of all samples. NO_3-N concentrations were determined using similar autoanalyzers that reduce nitrate to nitrite with concentrations determined colorimetrically. For atrazine, samples were passed through a solid phase extraction cartridge and then analyzed using gas chromatography with N-P or mass spectral (MS) detection. To compute discharges, sample concentrations were matched with the representative flow volume and multiplied. The products of these computations were then summed over time.

Results

Precipitation and streamflow

Mean annual precipitation values for the Goodwater Creek and Walnut Creek watersheds were 997 and 785 mm. For the Goodwater Creek watershed, precipitation for the 10-year period was 7% higher than the 60-year long term value of 929 mm; while for Walnut Creek, the 10-year mean annual was 4% lower than the 30+ long-term value of 818 mm.

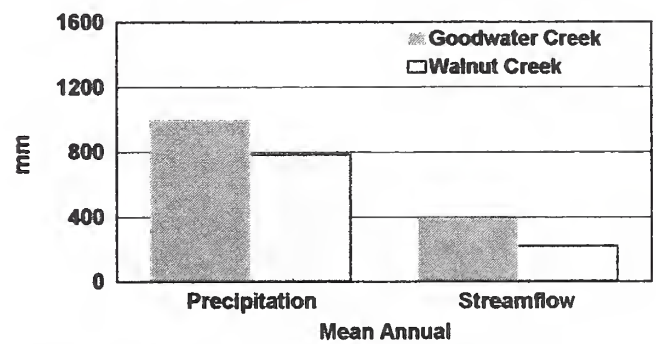

Figure 13. Mean annual precipitation and stream flow.

The 10-year mean annual streamflow values from Goodwater Creek and Walnut Creek were 392 and 223 mm, respectively (Figure 1).

Figure 2 shows the annual values of precipitation and streamflow during the 10-year period. Annual precipitation for Goodwater Creek ranged from 775 mm in 1992 to 1373 mm in 1993, while annual precipitation for Walnut Creek ranged from 497 mm in 2000 to 1288 mm in 1993. Highest streamflow from both watersheds occurred in 1993, with values of 785 and 865 mm from Goodwater Creek and Walnut Creek. Lowest streamflow from both watersheds occurred in 2000, with 50 mm of streamflow from Goodwater Creek and 7 mm from Walnut Creek.

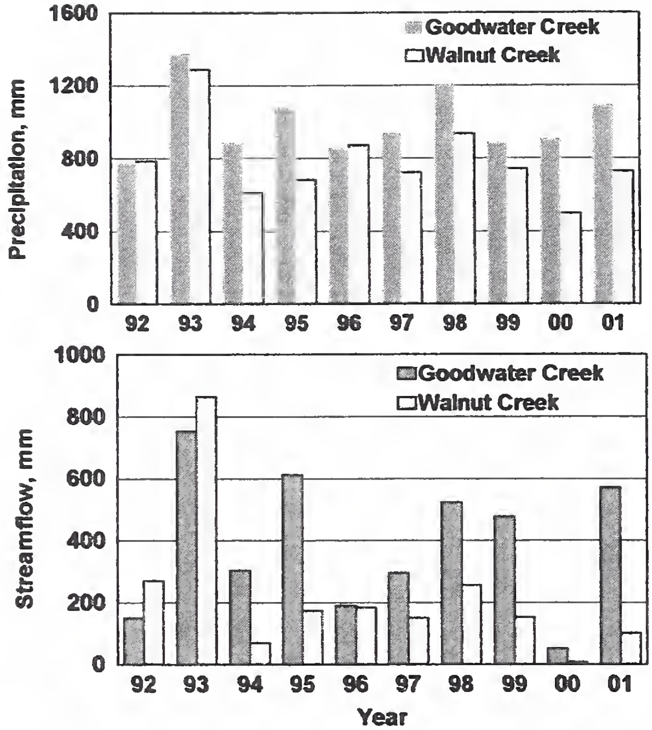

Figure 14. Annual precipitation and streamflow discharges.

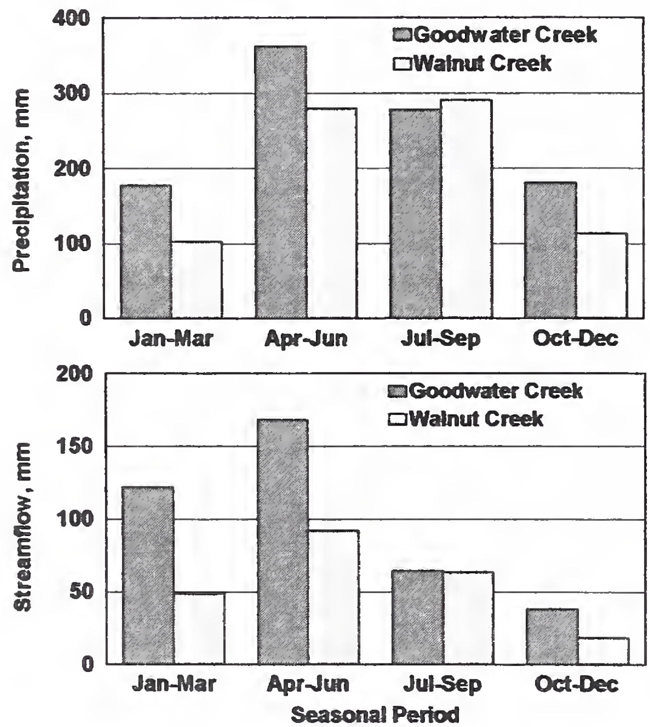

Figure 15. Seasonal precipitation and streamflow discharges.

For Goodwater Creek, 18, 36, 28, and 18% of the precipitation occurred for the Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec seasonal periods (Figure 3). Respective values for Walnut Creek were 13, 36, 37, and 14%. For Goodwater Creek, 31, 43, 16, and 10% of the streamflow occurred for the Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec seasonal periods. Respective values for Walnut Creek were 22, 41, 29, and 8%. On both an absolute and relative basis, the period of highest streamflow was the Apr-Jun period coinciding with the period of N fertilizer and atrazine application for row-crops in both watersheds.

Nitrate-nitrogen

Mean annual NO_3-N discharges from the Goodwater and Walnut Creek watersheds for the 10-year period were 6.9 and 20.4 kg ha^{-1}. Annual NO_3-N discharge from Goodwater Creek ranged from 0.75 kg ha^{-1} in 1993 to 11.8 kg ha^{-1} in 1993, while annual discharges from Walnut Creek ranged from 0.76 kg ha^{-1} in 2000 to 61.5 kg ha^{-1} in 1993 (Figure 4).

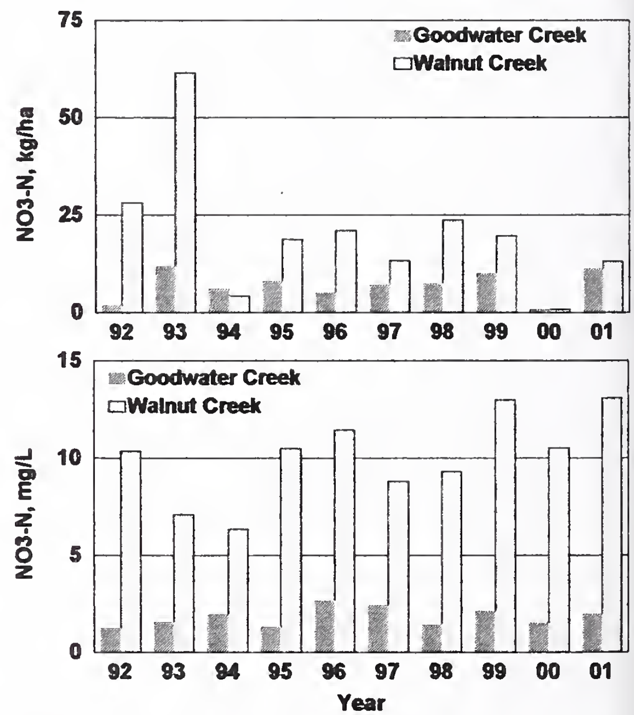

Figure 16. Annual discharges and concentrations of NO_3-N.

Flow-weighted NO_3-N concentrations from the Goodwater and Walnut Creek watersheds for the 10-year period were 1.8 and 9.2 mg L^{-1}. Figure 4 shows the year-to-year values of flow-weighted NO_3-N concentrations. Flow-weighted NO_3-N concentrations for Goodwater Creek ranged from 1.2 mg L^{-1} in 1992 to 2.7 mg L^{-1} in 1996, while flow-weighted concentrations for Walnut Creek ranged from 6.3 mg L^{-1} in 1994 to 13.1 mg L^{-1} in 2001. Flow-weighted concentrations for 6 of the 10 years from Walnut Creek exceeded the 10 mg L^{-1} MCL established by the EPA for drinking water.

Seasonal NO_3-N concentrations for the two watersheds are shown in Figure 5. For Goodwater Creek, seasonal flow-weighted concentrations for the Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec periods were 1.6, 2.1, 1.5, and 1.2 mg L^{-1}, while respective values for the Walnut Creek watershed were 7.2, 11.2, 7.6, and 9.7 mg L^{-1}.

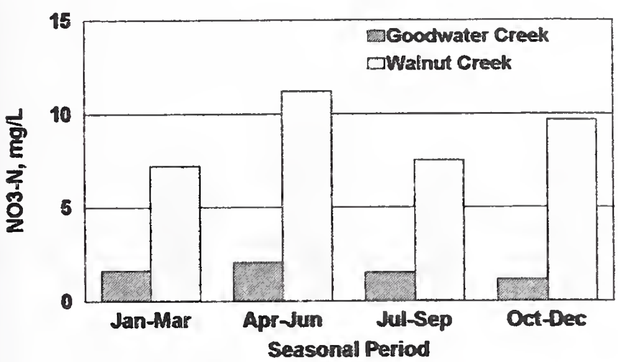

Figure 17. Seasonal NO_3-N concentrations.

Atrazine

Mean annual atrazine discharges from the Goodwater and Walnut Creek watersheds for the 10-year period were 18.1 and 1.8 g ha^{-1}. Annual atrazine discharge from Goodwater Creek ranged from 1.3 g ha^{-1} in 1992 to 36.7 g ha^{-1} in 1996, while annual discharges from Walnut Creek ranged from 0.10 g ha^{-1} in 2000 to 7.5 g ha^{-1} in 1993 (Figure 6).

Flow-weighted atrazine concentrations from the Goodwater and Walnut Creek watersheds for the 10-year period were 4.6 and 0.78 ug L^{-1}. Flow-weighted atrazine concentrations from Goodwater Creek ranged from 0.84 ug L^{-1} in 1992 to 19.3 ug L^{-1} in 1996, while flow-weighted concentrations from Walnut Creek ranged from 0.08 ug L^{-1} in 1997 to 1.9 ug L^{-1} in 2001 (Figure 6). Flow-weighted concentrations from Goodwater Creek exceeded the 3 ug L^{-1} MCL established by the EPA for 7 of the 10 years.

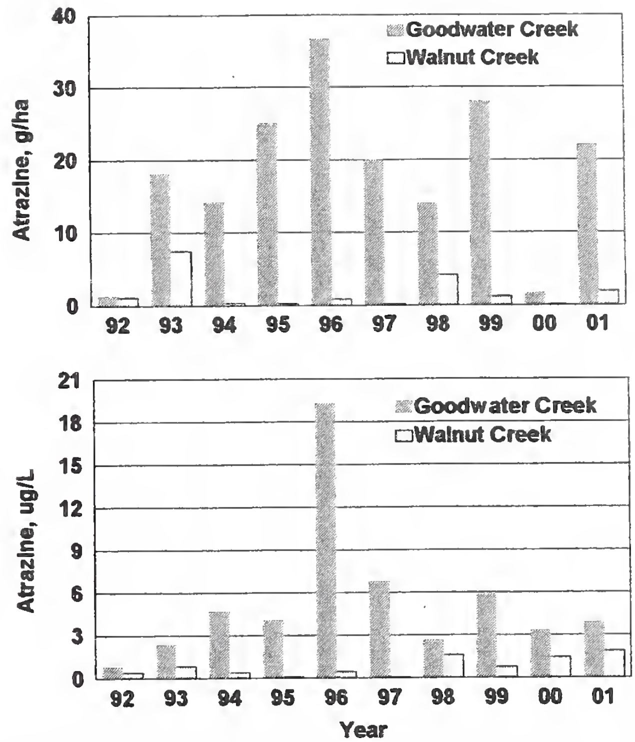

Figure 6. Annual discharges and concentrations of atrazine.

The importance of rainfall timing relative to atrazine application is shown by examining the 1996 data. Annual streamflow from the Goodwater Creek watershed was 190 mm (Figure 2), with 119 mm occurring in May. Atrazine discharge in May was 32.5 g ha^{-1}, representing 88% of the 36.5 g ha^{-1} annual loss.

Seasonal atrazine concentrations for the two watersheds are shown in Figure 7. For Goodwater Creek, seasonal flow-weighted concentrations for the Jan-Mar, Apr-Jun, Jul-Sep, and Oct-Dec periods were 0.70, 9.3, 2.3, and 0.53 ug L^{-1}, while respective values for the Walnut Creek watershed were 0.10, 1.3, 0.81, and 0.07 ug L^{-1}.

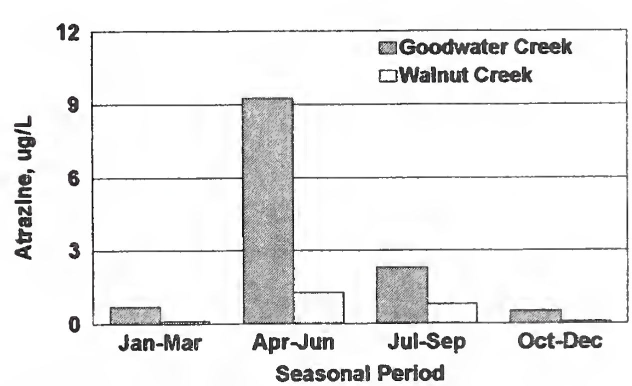

Figure 7. Seasonal atrazine concentrations.

Conclusions

NO_3-N and atrazine concentrations and discharges were measured over a 10-year period (1992-2001) from two Midwestern agricultural watersheds to better understand the influence of climate, soils, and topography on the transport of these chemicals through and out of the watersheds. One watershed was the 7,260-ha Goodwater Creek watershed located in north-central Missouri, the other was the 5,230-ha Walnut Creek watershed located in central Iowa. Our results showed that the primary water quality problem for each watershed was different. Annual flow-weighted NO_3-N concentrations from the Walnut Creek watershed exceeded the 10 mg L^{-1} maximum contaminant level (MCL) for drinking water 6 of 10 years. In contrast, the highest annual flow-weighted NO_3-N concentration from the Goodwater Creek watershed was 2.7 mg L^{-1}. Annual flow-weighted atrazine concentrations from the Goodwater Creek watershed exceeded the 3 ug L^{-1} MCL for drinking water 7 of the 10 years. Atrazine concentrations and discharges from the Goodwater Creek watershed were highly seasonal, with 43% of the mean annual stream flow and 86% of the atrazine discharge occurring during the April through June seasonal period. The highest annual flow-weighted atrazine concentration from the Walnut Creek watershed was 1.9 ug L^{-1}. One of the major differences between the watersheds is how water moves from the landscape into the Creek system. In the Walnut Creek watershed, about 74% of the streamflow is associated with tile drainage through an extensive network of field and county tile drains. NO_3-N is mobile in the soil and can readily move through the soil and into the tile drainage system. In the Goodwater Creek watershed, about 90% of the streamflow is associated with surface runoff from a well-dissected landscape. Data will continue to be collected from these watersheds to better understand how improved nitrogen and atrazine best management practices at the field scale and other management practices at the watershed scale are improving streamflow water quality for optimum downstream water quality benefits and ecosystem protection.

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The authors appreciate the reviews of John Sadler and Fessehaie Ghidey.

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Fifty Years of Watershed Research on the Fernow Experimental Forest, WV: Effects of Forest Management and Air Pollution in Hardwood Forests

M.B. Adams, P.J. Edwards, J.N. Kochenderfer, F. Wood

Abstract

In 1951, stream gaging was begun on five small headwater catchments on the Fernow Experimental Forest in West Virginia, to study the effects of forest management activities, particularly timber harvesting, on water yield and quality. Results from these watersheds, and others gaged more recently, have shown that annual water yields increase in proportion to the basal area cut, and that conversion of hardwood stands to conifers significantly decreases water yield. Accelerated nutrient leaching did not occur after heavy cutting, but did follow when herbiciding sustained barren conditions after clearcutting. Results from Fernow studies have shown that sediment increases in in-stream exports are minor and short-lived and mostly from roads when best management practices (BMPs) are conscientiously employed. Research on forest access roads and BMPs has provided important management guidelines for forest landowners. More recently work has focused on air pollution effects on hardwood forests. Ongoing research is evaluating the role of forest management on channel stability, nutrient cycling in polluted environments, and the interaction of forest management and air pollution on forest ecosystems.

Keywords: Fernow Experimental Forest, timber harvesting, water yield, nutrient cycling, sediment, watershed studies

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Introduction/Description

The Fernow Experimental Forest (FEF) was established to conduct research in forest and watershed management in the central and northern Appalachians and to provide relevant information to forest landowners. The 1868-ha FEF is located south of Parsons, West Virginia (Figure 1). The Elklick watershed, which later became the bulk of the FEF, was initially logged between 1903 and 1911 during the railroad-logging era (Trimble 1977). Most of the watershed was not farmed and the forest was allowed to regenerate naturally after logging. The federal government purchased the land in 1915 and dedicated it to forest and watershed research in 1934. The Elklick basin was selected because of its good soil, excellent young regrowth, ready access to many wood-using plants, and was considered to be representative of more than 5 million ha of mountainous forestland in West Virginia and adjacent states. Chestnut blight in the 1930s was the next major disturbance, and resulted in a 25 percent reduction in standing volume on the experimental forest. Closed during World War II, silviculture and watershed research began again in 1948 and have continued to date without interruption.

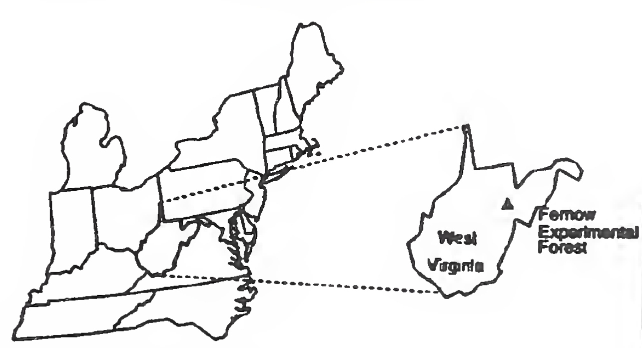

Figure 1. Location of Fernow Experimental Forest, WV.

The FEF is located in the Allegheny Mountains Section of the Central Appalachian Broadleaf Forest, and vegetation is classified as mixed mesophytic. Characteristic species include northern red oak (*Quercus rubra*), yellow-poplar (*Liriodendron tulipifera*), black cherry (*Prunus serotina*), sugar maple (*Acer saccharum*), sweet birch (*Betula lenta*), red maple (*A. rubrum*), and American beech (*Fagus grandifolia*). The mountainous topography has elevations ranging from 530 to 1115 m above sea level, with slopes of 20-50 percent common. Annual precipitation averages about 1480 mm and is distributed throughout the year. The growing season is May through October with an average total frost-free period of 145 days. Snow is common, but snowpacks generally last no more than a few weeks. Mean annual temperature is 9.2°C but temperatures reach -20°C during most winters. Large rainfall events normally are associated with hurricanes.

Soils are moderately deep and well-drained, formed in material weathered from interbedded shale, siltstone and sandstone. Soils average 1 m depth, and contain considerable stones and large gravels.

Research Results: Timber Harvesting

Effects on water yields

The earliest watershed studies on the FEF evaluated the effects of timber harvesting on streamflow and water yield. Five forested watersheds were gaged in 1951, then after a 6-year calibration period, four cutting treatments ranging from clearcutting to intensive selection were applied (Table 1). Other watersheds have been gaged during the last 50 years, and treatments applied, as the questions of interest to landowners and scientists evolved over time. Summaries of these studies (Reinhart et al. 1963, Kochenderfer et al. 1990, Hornbeck et al. 1993) have concluded that (1) removing significant amounts of timber increased annual water yield from a watershed, (2) the increases were related to the intensity of harvest and the basal area removed from the stand, with the largest increases coming from clearcut watersheds, and (3) generally, at least 25% of the basal area of a stand must be removed before a change in annual water yield can be detected. Greater flow increases during the growing season suggest reduced losses of soil moisture to transpiration after cutting trees. Even the increases from clearcuts were relatively short-lived, and annual water yields from harvested watersheds, with a few notable exceptions, returned to predicted flows within 5-10 years after treatment. This return to predicted levels coincides with crown closure of the stands for most of the watersheds. Use of herbicides on FEF6 and FEF7 to control regrowth significantly prolonged increases in annual flow relative to the watersheds that were only clearcut with no vegetation control (FEF3). Continuing reductions in annual water yield on FEF6 are attributed to the greater interception and transpiration by the planted conifer (*Picea abies*) stand compared to the original hardwood stand (Helvey 1967). Annual streamflow reductions have averaged 23% for the past six years on FEF6, with most of the significant decreases during the dormant season.

Changes in seasonal water yield probably reflect the severity of treatment as well as vigor of regrowth.

Table 1. Experimental Watersheds on the FEF.

WS & area (ha)	Treatment	Date
WS1, 29.9	Clearcut to 6 in. dbh	1957/58
	Urea application	1971
WS2, 15.4	Six 17 in. diameter limit cuts	1958-1996
WS3, 34.4	Two intensive selection cuts	1958- 1963
	Clearcut, all but buffer strip	1969/70
	Aerial application of fertilizer	1989-present
WS4, 38.9	Reference	Activated 1951
WS5, 36.4	Six single-tree selection cuts	1958-1998
WS6, 22.3	Lower and upper watershed clearcut and herbicided	1964-1969
	Planted with Norway spruce	1973
	Aerial herbicide applications	1975,1980
WS7, 24.3	Upper and lower watershed cut and herbicided	1963-1969
WS10, 15.2	Control	Activated 1984
WS13, 14.2	Control	Activated 1988

Growing season water yield increases due to harvesting were longer-lived for FEF6 than FEF7 because other vegetative regrowth (competing hardwoods) was controlled with aerial herbicide applications in 1975 and 1980 to release planted Norway spruce (Wendel and Kochenderfer 1984), while the rapid decline in growing season water yield increases on FEF3 was attributed to luxuriant vegetative regrowth (Aubertin and Patric 1974). Vegetation was set back to an earlier successional stage on FEF7 by the herbicide treatments (Kochenderfer and Wendel 1983). Because of the repeated herbicide treatments, stump sprouts were nearly eliminated on FEF7, and most regeneration originated from seeds. Semiwoody herbaceous vegetation (mainly *Rubus* spp.) and grasses remained dominant for 5 years after the cessation of herbicide treatments. On FEF3, woody vegetation comprised of seedlings and seedling and stump sprouts developed very rapidly. This deeper-rooted vegetation had better access to soil water, resulting in greater transpiration during the growing season than on FEF7.

Tree species composition after harvesting also can affect annual water yields, For example, lower than predicted growing season yields on FEF3 in the late 1980s-2002, may be due to the large increase in black cherry (from 5 % prior to cutting to 50% of basal area 20 years after cutting). Black cherry consistently transpires at the highest rate per unit of leaf surface area found in hardwoods (Kochenderfer and Lee 1973). It is also interesting to note that during the ten-year time period (1989-1999) that coincided with a fertilization treatment on FEF3, growing season flows were consistently less than predicted. Patric and Smith (1978) suggested that fertilization decreased throughfall and temporarily increased interception, partly through the mechanism of increased leaf area. These data provide some support for this hypothesis.

Effects on peak flows

Dormant season storm flows and peak flows draining the harvested watersheds differed little from those of the control watersheds, probably because similar evaporative losses from both sustained similar soil moisture levels. However, growing season peak flows were consistently greater on some clearcut watersheds where soil moisture was higher for a short period after cutting until vegetation regrowth. This effect was more pronounced for smaller storms. The number of runoff events that are large enough to constitute stormflow increased with clearcutting (Bates 2000). Snowmelt storm flows peaked earlier on clearcut watersheds than the control watersheds, due to greater net radiation on the snow cover of clearcut watersheds. Except for snowmelt and overland flow from logging roads, there were no dramatic timing changes in the hydrographs after harvest, and subsurface flow is the main runoff production mechanism both before and after harvests.

Effects on sediment yields

Sediment exports in stream water prior to treatment and on the reference watersheds ranged from 6 to 25 kg ha^{-1} yr^{-1} (Patric 1980, Kochenderfer et al. 1987). Clearcutting using an unplanned road system and no BMPs increased sediment yields to over 3000 kg ha^{-1} on FEF1 during logging (Kochenderfer and Hornbeck 1999) in 1957 and 1958, compared to only 97 kg ha^{-1} in 1970 when BMPs were carefully applied to FEF3. On both watersheds, sediment yields decreased rapidly to 44 kg ha^{-1} and 28 kg ha^{-1} respectively in 5 years. Most sediment was produced during storm flows (Kochenderfer et al. 1987). For all of these studies, turbidity or suspended sediment returned to pre-treatment or reference levels within a few years. Overland flow seldom was observed, only occurring on or directly below steep roads (Patric 1973).

Effects on stream temperature

Clearcutting FEF1 and leaving no streamside vegetation raised stream temperature 4.5°C during the growing season, and decreased temperature 2°C during the dormant season (Reinhart et al. 1963). Temperatures returned to pretreatment levels within 3 years. By contrast, clearcutting FEF3 in 1969 had no effect on temperature when a 50-ft wide vegetated buffer strip was left along the channel. Removal of that buffer strip increased stream temperature about 4°C during the summer it was cut. Channel shading was sufficient after 5 years of regrowth to return temperatures to pre-clearcutting levels (Patric 1980).

Effects on stream chemistry

Stream water nitrate concentrations are of concern to forest managers because forests provide the source of most drinking water in the U.S., and because nitrate concentrations can be a very sensitive indicator of disturbance and change in plant and soil systems. Many studies on the FEF have shown that forest harvesting, including clearcutting has little effect on nitrate concentrations. Clearcutting on FEF3 (Figure

2) resulted in maximum concentrations that were well below safe drinking water standards. By contrast, forest fertilization on FEF1 with 500 kg ha^{-1} of urea in 1971 resulted in increases of monthly maximum nitrate from 0-8 mg L^{-1} to 70 mg L^{-1}. Devegetating FEF6 and FEF7 with herbicides repeatedly for several years after clearcutting also caused substantial increases in nitrate concentrations. These results demonstrate the importance of vegetation in maintaining water quality through nutrient uptake and control of microclimate.

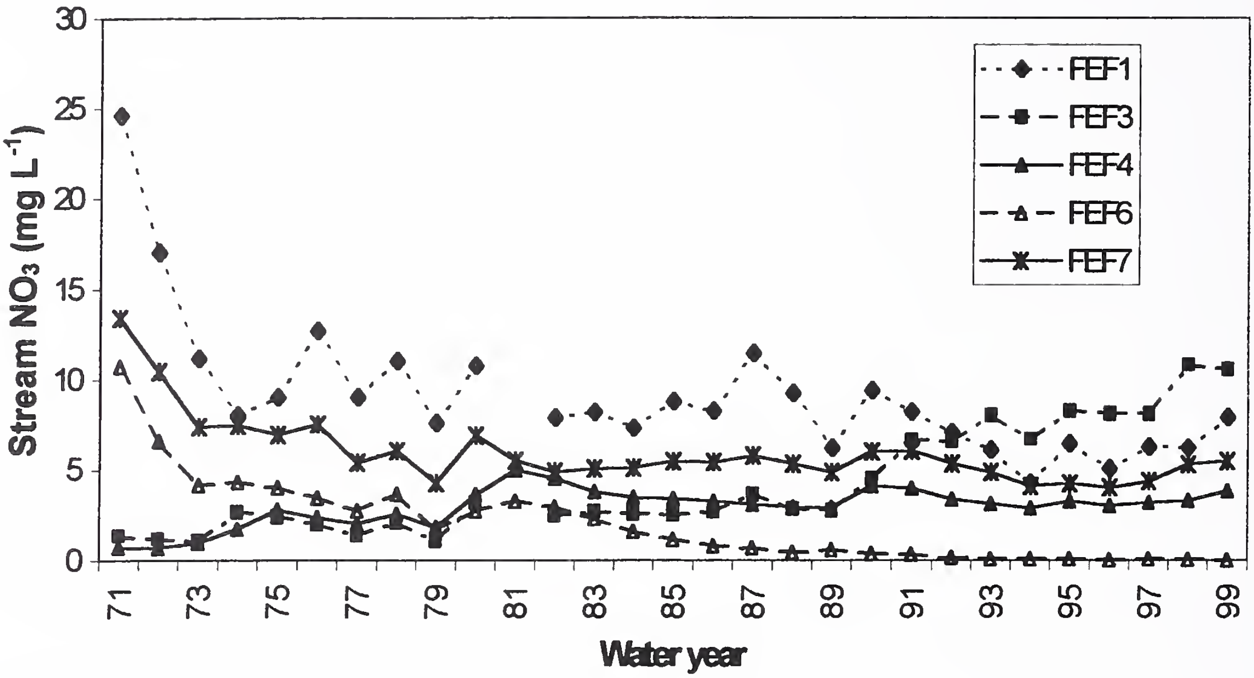

Figure 2. Levels of nitrate concentration at five sites, 1971-1999.

Research Results: Air Pollution

In recent years, scientists at FEF have been involved in monitoring air quality and evaluating the effects of air pollution on forest ecosystems. The first monitoring station in the nation-wide National Acid Deposition Program (NADP) came on-line in Parsons in 1978 as part of the research program at FEF. Data from this site showed that the central Appalachians receive some of the highest levels of nitrogen and sulfur deposition (i.e. acid rain) in the U.S., despite its rural location. Ozone monitoring at low and high elevation sites near the FEF, along with analysis of ozone concentrations throughout the central Appalachians, has shown that remote areas can have poor air quality due to long-distance transport of pollutants, and that existing monitoring networks may poorly capture the variation of air quality in mountainous and rural areas.

Research on the FEF also has shown that nitrogen may be a more significant actor than sulfur relative to its effects on forest ecosystems (Adams et al. 2000). Increased nitrogen deposition can result in leaching of base cations, particularly calcium and magnesium, from the soil (Edwards et al. 2002), with significant concerns for sustainable forest health. Acidification of streams may result from high levels of nitrate and/or sulfate deposition. No significant decreases in tree growth or forest health have been detected on the FEF as a result of elevated nitrogen and sulfur deposition, however.

Summary

More than 50 years of watershed research at the Fernow Experimental Forest has provided important information about the effects of forest management on annual water yield, storm flows, sediment export, stream temperatures, and nutrient cycling. In addition, a much more thorough understanding of the use of water and nutrients by forest ecosystems in the central Appalachians has resulted. Research on air pollution effects on forests has informed national policy discussions. Scientists on the FEF continue to conduct relevant research on long-term species composition trends in central Appalachian forests, the role and effects of fire, the integration of silvicultural activities for landscape level management for a variety of resources, including wildlife, water and forest productivity, and other important management and research questions. The research program at the FEF provides ample evidence of the utility and importance of long-term watershed research, and the need for well-designed experimental manipulations to address complex questions over the life of a forest.

Acknowledgments

The contributions of the following technicians who have faithfully and carefully collected watershed data over the years are acknowledged: Bud Fridley, John Campbell, Doug Owens, Melvin Owens, Clifford Phillips, Frank Long, Cloyd Rinehart. This manuscript was greatly improved by the reviews of Jim Burger, Jim Patric, and Bill Peterjohn.

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Integrating Science with Watershed Decision Making I

Calculating the Cost of Reducing Erosion from a Small Rangeland Watershed

Philip Heilman, Yanxin Duan, Ryan Miller, D. Phillip Guertin

Abstract

Sediment is an important pollutant in the United States. Attempts to control sediment are under consideration for water bodies where sediment-affected water does not support designated uses. To be economically efficient, policies to control sediment should achieve required reductions in sediment at least cost. On rangelands, quantifying the scope of sediment reduction available through land management is problematic given the difficulty in quantifying sediment detachment, transport and deposition processes, and watershed runoff and sediment yield. As many ranches are economically stressed, imposing additional costs to reduce sediment could drive some ranchers out of business. A constrained optimization model was built that simulates the effect of imposing a constraint to reduce watershed sediment yield. The model calculates a rancher's net return subject to technology and soil detachment and sediment yield constraints. By varying the sediment constraints and solving the model multiple times, an abatement cost curve can be estimated. A case study of the Walnut Gulch Experimental Watershed is examined in which the entire watershed is modeled as a single ranch. Results indicate little scope to significantly reduce erosion by reducing herd numbers in the short run. Although additional research is needed to quantify the effects of management on the vegetation community and sediment yields, automated methods to calculate abatement cost curves could improve rangeland water quality decision-making.

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Keywords: ranch management, erosion, sediment control, linear programming, Walnut Gulch

Introduction

As part of the Clean Water Act, states are expected to develop Total Maximum Daily Load (TMDL) plans to ensure that water bodies support designated uses. Sediment is a major water pollutant and medium for the transportation of other pollutants that impair water bodies. Development and implementation of TMDL plans is proceeding within each state.

In the west, TMDLs to address sediment are complicated by the great uncertainty about the rates of sediment movement and the associated problem of developing margins of safety, as well as issues such as assessing sediment damage to intermittent and ephemeral streams. Implementation of TMDLs generally involves the adoption of Best Management Practices (BMPs) in areas within watersheds contributing to the water quality problem. Because it is impossible to directly monitor and enforce emissions of non-point source pollutants, approaches to TMDL development at the watershed level will include encouraging landowners to participate in the planning process and the voluntary adoption of BMPs. Public land managers can cooperate by incorporating sediment reduction goals into their management plans.

If landowners are to voluntarily adopt different management systems, at a minimum landowners will expect to understand and agree with the magnitude of the changes required to achieve water quality goals. Further, in some cases, landowners will expect to receive economic incentives. On rangelands, the key issue to understand for management purposes is how management affects vegetation and how vegetation on one hand contributes to the production of beef, and hence net income, while on the other hand protecting soil and holding water on the watershed to reduce peak flows that will move sediment. This paper presents a preliminary effort to calculate the cost of constraining sediment on rangelands through vegetation management.

Methods

While soil conservation has been an ongoing effort on many ranches, the social goal of reducing sediment leaving rangelands is equivalent to an additional constraint on ranchers. A constrained optimization model mimics a profit-maximizing rancher selecting management systems that are feasible, while also meeting a defined sediment reduction requirement.

Study area

The study area is the Walnut Gulch Experimental Watershed in southeastern Arizona. The watershed headwaters are in the Dragoon Mountains and the watershed drains westward toward the San Pedro River. A summary of previous research on the watershed can be found in Renard et al. (1993). The watershed is in the USDA Natural Resources Conservation Service's Major Land Resource Area (MLRA) 41, Southeastern Arizona Basin and Range. The watershed is primarily in the Land Resource Unit (LRU) 41-3, in the 12 to 16 inch precipitation zone, except for a small area in the upper end in the 16 to 20 inch precipitation zone. There are 15 ecological sites on the watershed.

MLRA 41 is a transition zone between the Chihuahuan and Sonoran deserts, with most of the precipitation coming from summer monsoons due to convective thunderstorms, while also receiving precipitation during the winter from frontal systems. Climate is semi-arid or steppe and soils are comprised primarily of gravelly sandy loams.

Although portions of five ranches are located within Walnut Gulch, no single ranch is completely contained inside the boundaries of the watershed. To maintain privacy, we are considering an artificial ranch, the "Walnut Gulch Ranch" with boundaries that coincide with the Walnut Gulch watershed boundaries. Most, but not all of the existing fences on the watershed were mapped. To simplify the analysis to consider only what a rancher might accomplish, the major paved roads, the airport, the city of Tombstone and a mine covering 2,800 acres or 8 percent of the watershed were eliminated from consideration. Figure 1 shows the excluded area along with the pasture boundaries and the three most dominant ecological sites.

Constrained optimization model

Workman (1986) lists previous applications of constrained optimization in range management that do not consider erosion or sediment issues. The preliminary model was designed to assess the effect of short-term reductions in stocking rates on biomass under average conditions. The basic factors in the model and their relationships are shown in Figure 2. We ignore some important elements in range management such as fluctuations in precipitation and medium-term biological effects, namely the improvement in ecological condition that is possible with management. In this MLRA, such improvement is primarily reflected in an increase of perennial midgrasses. This increase in grass production is highly desirable because it results in greater and more consistent forage production for the rancher and less erosion and decreased sediment yields.

The model does consider several major processes in ranch management: vegetation production, grazing, decay, and herd management, as well as erosion and sediment delivery. The heart of the model is the production of vegetation from the NRCS ecological site guides that is converted into canopy cover and the C factor of the Universal Soil Loss Equation (USLE) as described in Wischmeier and Smith (1978). Although the application of the USLE on rangelands is problematic (Spaeth et al. 2003), because it is easy to use and understand, and there is no accepted alternative, variations of the USLE are commonly used in models run on rangelands such as SWAT and SPUR and applications such as Rangemap (Guertin et al. 1998).

As the current form of the constrained optimization model is a linear programming model, linear approximations are used to represent curvilinear relationships. The optimization model currently uses a sediment delivery ratio of 0.41 (Lane et al. 2000) to estimate the sediment yield at the outlet independent of the where in the watershed the soil eroded, or how much vegetation is on the watershed.

Model parameterization

Rangeland ecological sites are a widely used concept in management to account for a site's potential to produce similar kinds, amounts and proportions of vegetation,

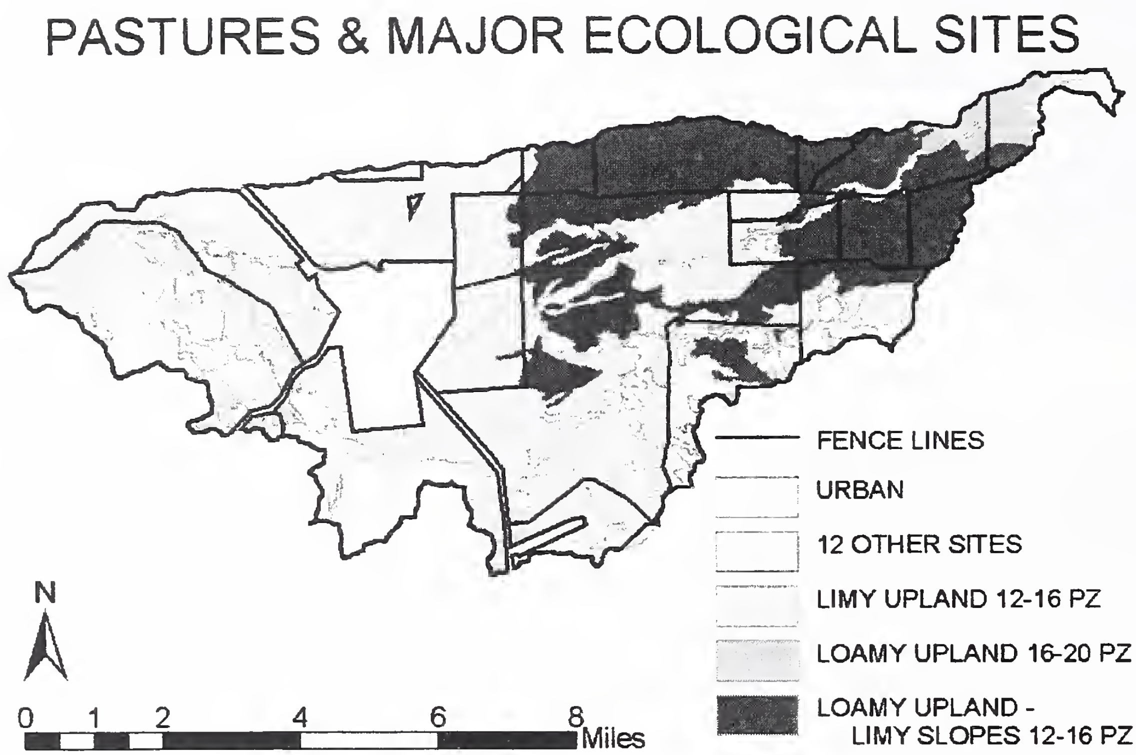

Figure 1. Walnut Gulch is a ranch consisting of pastures that are composed of various ecological sites.

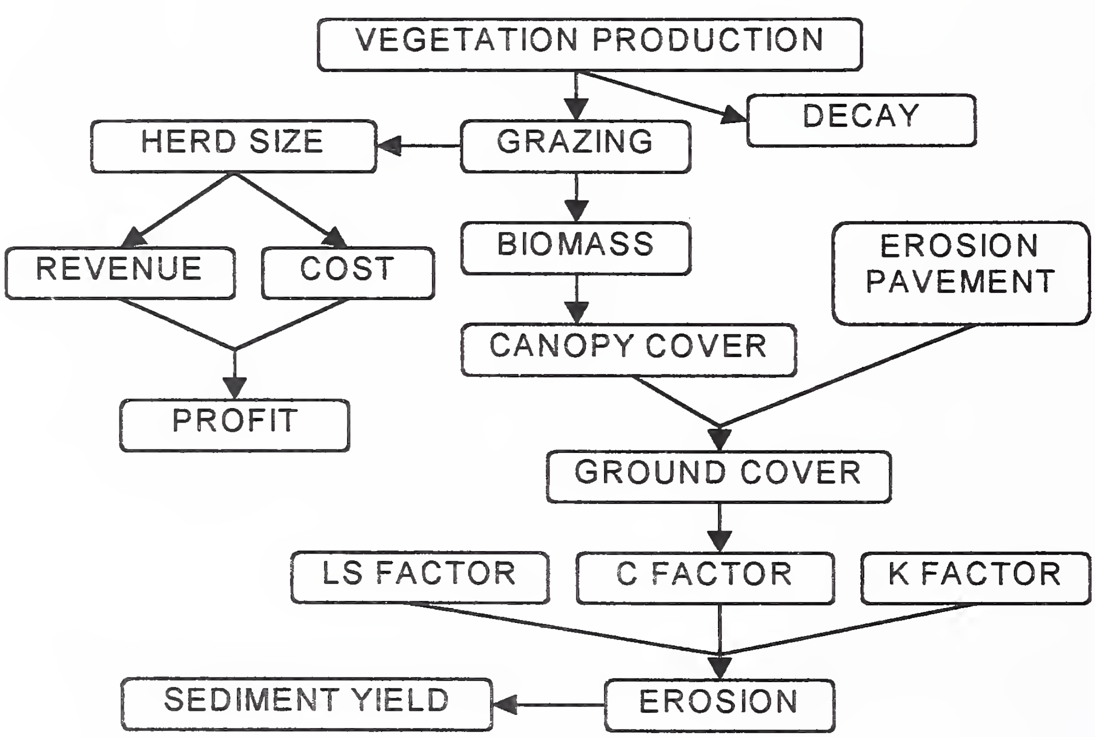

Figure 2. A number of factors contribute to the calculation of profit and sediment yield in the optimization model.

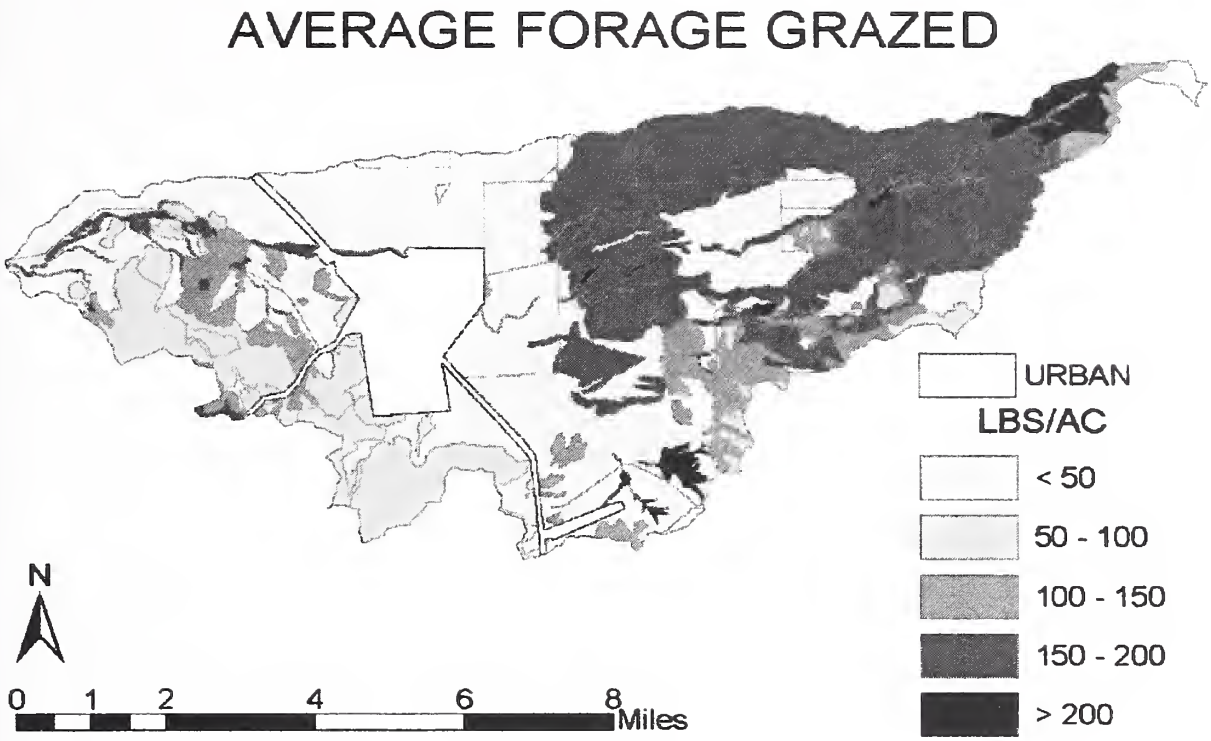

Figure 3. The optimization model distributes grazing according to forage availability.

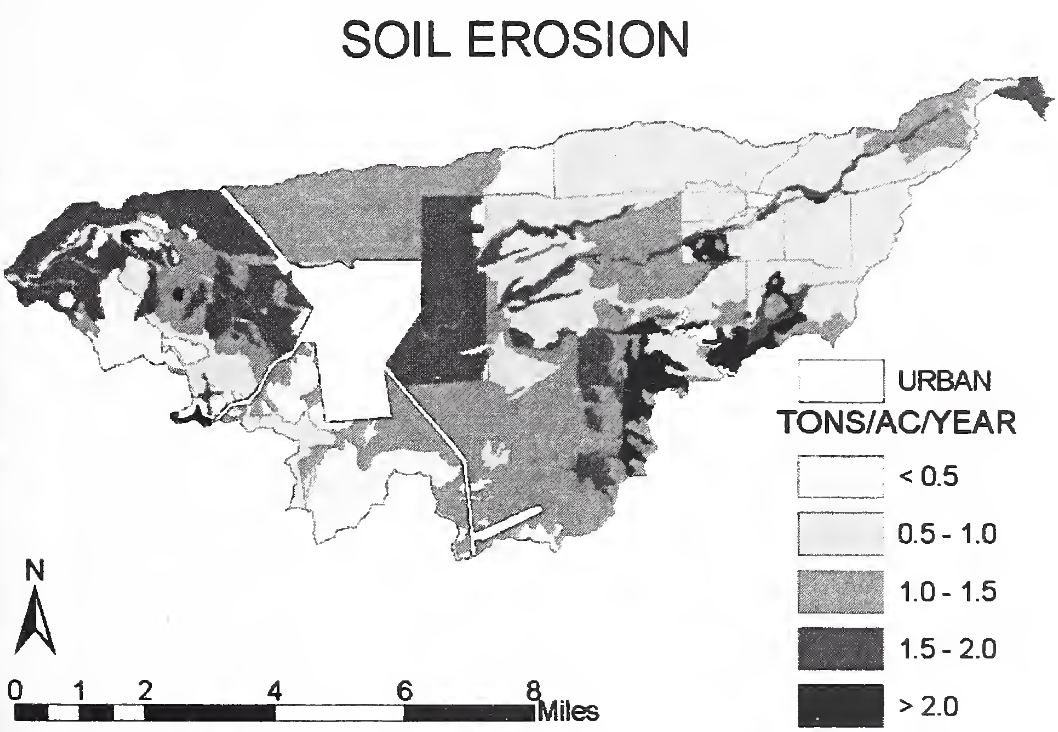

Figure 4. The USLE within the optimization model estimates soil detachment as affected by cover.

regardless of the current vegetation composition (NRCS 1997). The basic unit for this analysis is derived by overlaying pastures and ecological sites. All of the area within an ecological site in the same pasture is assumed to have uniform vegetation production and grazing intensity. The entire watershed is assumed to start in "fair" condition, which is by far the most common condition class in Arizona (Ruyle et al. 2000).

We assume the herd structure and size are adjusted while solving the model to limit grazing to a portion of the forage available. The model was solved with a utilization rate of 40% that is between the 35% used by the Forest Service and the older rule of thumb to "take half and leave half". A portion of the heifers are kept and the remainder sold along with the steers and cull cows. Prices and budgeting relationships were taken from Teegerstrom and Tronstad's (2000) Southeastern Arizona Region.

The USLE K*LS product was calculated using length and slope values from a 10 meter Digital Elevation Model of Walnut Gulch from the US Geological Survey as described in Hickey (2000). The rock fragment cover on the watershed was estimated using the slope-cover relationships developed for Walnut Gulch in Simanton and Toy (1994).

Model validation

The solution of the optimization model without a constraint on sediment should provide results that are similar to the current conditions on the watershed. The distribution of grazing according to the optimization model is shown in Figure 3. As one would expect, the Limy Upland sites did not support much grazing, while the sites to the east at higher elevations support more grazing, as do some bottomland areas to the south and west. As the production data used in the model come from the Ecological Site Descriptions, it is not surprising that there is a close match between the stocking rate from the model without sediment constraints and the safe initial stocking rates provided in the Ecological Site Descriptions, which are designed to be conservative. The area-weighted safe initial stocking rate for all 35,100 acres (55 sections) of the non urban portion of Walnut Gulch is a herd of 283 cows (approximately 5 head per section), whereas the model estimated a stocking rate of 326 cows (approximately 6 head per section).

The model estimates of the distribution of erosion across the watershed without enforcing a sediment constraint are shown in Figure 4. The estimate of erosion for the Limy Upland site is probably overestimated in part because of estimating the C factor with a linear relationship to cover which overestimates C at low values of cover. The estimate of 1.4 t/a for the Lucky Hills is well within the range of previous estimates at Lucky Hills. Simanton et al. (1980) reported actual erosion rates of 0.44, 1.78, and 0.61 t/a for subwatersheds at Lucky Hills for a 6-year period, although the two larger values are confounded by significant contributions from gully erosion. The overall average erosion rate calculated by the model on the rangeland portion of the watershed of 36,600 t/yr is about 1.1 t/a, which compares well with the 1.1 t/a reported for the whole watershed in Lane et al. (2000).

Because model erosion rates are similar to those reported in Lane et al. (2000), by applying their reported sediment delivery ratio of 0.41, the modeled sediment yield at the watershed's outlet is comparable to their estimate for overall annual sediment yield. They report an estimated mean sediment yield of 16,700 t/yr. An earlier study for a shorter period (Lane et al. 1997) reported an annual sediment yield for the watershed of 26,500 t/yr. The model sediment yield without actions to reduce sediment of 15,000 tons is reasonable given the uncertainty in sediment yields, even though the model does not consider any soil erosion that would contribute to the sediment yield coming from the town of Tombstone.

Results

If Walnut Gulch were to be managed as a ranch, it would be economically stressed if the rancher did not have an outside source of income. Table 1 shows an estimated budget for the ranch, in fair condition, before any sediment control measures are implemented, with an estimated annual loss of roughly $4,500. The budget includes depreciation and wages, so such an enterprise could be cash flow positive while the assets of the ranch deteriorate.

Constraining the average amount of sediment from the watershed in the current model formulation could be misleading because there is no effect of increased cover on the energy available to move sediment out of the watershed. Nevertheless, near the current amount of cover the results should be reasonable, so as a first step to understanding the cost of reducing erosion by cutting herd numbers, the model was solved for progressively lower amounts of sediment.

Table 1. Budget without sediment control measures.

Income	Head	$/head	Sales $
Heifers	64	323	20,538
Steers	116	304	35,117
Cull Cows	65	374	24,327
Total income	326		79,983
Costs			
Variable cost	326	101	32,988
Fixed cost			51,561
Total costs			84,549
Profit (income – costs)			-$4,566

Figure 5 shows the increasing cost to the rancher of reducing erosion by running fewer cattle. Since without a sediment constraint the hypothetical Walnut Gulch rancher was losing money, even a small sediment constraint could put the rancher out of business. Even if all the cattle were removed in the short run, only 5,000 tons, or one seventh of the total estimated annual erosion, would be reduced. If the rancher were to sell the ranch, additional calculations would be needed to assess the impact of alternative land uses on erosion and sediment yield. A more promising option would be managing the ranch to the good condition class. The model solution for good condition with no sediment constraints is a herd of 428 cows, profit of $10,200 and a reduction of 5,600 t/yr in annual watershed erosion from the fair condition case without sediment constraints.

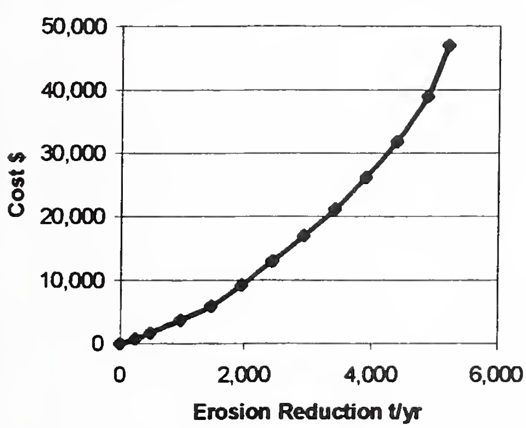

Figure 5. The cost to the rancher of reducing erosion.

In the current model formulation there is not much scope for reducing erosion simply by reducing the stocking rate. Even if the rancher could afford to reduce numbers, most of the soil protection comes from the rock cover (an average of 36% cover across the watershed), so management changes with moderate effects on vegetation will have limited effects on detachment, although there could be more significant effects on peak flows and channel processes. Evidence for this conclusion, at least for the portion of the watershed dominated by brush, can be seen in the fact that the Lucky Hills area has been fenced from grazing for more than 30 years, but still has not developed very much canopy cover (15% in May, 2003) and there are some signs of accelerated erosion.

Problems and future improvements

The study examined a simple constrained optimization model as a preliminary step in quantifying the cost of reducing sediment on rangelands. The list of improvements needed before this approach could be used with confidence is sobering in its length and complexity. From a management point of view, the scope of the management options should be widened to include facilitating practices such as additional fencing and water points to ensure a more even distribution of grazing. More important in the long-run would be the ability to alter the ecological composition of the plant community through management, although that would require a significant data collection effort and a dynamic optimization model. Installing stock ponds to act as sediment detention structures should also be an option in the model.

The second area needing significant research is the estimation of sediment yield as a function of management. A necessary first step would be the development of a historical sediment budget for Walnut Gulch. Changes to the sediment budget resulting from management could then be calculated using a distributed, continuous watershed model that would have to simulate the dominant channel processes and support the management practices mentioned above. The model would be calibrated using observed data from other locations and rainfall simulation experiments and simulation results would be used to define the relationships inside a revised optimization model. Issues such as urbanization and the uncertainty associated with precipitation should also be considered.

Conclusions

A method to estimate the cost of reducing erosion on a hypothetical ranch in southeastern Arizona was presented. On Walnut Gulch there appears to be little scope for significant reduction in erosion from reducing the stocking rate in the short term. This approach needs a number of improvements, nevertheless the approach provides a framework for assessing the cost to ranchers of reducing erosion, and ultimately sediment. The approach could provide better information on where changes are required, how large the changes need to be, and how much the changes will cost the rancher when developing TMDL plans on rangelands.

Acknowledgments

The authors appreciate the reviews of Leonard Lane and Dale Fox.

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Spatial Decision Support System for Identifying Priority Sites for Watershed Management Schemes

Jagarlapudi Adinarayana

Abstract

In the context of rural India, planning mainly consists of distribution of resources through various government-sponsored watershed management schemes, which are implemented by sectoral institutions in the district. A conceptual framework and a spatial decision support system for rural land use planning (SDSS/LUP) have been developed for supporting decision making on area selection for different watershed management schemes for conservation planning. SDSS/LUP is also intended to provide suggestions and hazard warnings for land use sustainability by combining data from the existing sources. The paper also discusses future national level Web decision support system for rural land use planning.

Keywords: conservation planning, geographical information systems, remote sensing, spatial decision support system, watershed management schemes

Introduction

Rural land use planning in India mainly employs prescriptive planning on a watershed basis through various mandated schemes, which are financed by the state or central government and implemented by sectoral institutions (top-down). Each scheme has a set of policies that are defined by the legislation. Under these watershed management schemes, planners at the district/sub-district level need to identify priority watershed/sub-watersheds for preferential treatments/land use plans.

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The Natural Resources Data Management System (NRDMS) of the Department of Science & Technology, Government of India is a multi-disciplinary program that is developing decision support systems (DSS) for decentralized planning using GIS technology. Its immediate clients are district-level planners and professional staff of line departments engaged in rural development activities. Our attempt, as a part of the NRDMS Program, in the development of spatial decision support system (SDSS) is to draw together the natural resources data of sectoral agencies, process them to computer-compatible format and build up a database for watershed planning in an integral manner.

Design Principles

What decisions are to be supported?

At the outset, a needs assessment was carried out amongst district-level staff to establish their requirements for spatial data. It proved difficult for them to articulate their needs but a number of specific requirements emerged from these discussions:

Decision type A: Area selection for schemes

A.1 Which are priority watersheds for interventions by various line departments?

A.2 Within a priority watershed, which sub-watersheds should be treated first?

A.3 Where, within a sub-watershed, are the hot-spots requiring interventions?

Decision type B: Site selection for infrastructure

B.1 Where should small-scale conservation infrastructure be built?

B.2 Where should water resources infrastructure be built or authorized?

Decision C: Land evaluation for changes in land use

C.1 Economic options
C.2 Conservation options
C.3 Implemented options
C.4 Radical options

How are these decisions actually made?

In the first instance, emphasis is given on selecting priority sites for conservation planning which will help the decision-makers to do their job more easily, accurately and consistently.

Each scheme is bounded by government policies, which have social, economic and biophysical dimensions. Policy is enshrined in the directives that establish the scheme and these commonly lay down criteria for site selection. For example, the National Watershed Development Program for Rainfed Agriculture (NWDPRA) lays down four criteria: < 30 % area is irrigated; < 750 mm average rainfall; no other scheme have been implemented; and size of a watershed is between 5000 and 10000 ha. But site selection according to these criteria is not so straightforward as it might appear. The physical criteria actually reflect the political intention to benefit the disadvantaged. Also, the concept of a watershed is not strictly hydrological; there are no standard maps of watersheds or any standard method for delineating them.

The SDSS approaches the problem stepwise. First, a topographical base is generated within the Geographical Information Systems (GIS) from 1:50000 scale Survey of India topographic sheets. The boundaries of hydrological watersheds are drawn manually to produce watersheds of the required size and these boundaries are digitized within the GIS.

Within any watershed there will be some 20 sub-watersheds, each of about 500 ha, that may be considered the primary planning units because they are small enough for concerted intervention. They must be ranked in order of priority for intervention and in the absence of the SDSS, local field staff will recommend sub-watersheds – preferring those with a range of land types, so that many rural departments can be involved and those with the most severe physical and social problems.

The SDSS adopts explicit criteria: degree to which sub-watershed satisfies the objectives of a particular scheme; actual problems with productivity due to erosion or other degradation processes (on-site effects); sediment delivery to reservoirs (off-site effects); extent of degraded land; and multi-criteria evaluation based on all of the above.

Using NDWPRA selection criteria

The sub-watersheds are prioritized according to decision rules formulated by NWDPRA. If the sub-watershed does not meet the minimum criteria (i.e. > 30 % irrigated area) it is rejected. Otherwise, the same criteria are used to rank sub-watersheds so that the less the proportion of irrigated land, the higher the priority. Interventions in the highest-priority sub-watershed should go furthest in satisfying policy objectives.

By erosion intensity

The sub-watersheds are prioritized according to the modeled intensity of erosion under present land use and management. Intervention in the highest-priority sub-watershed should result in the maximum reduction in erosion.

The area of a sub-watershed is not considered. Instead, the decision-maker chooses sub-watersheds in order of their erosion intensity until the maximum area that can be treated is reached.

Identification of hot-spots

Once a sub-watershed is identified for intervention, the question arises as to which areas within the sub-watershed are most critical. The decision is supported by modeled erosion intensity for the individual polygons of the GIS. The SDSS presents a map of the sub-watershed with the more critical areas, overlain by the road and stream network and contours to help identify these areas on the ground.

By sediment yield

Sub-watersheds are prioritized according to the modeled sediment delivery to watercourses or reservoirs. Intervention in highest-priority sub-watershed should result in the maximum decline in sedimentation.

By present land degradation status

Sub-watersheds are prioritized according to their proportion of degraded land, which are identified with the help of Indian Remote Sensing Satellite (IRS)-IC/ID remotely sensed data. Intervention in the most degraded sub-watershed should result in the maximum proportional land reclamation but this is by no means the same thing as optimum return for the effort expended.

Multi-criteria evaluation

Each single criterion evaluation provides a ranking of the sub-watersheds. In the case of the NWDPRA selection criteria, the evaluation may also reject sub-watersheds that do not meet the scheme's criteria. These individual rankings may be combined by a variety of methods, e.g. using DEFINITE software (Jassen and Herwijnen 1992).

The SDSS also greatly reduces the labor of following the more simple procedures and produce quantitative results, presented as a matrix of calculated or ranked values to assist decision-makers (Table 1). Ranked social and economic characteristics may also be included.

Table 1. Scenarios/Multi-criteria evaluation and priority ratings.

NWDPRA Watershed	Priority rating	
	Physical basis	Socio-economic . basis
Bairasagara	1	2
Chalamena Halli	4	3
Chonduru	3	1
Peresandra	2	4

Ramapatna (RP) Sub-watershed	Priority rating of physical characteristics		
	Soil erosion intensity	Sedimen t yield index	Extent of degraded lands
RP_East	1	2	3
RP_North	2	3	1
RP_West	3	1	2

Test Area and User Interface

In order to test and validate the concepts of SDSS, two blocks (Chikballapur and Gudibanda) in the southern part of India were selected for identifying the priority watersheds for the NWDPRA scheme. For sub-watershed prioritization, Ramapatna watershed was selected.

The SDSS must present a simple, intuitive interface to the user, who is presumed to be somewhat a novice to the use of computers and GIS. The interface displays a series of maps, complete with symbols and attributes, called themes. The user chooses which scheme to view by highlighting a check box. Simple tools enable the user to zoom in and out, and to obtain information on map areas and other details specific to each map delineation.

A series of views present: (1) input maps and tables, (2) derived maps, and (3) ratings that can be called upon by the user. The system manager, not the district level planners or other professional staff does the calculations of ratings, except in the case where the district staff can change the input maps or tables. It might be argued that this approach excludes the actual land users, simply perpetuating the traditional top-down planning ideology. But it should result in better-informed district-level staff who can bring new perspectives to their continual discussions with the land users.

Land Use Sustainability Assessment

SDSS has to provide a consistent level of information and analysis based on data that are actually available now, nation-wide, and this information has to be useful to today's decision makers without the need for specialist training. At present, the options are limited by the availability and scale of fundamental data. The problem of inadequate data is being handled by a rough land use sustainability assessment using the data that exists (Adinarayana et al. 2000).

To provide an immediate and useful service to the decision makers, physical hazards have been identified: for instance drought, soil erosion, or excessive percolation under irrigation. Then, indicators of these hazards for which the information can be obtained have been sought. These indicators, or limitations, have been ranked in order of the ease of obtaining data; and the subtractive approach described by Shaxson (1981) as land unsuitability has been applied.

The procedure has been programmed to successfully de-rate any parcel of land under consideration according to the severity of the limitations; arriving at a six fold classification comparable to the well known land capability classification (Klingebiel and Montgomery 1961) but with additional loops to accommodate rice and irrigated land. The defining values of each class are locally calibrated and the result is expressed with up to three degrees of confidence, depending upon the completeness and quality of data used in the assessment, for example "not better than Class C – with one degree of confidence <C (1)."

On the basis of the identification of hazards, district staff can design management packages to combat the threats to the sustainability of the desired land use, or recommend an alternative land use. In short, we are applying the threat identification and management concept outlined by Smith et al. (1999).

Conclusions and Future Developments

NRDMS centers are being set up in various districts in 10 Indian states. The decision support service that can be provided as of now meets the requirements specified during the needs assessment. As far as land evaluation is concerned, it is unsophisticated but it is robust and functions with the data that are actually available in every district in India.

With a better database we can provide a better service, and a program to upgrade the database should begin with addition of the 10 m contour data (which are already held by the Survey of India) to the digitized topographic sheets. Good use can also be made of the more recent, high definition satellite imagery. A cut-down version of ALES (Rossiter and Van Wambeke 1997) can be added to the SDSS to provide a framework for land suitability evaluation for specific crops/land use types, including basic financial/economic analysis.

SDSS/LUP addresses a single district only and is not sufficiently robust for the entire agricultural extension community to use for decision-making. Hence, it is proposed to develop a new Web based decision support system (DSS) for assisting the extension community in rural land use planning decision making.

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Management of Upper Lake Watershed

Pradip Kumar Nandi

Abstract

The Upper Lake of Bhopal (Latitude 23°12' to 23°16' N and Longitude 77°18' to 77°23' E), constructed in the 11th century AD, is rich in biodiversity and is a major source of potable water for the people of Bhopal having a population of 1.6 million. The lake is also a lifeline for farmers of the fringe area and for about 500 fishermen families. Hence protecting this valuable resource from pollution for sustainable water use is of great importance as a matter of policy. However, in practice management of this resource had hardly been a top priority, which has resulted in the deterioration of lake water quality consequent to rapid urban development in the watershed during later part of the last century. The infamous Bhopal gas tragedy in 1984, however, generated considerable environmental consciousness among the people and protection of this vital but vulnerable resource for sustainable use was started in 1995. The major objective of this conservation effort, being improvement of water quality, involved prevention of inflow of wastes and anthropogenic activities within and outside the lake area, offloading of accumulated nutrients through desilting, weed removal and aquaculture and promotion of wise use of water resource through mass awareness campaign. Integrated shoreline management thus involved providing a sewerage system in the urban watershed, creating a buffer zone between human settlement and the lake in the form of a road and a promenade in the urban fringe and a green belt of local species all along the lake periphery. There is a legal framework in place to control anthropogenic activity up to 50 meters of shoreline of the lake. Regular water quality monitoring revealed a positive impact of the conservation efforts. However, to sustain the efforts made so far a long-term management plan with active participation of all stakeholders are required.

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Keywords: Upper Lake, urban development, pollution, water quality, management plan

Introduction

The construction of storage reservoirs is an age old practice in India. Former rulers have contributed significantly by constructing large number of impoundments for providing drinking water to the people in their capitals and elsewhere. This was particularly necessary in arid, semi arid and other regions with highly erratic rainfall. Upper Lake of Bhopal, arguably the oldest among the large man-made lakes in central part of India, falls under this category. This Lake was created in the early 11th century AD by obstructing the natural flow of the Kolans, a rainfed tributary of Betwa river by constructing an earthen dam across the valley between the two hills now known as Idgah and Shamla hills.

With the passage of time, the administration of Bhopal City changed hands several times, and in the year 1956 it became the capital of State of Madhya Pradesh, the then largest state of India. Since then it noticed tremendous influx of people and consequent urban development especially on the northeast fringe of the Upper Lake. This caused increase in demand for potable water and thus pressure on the lake. Consequently the storage capacity of the Upper Lake had to be increased through raising the height of spillway. Though this had helped in easing the water supply, the increased anthropogenic activities in the watershed caused increased inflow of silt, untreated sewage, nutrients and pesticides from urban and rural areas and thus deterioration of water quality of the lakes. However, in view of its ecological importance, the Ministry of Environment and Forests, Government of India has recognized this lake along with another lake (Lower lake) located downstream of Upper Lake as wetland of national importance and designated them as Bhoj Wetland in 1998, and action for the conservation of Upper Lake was started in 1989 with emphasis on creation of a buffer zone of plantations between the lake and the human settlements. Later an integrated plan for the conservation and management of the lakes was conceived, and implementation of the same was started in 1995 through financial assistance of Japan Bank for International Corporation of Japan. Though the implementation of the project is still underway, its positive impacts are perceptible. This lake was declared as a Ramsar site in the year 2002.

Location of the Lake

The Upper Lake is located between latitude 23°12' - 23°16' N and longitude 77°18' - 77°23' E. It is a shallow tropical lake. It has a watershed area of 361 km^2 and a maximum submergence area of about 37 km^2. The attainment of maximum water level (508.04 meters above sea level) of the lake depends on the magnitude of monsoon rainfall (average being around 1150 mm) in the watershed area. The water level in the lake is maintained by discharging excess water through a spillway provided on the southern bank of the lake.

Watershed Area

The watershed area of the lake displays a complete range of urban and rural activity with varying intensities. The land use pattern of the Upper Lake watershed area is shown in Table 1. The soil of major part of lakebed and the watershed of Upper Lake is loamy. The soil is black in color and is derived from trap by the process of weathering.

Table 1. Upper Lake watershed land use.

Built Up Area	20.855 km^2
Crop Land	219.050 km^2
Plantation	9.600 km^2
Open Forest	5.225 km^2
Land with Scrub or Without Scrub	90.292 km^2
Barren Rocky/Stony	8.465 km^2
Other Lakes/Ponds	16.175 km^2
Total Watershed Area	361.000 km^2

Importance of the Lake

The lake is the principal source of potable water to the city of 1.4 million people. In case of normal rainfall the water supply from the lake is 29 million gallons per day (MGD). Otherwise, off take may be much less (i.e. 12 to 15 MGD) depending upon the initial water level after the monsoon.

The lake is also rich in bio-diversity, principal components being phytoplankton, zooplankton, macrophytes, aquatic insects and avifauna (both resident and migratory). The details of biodiversity of the lake is as follows:

- Macrophytes - 106 species belonging to 87 genera of 46 families, including 14 rare species.
- Phytoplankton - 208 species comprised of 106 species of Chlorophyceae, 37 species of Cyanophyceae, 34 species of Euglenophyceae, 27 species of Bacilariophyceae, and 4 species of Dinophyceae.
- Zooplankton - 105 species (41 Rotifera, 10 Protozoa, 14 Cladocera, 5 Copepoda, 9 Ostracoda, 11 Coleoptera, and 25 Diptera).
- Fish fauna - 43 species of natural and cultured species.
- Avifauna - 179 species (52 migratory, 28 local migratory, and 99 local).
- Insects - 98 species of 10 families.
- Reptiles and Amphibians - >10 species, including 5 species of tortoise.

Furthermore, a buffer zone along the fringe of the wetland has been created through plantation of 51 species of angiosperms with adequate representation of medicinal plants.

Problems

Due to tremendous population growth of the city (from just over 0.1 million in 1951 to about 1.6 million in 2001) and rapid urban development on the eastern and northern fringes of Upper Lake especially during the second half of the 20th century subjected the lake to various environmental problems resulting in deterioration of its water quality. The major causes of environmental problems of the lake are listed in Table 2.

Table 2. Environmental concerns affecting Upper Lake.

Problem	Causes
Reduction of storage capacity of the lake	Inflow of monsoon runoff and dry weather flow carrying silt and organic materials from urban and rural watershed. Addition of clay and bio and non-biodegradable materials through Idol immersion.
Obstruction to smooth flow through the spill channel of the lake	Deposition of silt.
Flourishing growth of invasive aquatic plants	Nutrient enrichment of lakes due to inflow of sewage and agricultural wastes.
Deterioration of water quality	Inflow of untreated sewage from the urban watershed. Dumping of municipal wastes not collected by the municipal corporation. Addition of organic and inorganic materials through Idol and Tazia immersion (due to religious activity of Hindu and Muslim communities). Direct human intervention and encroachment of fringe areas. Addition of detergents used for washing clothes.

Interventions in the Watershed for the Conservation of the Lake and to Improve its Water Quality

Interventions in the watershed are mainly preventive in nature and include the following:

1) **Demarcation of no construction zone:** The Bhopal Development Plan 2005 prohibits construction within 50 m of the Full Tank Level (FTL) of Upper Lake. Accordingly, a 50 m wide strip of land all along the FTL of the lake was demarcated and the document was provided to the concerned authorities (Bhopal Municipal Corporation and Directorate of Town & Country Planning) to take appropriate action against encroachment.

2) **Creation of buffer zones between the lake and the human settlements:** A 5.4 km Link road on the north-east and a 2.5 km long Lake View Promenade on the south east fringe of the lake were constructed which served the dual purpose of prevention of encroachment of lake fringes, being a physical barrier as well as reduction of traffic pressure through the city. The promenade became a recreational and relaxing site for the city dwellers.

3) **Buffer zone plantations:** In order to prevent encroachment for human settlements and cultivation and grazing within the lake area, buffer zones have been created particularly in the Western, Southern and Northern fringe of Upper Lake. Besides this, intensive plantation has been carried out in the watershed area of the lake to control soil erosion. The species selected are either biomass producing or having medicinal properties and are tolerant to both flooding and drought conditions. About 1.7 million plants have been planted in over 10 km^2 land for over a period of 12 years. In the program under social forestry, farmers were encouraged to raise fruit yielding trees along their crop fields and marginal lands. The results were quite encouraging.

4) **Watershed treatment:** In order to mitigate inflow of silt, agricultural residues and other wastes into the lakes, 73 check dams made of loose boulder/Gabion structures having a cumulative silt trapping capacity of about 0.35 million cum have been constructed across 28 inlet channels.

5) **Sewerage system:** Infrastructure (laying of 86.7 km pipeline through congested human settlements and construction of 8 sewage pump houses and 4 treatment plants) for diversion and treatment of 35 MLD domestic sewage from the lake is being developed.

6) **Solid waste management:** Infrastructure of Bhopal Municipal Corporation was strengthened through providing additional equipment and vehicles. Outcome being

additional collection and disposal of 70 MT of solid waste from the 18 municipal wards located in the urban watershed of the lake.

7) **Promotion of organic farming:** Intenssive cropping with use of inorganic fertilizers is being done in the in the rural watershed . Through monsoon run off, a significant part of these nutrients find their way into the lake, causing growth of aquatic vegetation in the lake. With a view to discourage the practice of inorganic fertilizers based intensive farming in the watershed, a promotion drive for the use of organic fertilizer produced by the farmers themselves using farm waste and cow dung was launched. These include hand on training to the farmers for making high quality compost using bacterial inoculum. The farmers find the method quite acceptable since crop yield was high compared to the use of compost produced by them through conventional method and there was considerable saving.

Stakeholder Participation

The participation of ordinary people in the project implementation is an essential and important feature. This has been achieved through a well coordinated awareness program involving political and religious leaders, district/city administration, local people, NGOs, schools, etc. The important stakeholders' participation in watershed management activities is as follows:

- Reforestation of watershed area through participation of farmers.
- Promotion of organic farming in the watershed through participation of farmers.

Impacts of Implemented Action

- Due to construction of silt traps across small feeding channels and creation of buffer zone plantation around the lake sedimentation of the lake was considerably reduced. The silt traps were also found to be very effective in trapping the organic debris from the rural watershed.
- Before creation of buffer zone plantation, land was either used for agriculture or infested with shoreline weeds. Either of the situations was not desirable. After plantation and protection, not only general ambience of the area improved but due appearance of ground flora soil erosion from the area become negligible. The area also becomes a major attraction for the avifauna. An avifauna census completed in the year 2001 has revealed 179 species, which include 52 migratory, 28 local migratory and 99 local avifauna species. Some of them are also arboreal reflecting positive impact of plantation on biodiversity. Increase in population of *Grus antigone*, a vulnerable species, has also been reported.
- In the plantation area the understory grass species become a regular source of cattle feed for the nearby villages. The plantation and protection activities also provided ample job opportunities for the villagers, of whom a significant number were of women. For the development of nurseries local women were involved wherever necessary.
- The Link road not only becomes a barrier against encroachment of Lake Area, but it helped in easing traffic pressure through the city.
- The project implementation has provided many job opportunities for the local people.
- Due to reduction of inflow of solid waste and silt and reduction of direct intervention of the people, no significant deterioration of water quality was observed. After the commissioning of sewerage system, further improvement in water quality of the lake is expected.

Lessons Learned

The people's participation for the success of Lake Conservation program is a must.

Future Action Plan

These include the following:

- Constitution of Lake Conservation Authority to sustain the conservation efforts made under the project.
- Declaration of buffer zone plantation area as well as the southern transitional zone of the Upper Lake as a bird sanctuary, a first in state of Madhya Prudish, to provide long-term protection to the plantations raised and to promote eco-tourism.

Watershed Management – Policies and Practices in a Humid Tropical Region

K. Shadananan Nair

Abstract

In many tropical regions, increasing population and associated water needs in domestic, industrial and agricultural sectors exerts tremendous pressure on the watersheds. Even in water rich regions in the humid tropics like the State of Kerala in India, this affects the economy and day-to-day life. The State receives three times the global average precipitation and there exist a number of rivers, making even the demarcation of watershed difficult. But, more than 80% of the water in the rivers wastefully flows into the Arabian Sea. Kerala experiences serious seasonal water shortages, as a combined result of improper management, careless use, land and water resources degradation and heavy population. There are several rules and regulations for the protection of watersheds, with provision for optimum utilization of resource. Failure in its implementation often poses threat to the watersheds' environment. Land and water degradation has become a major issue here. The State administration has recently formulated certain development programs at various levels in the watersheds, incorporating modern and traditional methods. This paper is a detailed investigation on the water related problems in the watersheds of Kerala and on the various projects planned as part of a water policy for the watersheds' protection and management. Drawbacks in the implementation of existing laws and its effect on the watersheds' environment have been examined. Assessment of the water resources and possible impacts of a changing environment on them has been made, for providing guidelines for better management.

Keywords: Kerala, watershed management, water policy, changing environment

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Erosion III

Soil Contributions to Sediment Properties in Walnut Gulch Experimental Watershed: Influence of Slope Factors

F.E. Rhoton, W.E. Emmerich, D.C. Goodrich, D.S. McChesney, S. Miller

Abstract

Variations in soil profile thickness, surface soil properties, erosion rates, runoff, and sediment properties within similar soil types and watersheds can generally be explained by slope factors that influence soil erodibility. This study was conducted to determine the effects of surface morphometry on the distribution of watershed soil properties that control erodibility and sediment properties. Each major soil type in six sub-watersheds in Walnut Gulch Experimental Watershed was sampled intensively along transects positioned to represent the normal landscape features associated with a particular mapping unit. At each sampling point, data were recorded for latitude-longitude, slope gradient, slope position, and slope aspect. Suspended and bedload sediment samples were collected from flumes located at the mouth of each sub-watershed. Clay contents of the soils and sediments ranged from 125.0 to 152.7 g kg^{-1} with averages of 136.8 and 178.1 g kg^{-1}, respectively. Enrichment ratios (ER) calculated for each watershed indicated that suspended sediments were enriched in clay, relative to the soils, by a factor that ranged from 1.02 to 1.68. The aggregation index (AI), a measure of relative erodibility, ranged from 18.0 to 31.9. The correlation coefficient (r) determined for ER vs. AI was - 0.927 ($P \leq 0.05$). The data indicate that watersheds with the lowest AI are producing the greatest amount of suspended sediment. The data also indicate that the highest soil AI values occur on summit, shoulder, footslope and toeslope positions, on slopes steeper than 13%, and on NW-, N-, and NE-facing slopes. These results indicate that this approach could be used to improve our understanding of hillslope erosion processes, and the accuracy of erosion prediction models.

Keywords: erodibility, aggregation, carbon, clay

Introduction

The severity of erosion is largely determined by factors such as rainfall characteristics, topography, and vegetative cover. If these factors remain constant and the soil resource changes, variations in erosion losses can be attributed to variations in soil properties that influence soil erodibility (Bryan 1969). Basically, soil erodibility is determined by aggregate stability by virtue of its control over porosity and infiltration rates. Soil aggregates of low stability are dispersed by relatively low rainfall energies leading to surface sealing, increased runoff, and a high proportion of easily transported fine particles. Generally, the level of aggregate stability depends on the content of bonding agents in the soil such as clay, Fe and Al oxides, and organic C which have the ability to bind soil particles into stable units. Generally, the soil clay fraction serves as the building block for aggregate stability. Thus, the degree to which clay particles in soil aggregates disperse in water can be taken as a measure of aggregate stability.

The distribution of soil properties that influence aggregate stability in the landscape can vary as a function of slope position, slope class, and slope aspect. Franzmeier et al. (1969) found greater organic C contents and darker soil colors on north-facing slopes associated with lower temperatures and higher water contents. They also measured coarser particle size distributions on mid-slope positions, and higher concentrations of basic cations on the lower slope positions. Similarly, Hanna et al. (1982) indicated that north-facing slopes contained 20% more available water relative to south-facing slopes, and that soils on

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east-facing slopes were the driest. In terms of slope position, soils on backslope and footslope positions contained more available water than summit and shoulder positions. Rhoton et al. (1998) related water dispersible clay contents to the distribution of Fe oxides among soil drainage classes. Water dispersible clay contents and soil erodibility were at a minimum on the lower, wetter slope positions which favored formation of poorly crystalline Fe oxide phases most influential in aggregate stability.

This research was conducted on the basis of these previous findings, which have demonstrated that numerous soil properties that influence soil erodibility vary in magnitude with changes in surface morphometry. Our objectives were to characterize the relationships between soil erodibility and slope factors within a large semiarid watershed.

Methods

Site characteristics

The research was conducted on the Walnut Gulch Experiment Watershed (Figure 1) near the town of Tombstone in southeastern Arizona (31 deg. 43 min. N. Lat., 110 deg. 41 min. W. long). The watershed encompasses 150 km^2 with elevations ranging from 1220 to 1890 m. In terms of geology, the watershed is located primarily in a high foothill alluvial fan portion of the larger San Pedro River Watershed. The alluvium is primarily composed of Cenozoic age clastic clays and silts. The remaining mountainous portion of the watershed consists of limestone, weathered granite, and igneous intrusions ranging in age from pre-Cambrian to Quaternary. The mean annual temperature at Tombstone is 17.6 °C, and the average total annual precipitation is 324 mm (Renard et al. 1993).

Study approach

Six sub-watersheds were selected for study within the larger watershed. Each of these sub-watersheds was instrumented with a supercritical flume (Renard et al. 1993) and associated flow measuring equipment. Inlet drop boxes constructed in the floor of the flume were used for the collection of bedload samples once the sediment passed through the 6.4 mm slotted covers. Suspended sediments were collected with vertical samplers mounted on the face of the flume. As with bedload sediment, only the < 6.4 mm fraction was collected.

Watershed soils were sampled on the basis of acreage comprised by individual soil mapping units within a given sub-watershed. Initially, digitized soil surveys were superimposed on digital elevation models of each sub-watershed, then a sampling transect length of 1000 m was chosen for each 200 ha occupied by a given soil mapping unit. Transects were positioned to insure a maximum number of slope factors were represented by the samples. Soil samples were collected along the transects with each change in slope position. At each point, soil samples were collected from the surface 5.0 cm at three locations, perpendicular to the slope, approximately 10 m apart. These were composited to form a single sample, sieved to < 4 mm and sealed in a plastic bag. At each sample point the following data were recorded: latitude – longitude, slope position, slope aspect, and slope gradient.

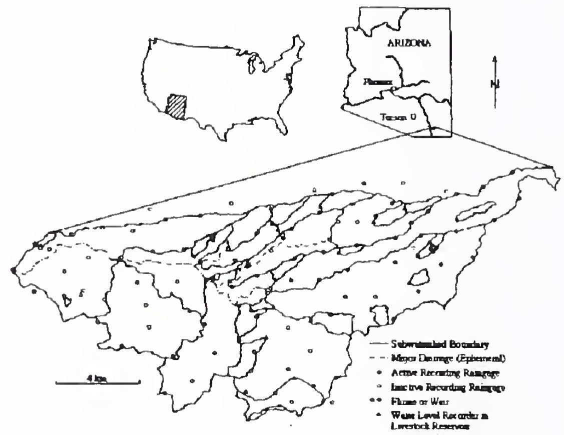

Figure 1. Location of the Walnut Gulch Experimental Watershed.

Laboratory analyses

Prior to analysis, all soil and sediment samples were air-dried and further sieved to < 2 mm. Particle size distribution was determined by standard pipette methods after the samples had been shaken overnight in Na hexametaphosphate (Soil Survey Staff 1984). Water dispersible clay (WDC) was estimated by the same methods, except only distilled water was used during the dispersion phase. Soil pH was measured in a 1:1 soil/distilled water (v-v) suspension (McLean 1982). Total C was determined by combusting a 0.5 g sample in a Leco CN-2000 carbon-nitrogen analyzer. Inorganic C contents were determined by treating a separate 1 g sample with 5N HC1 in a sealed decomposition vessel fitted with a rubber septum. Carbon dioxide pressure generated by the acid-decomposition of the sample was measured with a

Tensi-meter probe inserted through the septum. The pressure readings were converted to C contents and subtracted from the total to give organic C content. Quantitative soil color was measured with a Minolta Chroma Meter. Magnetic susceptibility was determined with a Bartington MS-2 Magnetic Susceptibility Meter. All sediment samples were analyzed identically to the soils.

The data obtained for total clay and WDC contents were used to calculate an aggregation index (AI) for the watershed soils based on the method of Harris (1971) as follows:

$$100\left(1-\frac{\text{WDC}}{\text{total clay}}\right).$$

Results

The data for selected soil properties are shown as an average for the six sub-watersheds (Table 1). Based on these results, some soil property data reflect differences in parent material composition between watersheds. The most obvious differences exist between watershed 7 (W7) and the other watersheds in terms of total clay contents, organic C, AI, and magnetic susceptibility (MS). A significant percentage of the soils in W7 were formed from igneous residuum (i.e., granite, granodiorite), which is more resistant to weathering processes relative to the limestone parent materials in other watersheds. Consequently, W7 soils are expected to have less clay and organic C, and a lower AI value, as is the case. Conversely, W7 soils had the highest MS that probably reflects a higher magnetite content in the igneous parent rocks. The higher overall soil color readings are due to the lighter colored granitic rocks and lower organic C contents. By contrast, W9 had a substantial acreage of soils formed from the weathering of fine grained, igneous rocks (i.e., andesite, basalt) that are composed of more weatherable minerals. Such parent materials should produce soils with finer particle sizes. The soils in W9 had the highest total clay, organic C, and AI, as well as the darker (redder) soil colors (hue, value). Clearly, parent material influenced soil erodibility in the Walnut Gulch Experimental Watershed.

Particle size (clay contents) and organic C data are compared for the watershed soils, suspended sediment, and bedload sediment in Table 2. The watershed soil data represent a weighted average calculated by multiplying the relative acreage occupied by a given soil type in a sub-watershed times the value for a specific soil property. The values were summed for all soil types to give a weighted average for the watershed. Generally, both the total clay and organic C data followed the same relative trend between watersheds as did the simple averages in Table 1. Watersheds 9 and 10 had the highest total clay contents on a weighted average basis followed by W11 and W15 with similar concentrations, while W3 and W7 had the lowest

Table 1. Selected soil physical and chemical properties averaged for individual watersheds.

Sub-watershed	n	Clay content: Total	Clay content: Water dispersible	Carbon content: Total	Carbon content: Organic	Aggregation index	pH	Magnetic suscep.	Munsell color[2]: Hue	Munsell color[2]: Value	Munsell color[2]: Chroma
		$g\,kg^{-1}$						$10^{-8}m^3\,kg^{-1}$			
3	81	132.5 b[1]	108.3 abc	23.2 b	10.7 bc	18.0 d	8.6 a	197.9 cd	7.1 b	3.1 b	1.9 b
7	49	118.2 c	91.1 d	18.6 c	8.5 d	22.8 c	7.9 b	799.7 a	8.2 a	3.3 a	2.0 ab
9	115	163.4 a	111.2 ab	19.3 c	12.1 b	31.9 a	7.4 c	294.4 b	6.5 d	2.9 c	1.7 d
10	74	159.6 a	116.2 a	16.4 c	11.5 bc	28.1 b	6.9 d	189.4 d	6.4 d	3.0 c	1.8 cd
11	47	133.6 b	101.7 bcd	26.8 a	11.8 b	23.9 c	8.5 a	264.4 bc	6.8 c	3.1 b	1.5 e
15	92	140.4 b	98.4 cd	29.2 a	14.2 a	28.2 b	7.9 b	216.9 cd	6.9 c	3.1 b	1.8 c

[1] Values followed by same letter are not statistically different at $P < 0.20$ based on Duncan's new multiple range test. [2] All Munsell colors are from wet samples. The hues are YR (yellow red).

concentrations. Organic C contents of the soils were more uniform with the highest contents recorded for W15 (12.9 g kg^{-1}) and the lowest for W7 (9.6 g kg^{-1}). The total clay contents of the suspended and bedload sediments (Table 2) in combination with soil AI were used as an indicator of relative erosion losses from the watersheds. The organic C data are included for purposes of explaining potential differences in AI and suspended sediment concentrations. Relative to the soils, the clay contents of the suspended sediments were enriched by an average factor of 1.38. The greatest enrichment (1.68) occurred in W3, and the smallest (1.02) was in W9. These two watersheds had the highest and lowest AI, respectively (Table 1). The ratios of suspended and bedload sediment clay to soil clay were correlated against the soil AI for individual watersheds. This gave a correlation coefficient (r) of -0.927 ($P \leq 0.05$) for suspended sediment. The only apparent discrepancy in the data is the relatively high enrichment ratio for W15 considering its high AI. Soils in W15 had the highest organic C contents that suggests the sediment is transported in an organic C stabilized, clay aggregate form. Bedload sediment data tends to support this explanation since W15 had the highest clay contents and the highest bedload:soil clay ratio. Apparently, bedload sediment in W15 contains a relatively high proportion of larger, stable clay aggregates. With the exception of W10, the bedload:soil clay ratios were approximately 50% lower than W15. The r value for bedload:soil clay ratios vs.. AI was only 0.51. Obviously, the bulk of the fine sediments leaving the watersheds is in a suspended form.

The organic C contents also indicate that the suspended sediment was enriched and the bedload sediment depleted relative to the watershed soils. The average ratios were 2.13 and 0.66, respectively. The highest suspended sediment concentrations came from the lower AI soils, again with the exception of W15. The r obtained for suspended/soil organic C vs.. AI was -0.866 ($P \leq 0.05$). On the average, the organic C concentrations in suspended sediments are twice that of the watershed soils. The bedload:soil organic C ratios indicate that the highest value (0.96) occurred in W15 and lowest in W3 (0.38) and W11 (0.36). The r value for these ratios vs. AI was 0.61.

Table 2. Total clay and organic C contents of suspended and bedload sediments, and watershed soils.

	Total clay contents					Organic C contents				
Sub-watershed	Watershed soils	Suspended sediment	Bedload	Suspended/ soils	Bedload/ Soils	Watershed soils	Suspended sediment	Bedload	Suspended/ soils	Bedload/ soils
	———	gkg^{-1}	———			———	gkg^{-1}	———		
3	129	216	60	1.68	0.46	11.1	32.1	4.2	2.89	0.38
7	125	175	57	1.40	0.46	9.6	24.9	6.8	2.59	0.71
9	153	156	85	1.02	0.56	11.6	19.9	8.2	1.72	0.70
10	146	171	105	1.17	0.72	11.0	19.3	9.1	1.76	0.83
11	134	168	49	1.25	0.37	11.9	21.6	4.2	1.81	0.36
15	134	182	108	1.36	0.81	12.9	26.0	12.4	2.01	0.96

Table 3. Distribution of watershed soil properties as a function of slope position.

Slope Position	Total clay	WDC	Total carbon	Organic carbon	Aggregation index	pH	Magnetic susceptibility	Hue	Value	Chroma
	Soil Property									
	---------- $g kg^{-1}$ ----------						$10^{-8} m^3 kg^{-1}$			
SU[1]	146 ab[2]	106 ab	21.5 a	11.6 a	26.8 abc	7.8 a	257 abc	6.9 YR a	3.1 a	1.8 b
SH	143 ab	99 b	23.6 a	12.6 a	29.2 a	7.8 a	333 a	7.0 YR a	3.1 ab	1.8 b
UBS	140 ab	110 ab	22.7 a	12.5 a	25.7 cd	7.8 a	284 ab	7.0 YR a	3.1 ab	1.7 b
MBS	138 ab	100 b	23.6 a	11.3 ab	26.3 bcd	7.8 a	337 a	7.1 YR a	3.1 ab	1.8 b
LBS	152 a	116 a	24.1 a	12.5 a	24.2 d	7.8 a	313 a	6.9 YR a	3.1 ab	1.9 ab
FS	146 ab	103 b	14.8 b	9.7 bc	28.5 ab	7.4 b	204 bc	6.4 YR b	3.0 ab	1.9 ab
TS	135 b	99 b	15.4 b	9.4 c	27.2 abc	7.8.b	190 c	6.5 YR b	3.0 b	1.9 a

[1] SU = summit, SH = shoulder, UBS = upper backslope, MBS = mid backslope, LBS = lower backslope, FS = footslope, TS = toeslope

[2] Values followed by same letter are not statistically different at $P \leq 0.20$ based on Duncan's new multiple range test.

The distribution of soil AI as a function of watershed slope factors was determined for the overall watershed using the data from all 457 samples. Additional soil properties were included in these evaluations to assess the dependence of AI on their distribution. The effect of slope position on AI is shown in Table 3. The data indicate that the higher soil AI occurred on the summit (SU), shoulder (SH), footslope (FS), and toeslope (TS) positions. The three backslope positions (UBS, MBS, LBS) had a lower AI although in some cases they were not significantly different ($P \leq 0.20$) from other positions. No clear explanation exists for why the backslope positions have the lowest AI based on this set of soil properties, however, magnetite appears to be accumulating on these slope positions, as indicated by MS. This indicates a coarser particle size distribution as described for mid-slope positions in other studies (Franzmeier et al. 1969). The lower hue and slightly higher chroma values found in the FS and the TS positions may indicate redder soil colors than upslope due to high Fe contents that normally increases the AI.

The distribution of AI as a function of slope class (gradient) indicated that the A, E, and F slopes had significantly ($P \leq 0.20$) greater values than the B, C, and D slopes (Table 4). In this case, the higher AI appears to be due to greater total clay and organic C concentrations. The occurrence of the higher AI on the steeper slopes (E, F) are the result of the Graham/Lampshire mapping units that predominate on the steeper slopes. These soils characteristically are relatively high in clay and organic C.

Table 4. Distribution of watershed soil properties as a function of slope class.

Slope Class	Total clay	WDC	Total carbon	Organic carbon	Aggregation index	pH	Magnetic susceptibility	Hue	Value	Chroma
	gkg^{-1}						$10^{-8}m^3\,kg^{-1}$			
A (0-2%)	148 b[1]	105 bc	18.3 b	10.4 d	27.8 a	7.6 c	239 b	6.7 YR c	3.1 b	1.8 bc
B (3-5%)	130 c	97 c	22.4 a	9.9 d	24.8 c	8.1 a	323 a	6.8 YR c	3.2 a	2.0 a
C (6-8%)	142 b	105 bc	21.3 a	10.2 d	24.9 c	7.8 bc	314 a	6.8 YR c	3.1 b	1.8 b
D (9-12%)	145 b	108 b	24.1 a	12.0 c	24.7 c	7.9 ab	318 a	7.1 YR a	3.1 b	1.8 c
E (13-20%)	167 a	121 a	23.9 a	15.2 b	27.7 b	7.6 c	220 b	6.9 YR bc	2.9 c	1.7 d
F (> 20%)	160 a	104 bc	22.6 a	16.8 a	33.7 a	7.2 d	392 a	7.1 YR ab	2.8 d	1.6 d

[1] Values followed by same letter are not statistically different at $P \leq 0.20$ based on Duncan's new multiple range test.

Table 5. Distribution of watershed soil properties as a function of slope aspect.

Slope Aspect	Total clay	WDC	Total carbon	Organic carbon	Aggregation index	pH	Magnetic susceptibility	Hue	Value	Chroma
	gkg^{-1}						$10^{-8}m^3\,kg^{-1}$			
N (0-45°)	144 ab[1]	102 ab	241 a	12.5 a	26.5 abcd	7.7 bc	465 a	7.1 YR b	3.0 cd	1.8 b
NE (45-90°)	139 b	102 ab	243 a	11.2 a	27.7 abc	7.9 b	418 ab	7.5 YR a	3.2 a	1.9 ab
E (90-135°)	144 ab	110 a	230 ab	10.8 a	23.3 e	8.2 a	189 e	7.0 YR bc	3.2 ab	1.9 a
SE (135-180°)	149 ab	115 a	232 ab	10.8 a	24.4 de	7.9 b	199 e	6.8 YR cd	3.1 bc	1.8 b
S (180-225°)	140 b	103 ab	221 ab	11.0 a	25.8 bcde	7.8 bc	250 de	6.7 YR cd	3.1 bc	1.8 b
SW (225-270°)	138 b	98 b	210 ab	11.9 a	28.8 a	7.6 cd	360 bc	6.9 YR bc	3.1 b	1.8 ab
W (270-315°)	145 ab	107 ab	218 ab	12.2 a	25.1 cde	7.9 b	267 de	7.1 YR b	3.0 cd	1.8 ab
NW (315-360°)	158 a	111 a	205 b	12.7 a	28.5 ab	7.4 d	307 cd	6.6 YR d	3.0 d	1.8 b

[1] Values followed by same letter are not statistically different at $P \leq 0.20$ based on Duncan's new multiple range test.

In terms of slope aspect (Table 5), the greatest AI occurred on the N-facing slopes (NW, N, NE) with the exception of the SW-facing slope data. The lower AI was associated with the E-, SE-, S-, and W- facing slopes where average annual temperatures are warmer, and soil water and organic C contents are lowest (Franzmeier et al. 1969, Hanna et al. 1982). The organic C data from this study follows a similar trend, with the lowest concentrations recorded on the E-, SE-, and S- facing slopes. The MS data follow the same general trend as AI. Specifically, the highest readings are on the N-, NE-, and SW- facing slopes, with the lowest on E-, SE-, and S- facing slopes. Apparently, cooler temperatures associated with the N-facing slopes has resulted in less oxidation of the magnetite soil component which is responsible for the magnetic signature in this environment.

Conclusions

This study has shown that a weighted average approach to the characterization of basic watershed soil properties, that influence soil erodibility, has the potential to improve our understanding of hillslope erosion processes in semiarid environments including the composition of sediment in streams draining individual watersheds. The fact that an aggregation index computed for individual soils is highly correlated with suspended sediment properties, and varies as a function of slope position, slope gradient, and slope aspect indicates that soil erodibility zones can be differentiated in watersheds using digital elevation models and digitized soil surveys. The incorporation of this information into soil erosion prediction models has the capability to improve their accuracy at the watershed scale. Additionally, from an environmental science perspective, this approach can provide scientists with a reliable estimate of carbon fluxes, and a means of assessing the movement of chemical contaminants in semiarid watersheds.

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Suspended-Sediment-Transport Rates at the 1.5-Year Recurrence Interval for Ecoregions of the United States: Transport Conditions at the Bankfull and Effective Discharge

Andrew Simon

Abstract

Historical flow and suspended-sediment transport data from more than 2,900 sites across the United States have been analyzed in the context of estimating flow and suspended-sediment transport conditions at the 1.5-year recurrence interval flow ($Q_{1.5}$). This is particularly relevant with the renewed focus on stream restoration activities and the urgency in developing water-quality criteria for sediment. Arguments are developed that in lieu of form-based estimates of say the bankfull level, a flow of a given recurrence interval ($Q_{1.5}$) is more appropriate to integrate suspended-sediment transport ratings for the purpose of defining long-term transport conditions at a site. At the $Q_{1.5}$ the highest median suspended-sediment concentrations occur in semi-arid environments the highest yields occur in humid regions with erodible soils and steep slopes or channel gradients. Suspended-sediment yields for stable streams are used to determine "background" or "reference" sediment-transport conditions in eight ecoregions where there is sufficient field data. The median value for stable sites within a given ecoregion are generally an order of magnitude lower than for non-stable sites.

Keywords: TMDL, bankfull, effective discharge

Introduction

Sediment is listed as one of the principle pollutants of surface waters in the United States, both in terms of sediment quantity ("clean sediment") and sediment quality due to adsorbed constituents and contaminants. Fully mobile streambeds, and deposition of fines amidst interstitial streambed gravels can pose hazards to fish and communities of benthic macro-invertebrates by disrupting habitats, degrading spawning habitat, and reducing the flow of oxygen through gravel beds. Although lethal or sub-lethal levels are unknown at this time, high concentrations of suspended sediment, perhaps over certain durations can adversely affect those aquatic species that filter and ingest water. It is critical, therefore, to clearly identify the potential functional relation between an impact due to sediment and the sediment process so that appropriate parameters are analyzed.

Hundreds of thousands of kilometers of stream channels have been designated as being impaired due to sediment. States, Territories and Tribes are required to determine the maximum allowable loadings to, or in a stream that does not impair the "designated use" of that particular water body. This measure has been termed a "TMDL" (total maximum daily load). However, this by no means indicates that a TMDL for sediment transport should be expressed in terms of a total load, or a daily-maximum load. In fact neither of these metrics is probably appropriate for sediment and other means of describing reference, impacted and impaired sediment-transport conditions at a site are more meaningful and scientifically defensible. In lieu of form-based estimates of the bankfull level, a flow of a given frequency and recurrence interval is perhaps more appropriate to integrate suspended-sediment transport rates for the purpose of defining long-term transport conditions at sites from diverse regions.

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Bankfull and effective discharge

The *bankfull discharge* is the maximum discharge that can be contained within the channel without overtopping the banks (Leopold et al. 1964) and generally accepted to represent the flow that occurs, on average, every 1.5 years ($Q_{1.5}$). Dunne and Leopold (1978) described the discharge at the bankfull stage as the most effective at forming and maintaining average channel dimensions. This has led to the term "bankfull discharge" being often used interchangeably with the terms "effective discharge," "channel-forming discharge," and "dominant discharge." One of the primary reasons for this confusion is that originally defined bankfull discharge and the dimensions represented by hydraulic geometry relations refer to stable channels. A bankfull level in unstable streams can be difficult to identify particularly in erosional channels because of a lack of depositional features and because channel dimensions, including water-surface elevations (of specific discharges), are changing with time. In searching for a meaningful discharge or range of discharges to compare sediment-transport rates it may be best to avoid form-based "bankfull" criteria, instead using a consistent flow-frequency value that can be linked to geomorphic processes, alluvial channel form, and hence, sediment-transport rates, such as the effective discharge.

The *effective discharge* is the discharge or range of discharges that transports the largest proportion of the annual suspended-sediment load over the long term (Wolman and Miller 1960). Pickup and Warner (1976) found the return period of the effective discharge to range between 1.15 and 1.4 yr (using the annual maximum series) using bed load transport equations to estimate sediment transport.

The purpose of the research reported here was to test the hypothesis that suspended-sediment concentrations and yields could be regionalized for the conterminous United States. Although clean sediment can adversely affect habitat and other designated uses in a variety of ways, this paper will be limited to discussions and analysis of methods and techniques for analyzing impacts due to suspended sediment.

Availability of Data and Regionalization

Analysis of suspended-sediment transport at the national scale requires a large database of suspended-sediment concentrations with associated instantaneous water discharge. Data of this type permit analysis of sediment-transport characteristics and the development of rating relations (Glysson 1987). The U.S. Geological Survey (USGS) has identified more than 6,000 sites nationwide where at least 1 matching sample of suspended sediment and instantaneous flow discharge have been collected (Turcios and Gray 2001). At more than 2,900 of the sites there is sufficient data (minimum of 30 matching samples) to develop relations between flow and suspended-sediment concentration and load. This massive historical database serves as the foundation for analyzing sediment-transport characteristics over the entire range of physiographic conditions that exist in the United States. It should be stressed that the sediment-transport rates reported here represent two phases of sediment movement; wash load (generally silts and clays) and suspended bed-material load (generally sands) but excludes bed load. Stream systems dominated by bed load, therefore, may not be well represented here.

To be potentially useful for practitioners in stream restoration and water quality studies, sediment-transport relations derived from this existing database must be placed within a conceptual and analytical framework such that they can be used to address sediment-related problems at sites where no such data exist. Sediment-transport characteristics and relations need to be regionalized according to attributes of channels and drainage basins that are directly related to sediment production and transport. In the following eight ecoregions a sufficient number of sites were visited to determine relative channel stability, thereby providing a basis to compare differences in suspended-sediment transport between stable and unstable sites: Coast Range (#1), Northern Rockies (#15), Arizona/New Mexico Plateau (#22), Flint Hills (#28), Central Irregular Plains (#40), Middle Atlantic Coastal Plain (#63), Southeastern Plains (#65), Mississippi Valley Loess Plains (#74). A stable channel is one that over a period of years does not experience net changes in width, depth, gradient, or planform and can essentially transport all sediment delivered from upstream without net erosion or deposition.

Methods

Effective discharge calculations

To provide a check on the validity of the $Q_{1.5}$ as an estimate of the effective discharge for suspended sediment a three-step process is required: (i) construct a flow-frequency distribution; (ii) construct a sediment-transport rating relation; and (iii) integrate the two relations by multiplying the sediment-transport rate for a specific discharge class by that discharge.

The discharge class with the maximum product is defined as the effective discharge (Andrews 1980). The flow data used for this analysis should be of the greatest available frequency, such as those corresponding to 15-minute stage data, but these are often hard to obtain. We were able to obtain 15-minute flow data for 10 sites located in Mississippi, and in lieu of the 15-minute flow data, mean-daily flow data were obtained for about 500 sites representing 17 different ecoregions. Flow data were ranked and then subdivided into 33 discharge classes (Yevjevich 1972). This procedure was used for the 10 sites in Mississippi and the other 500 sites across the United States to test the recurrence interval of the effective discharge.

Suspended-sediment transport rating relations

A first approximation, suspended-sediment transport rating (Glysson 1987, Simon 1989a) of discharge versus concentration was plotted in log-log space and regressed with a power function. Trends of these data often increase linearly and then break off and increase more slowly at high discharges because although sand concentrations continue to increase with discharge, the silt-clay fraction attenuates, causing the transport relation to flatten. To alleviate the problem of overestimation of concentrations at high flow rates caused by this attenuation, a second (or even third) linear segment (in log-log space) is often fitted with the upper end of the data set (Simon 1989a). The concentration at the midpoint of each discharge class is then calculated from the rating relation and multiplied by the discharge and its percent occurrence. The discharge class containing the highest value is, by definition, the effective discharge.

Recurrence interval of the effective discharge for suspended sediment

The effective discharge (Q_{eff}) was calculated using the above procedure for the 10 sites in Mississippi using 15-minute flow data and the other 500 sites using mean-daily flow data. Results show, that for a given ecoregion, the median recurrence interval of the effective discharge for suspended sediment ranges from 1.1 to 1.7 yr. The $Q_{1.5}$ was obtained for all sites from log-Pearson III analysis of the annual-maximum series and compared to the effective discharge calculated by the above procedure. For the 10 Mississippi streams analyzed, the $Q_{1.5}$ proved to be a good approximation being on average, about 10% greater than the calculated effective discharge (Simon et al. 2001). Results from the other ecoregions show, as expected, a greater range given the diversity of geomorphic conditions, with the median ratio of Q_{eff} to $Q_{1.5}$ between 0.6 to 1.3. Still, results showing the remarkably consistent recurrence interval value for the effective discharge indicate that using the $Q_{1.5}$ as a *measure* of estimating the effective discharge at the remaining study sites is reasonable. We may then be able to extend this argument to the bankfull discharge given the numerous authors that have found that the bankfull and $Q_{1.5}$ discharges are similar for regions as diverse as the arid American Southwest (Odem et al. 1999) and the Pacific Northwest (Castro and Jackson 2001).

The consistent results supporting the use of the $Q_{1.5}$ as a measure of the effective discharge are not meant to be definitive for all streams in every ecoregion of the United States but as a mechanism to define and compare suspended-sediment transport rates from historical datasets from the different ecoregions spanning the country. Further, the selection of a single flow frequency, in this case the $Q_{1.5}$, provides a degree of internal consistency by which to compare suspended-sediment transport rates from diverse regions of the United States.

Regional flow relations at the $Q_{1.5}$

The annual maximum peak-flow series for each of the sites with available data was used to calculate the effective discharge ($Q_{1.5}$) from the log-Pearson Type III distribution. The resulting $Q_{1.5}$ data were sorted by Level III ecoregion and regressed with drainage area. In eight of the ecoregions, there were an insufficient number of sites to develop regression relations. Of the remaining 76 ecoregions, 75% of the derived relations included at least 17 sites and had r^2 values of at least 0.60; 50% of the relations included at least 27 sites and had r^2 values of 0.80.

Suspended-sediment transport rates at the effective discharge by Level III ecoregion

Using the procedures for developing suspended-sediment transport relations and the $Q_{1.5}$ described above, values of concentration and yield (load divided by drainage area) were obtained for each site. This was accomplished by applying the calculated $Q_{1.5}$ to the suspended-sediment rating relation to obtain the transport rate. So as not to extrapolate relations beyond measured bounds, sites were excluded from the analysis if the $Q_{1.5}$ exceeded the maximum sampled discharge by 50% or more. The remaining dataset (2430 sites) was sorted by ecoregion to differentiate between regional trends in suspended-sediment transport. Suspended-sediment transport data at the $Q_{1.5}$ are reported in terms of concentration (mg/l) and also as a yield (t/d/km^2) to compare streams of varying size within ecoregions.

Results

Values of suspended-sediment transport within a single ecoregion may represent a broad range of conditions including various states of channel and watershed stability, dominant bed-material size class, and anthropogenic influence. Still, about 70% of the ecoregions have inter-quartile ranges for suspended-sediment yield within a single order of magnitude. Large inter-quartile ranges (two orders of magnitude) in ecoregions such as the Mississippi Valley Loess Plains (#74), Coast Range (#1), and Northern Piedmont (#60) represent areas where anthropogenic disturbances combined with erodible soils and high, seasonal rainfall create conditions for large increases in suspended-sediment yields.

Measured suspended-sediment concentrations at the $Q_{1.5}$ reached more than 100,000 mg/l in some of the semiarid streams of the southwest such as in the Arizona/New Mexico Plateau (ecoregion 22). In fact, ecoregions in this part of the United States have some of the highest median concentrations in the nation owing to large quantities of available sediment in storage, limited vegetative cover, and the flashy nature of runoff events. Examples of these ecoregions include the Southwest Tablelands (#26; 9530 mg/l), Mojave Basin and Range (#14; 5150 mg/l), and the Arizona/New Mexico Plateau (#22; 4140 mg/l). Midwestern ecoregions such as the Central Great Plains (#27; 3770 mg/l), the Nebraska Sand Hills (#44; 2110 mg/l), and the Mississippi Valley Loess Plains (MVLP) (#74; 2170 mg/l) also showed high median values at the $Q_{1.5}$. Of these, only the MVLP can be considered a humid region and the high median concentration reflects the highly erodible nature of the loess hills and the generally unstable conditions of the stream systems. As expected, the lowest values occurred in ecoregions characterized by gently sloping gradients such as the Southern and Middle Atlantic Coastal Plains (#s 75 and 63, respectively) and those characterized by shallow soils and resistant bedrock such as the Northern Rockies (#15) and the Laurentian Plains and Hills (#82).

A somewhat different picture of peak values emerges from the national distribution of suspended-sediment yields at the $Q_{1.5}$. The highest median-yield values occur in humid regions such as the MVLP (#74; 173 t/d/km^2) and the Coast Range (#1; 55.8 t/d/km^2) where plentiful flow energy is available for sediment transport and where over-steepened channel gradients in the case of the former and accelerated mass wasting in upland areas in the case of the latter produce high suspended-sediment yields. Areas of the semiarid southwest have moderate suspended-sediment yield values where flows tend to attenuate rapidly downstream through infiltration, thereby reducing transport rates with increasing drainage area. The geographic distribution of lowest median yields shows a similar pattern to the distribution of the lowest median concentrations. Differentiation based on Level III ecoregion is further supported by the expected systematic decrease in both median concentrations and yields as one moves downslope from the Blue Ridge (#66) through the Piedmont (#45), Southeastern Plains (#65) to the Middle Atlantic (#63) and Southern (#75) Coastal Plains.

These yield results differ somewhat from the classic paper by Langbein and Schumm (1958) who reported peak annual sediment yields in semiarid environments where the dominant vegetation type changed from desert shrub to grassland. Whereas our results showed some of the highest suspended-sediment concentrations in these areas, the greatest median yield values in the continental United States occurred in the humid, yet unstable systems of the MVLP and the Coast Range of the Pacific Northwest. Ample supplies of precipitation and flow energy in the disturbed streams of the MVLP provided large quantities of channel sediments, particularly streambank materials, while mass wasting of disturbed uplands areas in the Coast Range made available plentiful amounts of sediment that produced great quantities of suspended sediment per unit area. Areas of the lower Midwest and areas flanking the Appalachians showed moderately high suspended-sediment yields. The relatively high values for these areas could largely be due to land disturbances and the consequent remobilization of historically stored sediment.

Background or "reference" suspended-sediment transport conditions

Rates and concentrations of suspended-sediment transport vary over time and space due to factors such as precipitation characteristics and discharge, geology, relief, land use, and channel stability. It is unreasonable to assume that "natural" or background rates of sediment transport will be consistent from one region to another. Within the context of channel design for stream restoration and developing water quality targets for sediment, there is no reason to assume then that "target" values should be consistent on a nationwide basis. Similarly, it is unreasonable to assume that channels within a given region will have consistent rates of sediment transport. This reflects differences in the magnitude and perhaps type of erosion processes that dominate a subwatershed or stream reach.

To identify those sediment-transport conditions that represent impacted or impaired conditions, one must first be able to define a nondisturbed, stable, or "reference" condition for the particular stream reach. In some schemes, the "reference" condition simply means "representative" of a given category of classified channel forms or morphologies (Rosgen 1985) and as such may not be analogous with a "stable," "undisturbed," or "background" rate of sediment production and transport. Although the Rosgen (1985) stream classification system is widely used to describe channel form, stream types D, F, and G are, by definition, unstable (Rosgen 1996). These stream reaches, therefore, would be expected to produce and transport enhanced amounts of sediment and represent impacted conditions. Thus, although it may be possible to define a "representative" reach for stream types D, F, and G, a "reference" condition transporting "natural" or background rates of sediment will be difficult to find.

As an alternative scheme, the channel evolution frameworks set out by Schumm et al. (1984) or Simon and Hupp (1986) and Simon (1989b) are proposed. With stages of channel evolution tied to discrete channel processes and not strictly to specific channel shapes, they have been successfully used to describe systematic channel-stability processes over time and space in diverse environments subject to various disturbances. An advantage of a process-based channel-evolution scheme is that Stages I and VI represent two true "reference" conditions. In some cases, such as in the midwestern U.S. where land clearing activities near the turn of the twentieth century caused massive changes in rainfall-runoff relations and land use, channels are unlikely to recover to Stage I, pre-modified conditions. Stage VI, re-stabilized conditions are a more likely target under the present regional land use and altered hydrologic regimes and can be used as a "reference" condition. However, in pristine areas where disturbances have not occurred or where they are far less severe, Stage I conditions can be used as a reference.

The working hypothesis for determining background or "reference" values for suspended-sediment transport in this study is that stable channel conditions and therefore background or "natural" sediment transport rates can be represented by channel evolution Stages I and VI. As expected, Stage VI sediment-yield values are considerably lower for each quartile measure in each of the ecoregions. The median value for stable sites within a given ecoregion are generally an order of magnitude lower than for nonstable sites. The results show a four order-of-magnitude range of median "reference" values for the eight ecoregions, further supporting the premise that water quality targets for sediment need to be done at least at the Level III ecoregion scale, if not smaller. These results should be considered preliminary as more sites in each of the ecoregions are evaluated for stage of channel evolution, and the data set is further differentiated by dominant bed-material size class.

Summary and conclusions

Using the ecoregion concept devised by Omernik (1995), historical flow and suspended-sediment transport data from more than 2,900 sites nationwide have been analyzed to develop regional-flow curves and suspended-sediment transport rates for each ecoregion. Data from about 500 sites across the U.S. were used to calculate the recurrence interval of the effective discharge for suspended sediment transport. Median values for the 17 ecoregions tested ranged from 1.1 to 1.7 yr. Thus, the $Q_{1.5}$ proved to be a reasonably good measure of the effective discharge for suspended sediment and was used in conjunction with derived suspended-sediment transport relations to calculate concentrations and yields at all sites. Peak median concentrations occurred in the semiarid areas of the southwestern U.S., while maximum yields occurred in the Mississippi Valley Loess Plains and the Coast Range. Background or "reference" suspended-sediment transport conditions were determined by sorting the data into stable and unstable sites using the Simon and Hupp (1986) and

Simon (1989b) model of channel evolution and by taking the median value for stable sites (Stages I and VI) in a given ecoregion.

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Impact of Grass Hedges on Sediment Yield from a HEL Watershed

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Abstract

Stiff-stemmed grass hedges offer many opportunities to control erosion and other environmental contaminates leaving a field. The objective of this research was to evaluate the erosion-control effectiveness of narrow rows of grass hedges planted on 15.5-m spacings within a 6-ha watershed located in the deep loess hills region of western Iowa. Because only one watershed was planted in grass hedges, three different approaches were used to evaluate the erosion control effectiveness of the grass hedges as measured by sediment yield at the watershed outlet. The first approach was to compare measured surface runoff and sediment yields from the 1975-1991 period without hedges to the 1992-2002 period with hedges. The second approach was to develop a linear regression between annual sediment yield and surface runoff from data collected during the 1975-1991 non-hedge period and then estimate sediment yield without hedges for the 1992-2002 period from measured runoff values. The third approach was to use the WEPP Watershed model. The model was calibrated with data from the 1975-1991 period and then used to predict runoff and sediment yields without hedges for the 1992-1999 period using measured climatic and cropping and management inputs. Predicted sediment yields without hedges for the second and third approaches were compared with measured values with hedges. Stiff-stemmed grass hedges planted within the 6-ha watershed reduced sediment yield from 39 to 64% depending upon the approach used. Grass hedges had little to no impact on surface runoff losses, thus grass hedges acted as leaky dams temporarily ponding runoff, reducing sediment concentrations, and reducing sediment yield at the watershed outlet.

Keywords: erosion, erosion modeling, surface runoff, WEPP

Introduction

Stiff-stemmed grass hedges have been used in developing countries for many years as a natural way to trap sediment, bench the landscape, and reduce the inter-hedge slope (Alberts and Neibling 1994). The approach generally taken has been to select a native grass that has plant characteristics such as rigid stems, high stem density, rapid tillering, and tolerance to sediment deposition. While quantitative information on the mass of sediment deposition is relatively limited, observations indicated that sediment buildup behind grasses is quite high, often a meter or so every decade. Stem density is often so high that a temporary pond forms behind the grass hedge that does not drain completely until hours after a runoff event. The natural effectiveness of the hedge is also supplemented by pieces of residue and other organic debris that wash down to form an initial mat or barrier that the runoff must move through before being discharged. In the early 1990s, a Grass Hedge Workgroup was formed comprised of scientists, conservationists, and others interested in this technology and how it might be used in the U.S. as an alternative to expensive terrace construction (Kemper et al. 1992). Much of the research to investigate the erosion-control effectiveness of grass hedges has been conducted by the USDA-Agricultural Research Service in conjunction with personnel from the USDA-Natural Resources Conservation Service. Primary research sites were Holly Springs, MS (McGregor et al. 1999), Columbia, MO, and Treynor, IA (Kramer and Alberts 2000). For small plots where sheet-rill erosion dominates the erosion process, such

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as at Holly Springs and Columbia, research has shown that stiff-stemmed grass hedges reduce mean annual sediment losses by 60 to 80% compared to similar treatments without hedges. It is much more difficult to evaluate the influence of stiff-stemmed grass hedges on erosion control within a watershed with complex topography and steep slopes, and where much of the runoff within the watershed flows to ephemeral gully areas (channels) for movement out of the watershed. The objective of this research was to evaluate the erosion-control effectiveness of narrow rows of stiff-stemmed grass hedges planted on 15.5-m spacings within a 6-ha watershed in the deep loess hills region of western Iowa, an area comprised primarily of Highly Erodibile Land (HEL). The study spans a 28-year period (1975-2002), with 17 years of surface runoff and sediment yield measurements without grass hedges and 11 years of similar measurements with a network of established grass hedges.

Methods

Study area

The watershed is within Major Land Resource Area (MLRA) 107, the Iowa and Missouri Deep Loess Hills region, an area of 5.3 M ha. Corn and soybeans are the principal row crops.

The topography is very rolling, with slopes of 2 to 4% on the ridges and valleys and 12 to 16% on the sides. The soils generally have a silt loam texture throughout their profile and are well drained and highly erodible. Principal soil series are the Typic Hapludolls, Typic Udorthents, and Cumulic Hapludolls.

Most of the precipitation occurs as rainfall. During the spring and early summer, intense rainstorms often occur on soil that has been tilled. Most of the soil water recharge occurs in the fall when slow-moving cold fronts create long-duration, low-intensity rainfall events.

1975-1991 period

The watershed was instrumented in 1974 with a broad crested V-notch weir, a water-stage recorder, and an automatic pumping sampler to collect sediment samples during a runoff event. During this 17-year period, the watershed was cropped to continuous corn. Typical tillage operations included two to three tandem diskings, anhydrous ammonia application, planting with a 4-row planter, rotary hoeing, one to two cultivations, and harvesting. The corn stalks were seldom chopped after harvesting. Lines for parallel, narrow rows of grass hedges were established on the general contour in April 1991. Because no guidelines were available on hedge establishment and spacing, a hedge spacing of 15.5 m was chosen to accommodate 16 rows of corn between each hedge (Figure 1). It was recognized early in the experiment that establishing and maintaining hedges that crossed ephemeral gullies would be a challenge. Clumps of Miscanthus were transplanted in areas where the hedges were to cross major ephemeral gully channels. Cave-in-Rock Switchgrass was drilled in May on the south side of the watershed.

Figure 1. Topography of 6-ha watershed showing

Switchgrass hedge width and interval.

Surface runoff and sediment yield at the watershed outlet were measured on an event basis during this 17-year period. Some snow-melt events in the early Spring were missed because the gauging site was out of operation.

1992-2002 period

On the north side of the watershed, Alamo Switchgrass was drilled in May 1992. Hedges were overseeded after initial drilling to establish a satisfactory stand with high stem densities. Clumps of Miscanthus and Switchgrass were transplanted in some of the ephemeral gully crossings in 1992. The transition from a non-hedge watershed to one of established hedges took several years, but 1992 starts the period of established grass hedges. Cropping and management operations continued through 1996 as they had since 1985, with continuous corn and heavy disking as the primary tillage operation. In 1997, 1998, and 1999; cropping changed to narrow-row soybeans planted with a no-till drill. A corn-soybean no-till rotation system was initiated in 2000.

In 1998, the highest surface runoff year of record (28 years) caused deep rills to form in most of the ephemeral gully channels. Considerable surveying was done in March and April of 1999 to characterize soil surface elevations. A bulldozer and small scraper were used to fill some of the deeper rills in April and May of 1999. The third highest runoff year of record occurred in 1999, which required tandem disking to fill in deep rills. In May of 2001, clumps of Switchgrass were transplanted in many ephemeral gully crossings to rebuild the hedges.

Because there was not an adjacent watershed that was managed similarly without grass hedges, three approaches were used to evaluate the effectiveness of stiff-stemmed grass hedges as an erosion- control practice.

Evaluation Approaches

Period

This approach will compare surface runoff and sediment yield differences between the 17-year period without grass hedges (1975-1991) to the 11-year period where grass hedges were established (1992-2002).

Statistical

Data collected during the 1975-1991 period were used to develop a linear relationship between measured sediment yield and measured runoff. The relationship was:

Sediment Yield (Mg/ha) = 0.3386* Runoff (mm)

with a regression coefficient (r^2) of 0.78. The equation was then used to predict annual sediment yields from runoff measured during the 1992-2002 period.

WEPP watershed model

The WEPP Watershed model (ver. 99.5) was used in the assessment (Ascough et al. 1997). The model predicts the effects of agricultural management practices and will accommodate spatial and temporal variations in topography, soil properties, and land use conditions within agricultural watersheds generally less than 260 ha in size. The model contains three primary components: hillslope, channel, and impoundment. The six input files required to run the WEPP Watershed model are described briefly.

Climate input file

Measured daily precipitation, duration, time to peak, peak intensity, minimum temperature, maximum temperature, and wind velocity are required. Other daily input parameters, such as solar radiation and dew point temperature, were generated using WEPP's climate generator (CLIGEN).

Slope input file

Based on the direction of flow to the channels, the watershed was segmented into eight hillslopes and three channels. A DEM of the watershed was used to develop slope input parameters for both hillslopes and channels.

Cropping and management input file

Actual tillage dates and implements for each year of simulation were used. Two of the channels had similar management to the hillslopes. The main channel was a grassed waterway, which was represented in the input file.

Soil file

A soil file for the Monona soil (Typic Hapludoll) was used.

Channel input file

Channel input parameters include peak runoff calculation option, friction slope, channel erodibility, and critical shear stress. WEPP default estimated values were used for erodibility and critical shear stress.

Structure input file

This file provides water and sediment routing linkage. The file was created based upon topographic maps according to how the watershed was divided into hillslopes and channel elements and the direction of runoff between the elements.

Results

Period comparison approach

Nearly all the sediment yield from the watershed occurred during the months of March through September. Table 1 shows mean monthly precipitation and surface runoff values for the 1975-1991 non-hedge and 1992-2002 hedge periods for these seven months. Total March through September monthly precipitation and surface runoff for the 1992-2002 period were 5% and 55% higher than those measured for the 1975-1991 period, respectively. As already noted, precipitation and surface runoff values were unusually high in 1998 and 1999.

Table 1. Mean monthly precipitation and surface runoff.

Month	Precipitation		Surface Runoff	
	1975-1991	1992-2002	1975-1991	1992-2002
	mm			
Mar	61	43	1.6	4.2
Apr	78	92	2.0	2.0
May	121	121	14.4	7.4
Jun	115	145	8.7	23.1
Jul	91	114	2.4	12.5
Aug	108	110	5.1	12.9
Sep	88	71	6.6	1.2
Totals	662	696	40.8	63.3

As shown in Table 2, the impact of the grass hedges, including the changes in cropping and tillage practices that occurred in 1997, reduced sediment yield by 56% when mean annual monthly totals for the 1975-1991 and 1992-2002 periods are compared. Sediment yields from two years, 1998 and 1999, accounted for 82% of the total for the 11-year period.

Table 2. Mean monthly sediment yields.

Month	Sediment Yield		Change
	1975-1991	1992-2002	
	Mg/ha		%
Mar	0.02	0.005	-75
Apr	0.97	0.39	-60
May	6.97	1.36	-80
Jun	8.54	4.18	-51
Jul	0.21	1.18	+462
Aug	0.17	0.33	+94
Sep	0.17	0.01	-94
Totals	17.05	7.46	-56

In 1998, failures from excessive concentrated flow breaching the hedges caused headcutting in the ephemeral gullies and undercutting of some of the hedge rows. Some previously deposited sediment was obviously lost from the watershed. Failures occurred again in 1999, although not as extensive as in 1998. Despite higher precipitation, higher runoff, entrainment of previously deposited sediment, and soil disturbance required to repair deep gullies, sediment yields during the grass hedge period were still 56% lower than for the non-grass hedge period.

Regression prediction approach

Measured surface runoff and sediment yields for the 1992-2002 period are shown in Table 3 with the estimated sediment yields from the relationship. Based on this simple analysis, the grass hedges and changes in cropping and tillage practices reduced sediment yield by 67%, from 22.4 to 7.5 Mg/ha. Even though the analysis is confounded by changes in cropping and tillage beginning in 1997, these changes probably had minimum effect on measured sediment yields in 1998 and 1999. Assuming that no-till reduced sediment yields by 75% for 2000, 2001, and 2002, the mean annual estimated sediment yield for the period would be reduced to about 24 Mg/ha. Based on these assumptions, the mean annual estimated sediment yield becomes 20.3 Mg/ha compared to the 7.5 Mg/ha measured value, a reduction of 64%. In 1998 and 1999, only a small portion of the sediment that had been trapped the previous 6 years is thought to have been transported out the watershed, primarily because the rills were deep, extending well below the tillage zone (~0.75 m), and narrow (~0.33-m wide).

Table 3. Measured surface runoff and sediment yields. Estimated sediment yields represent those losses expected if grass hedges had not been planted.

Year	Runoff	Sediment Yield	
		Measured	Estimated
	mm	Mg/ha	
1992	4.3	0.02	1.4
1993	156.3	3.8	52.9
1994	22.5	0.2	7.6
1995	3.2	0.1	1.1
1996	78.0	4.4	26.4
1997	18.1	1.4	6.1
1998	193.5	41.4	65.5
1999	158.1	25.8	53.5
2000	64.4	4.6	21.8
2001	2.7	0.07	0.9
2002	28.4	0.21	9.62
Totals	729.5	82.0	246.8
Mean	66.3	7.5	22.4

WEPP prediction approach

Predicted annual runoff and sediment yield values from the calibrated model compared to those measured at the watershed outlet for the 1975 – 1991 period are shown in Figure 2.

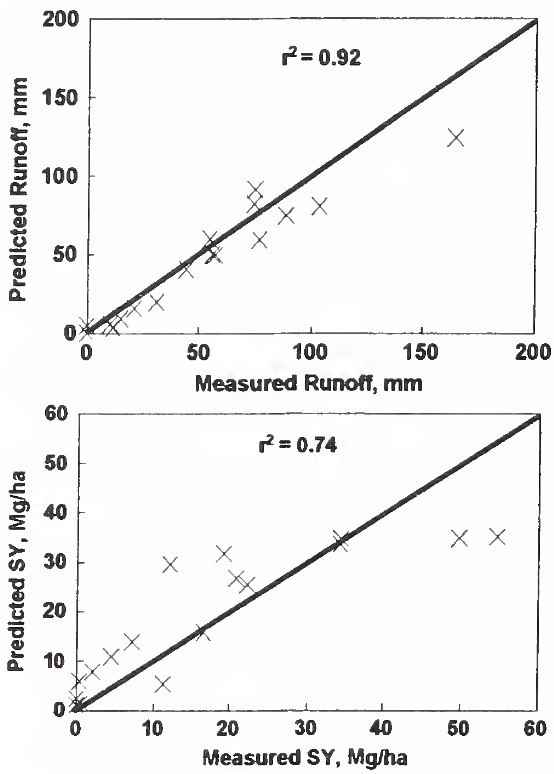

Figure 2. Relationships between predicted runoff and sediment yields (SY) from the WEPP Watershed model and measured values for a 17-year period (1975-1991).

Mean annual predicted and measured surface runoff losses were 45.6 and 55.0 mm, respectively, with a correlation coefficient of 0.96. Initially, the model was over predicting runoff, so the soil hydraulic conductivity value (Ksat) in the Soil input file was adjusted. Mean annual predicted and measured sediment yields were 18.6 and 17.0 Mg/ha, respectively, with a correlation coefficient of 0.86.

Table 4 shows some additional descriptive statistics that are useful in evaluating the performance of the model for predicting sediment yield. The calibrated model is over predicting all the statistic parameters in Table 4. The relative difference in the mean statistic is the smallest because the model under predicted a 49.7 Mg/ha sediment yield by 43% in 1977 and a 54.6 Mg/ha sediment yield by 56% in 1984.

Table 4. Statistical parameters for measured and predicted sediment yields (1975-1991).

Statistic	Sediment Yield Measured	Sediment Yield Predicted	Relative Difference
	---------Mg/ha--------		%
Mean	17.0	18.6	9.2
50% Percentile[1/]	12.0	15.8	31.0
75% Percentile	22.2	31.8	43.3
25% Percentile	2.0	5.9	189.7

[1/] Median value.

It is assumed that the positive bias in the calibrated model will be represented in the predictions for the 1992-1999 period. Until further calibration can be accomplished, it will be assumed that model predictions of sediment yield, particularly for years with low sediment yield values, will have no bias and when compared to measured values, differences will represent the influence of the grass hedges on sediment yield. Mean annual predicted and measured surface runoff losses were 85 and 79.2 mm, respectively (see Table 5). Mean annual predicted and measured sediment yields were 15.8 and 9.6 Mg/ha. Assuming the difference in these values are due to the impact of grass hedges in trapping sediment on the hillslopes and in the channels, the hedges reduced sediment yield by 39% over the 8-year period. As previously discussed, many of the grass hedges crossing ephemeral gully channels were breached and undercut from excessive runoff in 1998 and 1999. Measured sediment yields in those years include some sediment that had been trapped prior to 1998.

In 1998, the measured sediment yield was 41.4 Mg/ha, the third highest on record. Predicted sediment yield was 31.0 Mg/ha. When major rilling occurs on these soils as in 1998 and 1999, the depth of the rill is usually greater than the width because the rill erodibility value does not change with depth. Rill depth became so great in some concentrated flow areas that a bulldozer and small scraper were required to fill in the rill cut. In general, the breaches and cuts through the hedges were narrow relative to the width of the sediment deposition area, implying that only a fraction of the previously trapped sediment was lost from the watershed. Much of the measured sediment yield in 1998 and 1999 came from ephemeral gullies that developed deep rills in the inter-hedge areas.

Table 5. Measured and predicted surface runoff and sediment yields from the WEPP Watershed model (1992-1999 only). Predicted values represent those losses expected if grass hedges had not been planted.

Year	Runoff		Sediment Yield	
	Measured	Predicted	Measured	Predicted
	------- mm--------		--------Mg/ha-------	
1992	4.3	27.3	0.02	1.5
1993	156.3	157.0	3.8	35.9
1994	22.5	21.4	0.2	18.4
1995	3.2	17.6	0.1	0.9
1996	78.0	47.7	4.4	25.6
1997	18.1	24.0	1.4	0.1
1998	193.5	190.0	41.4	31.0
1999	158.1	195.0	25.8	13.0
Totals	634.0	680.0	77.1	126.4
Means	79.2	85.0	9.6	15.8

Conclusions

Surface runoff and sediment yields were measured from a 6-ha watershed located in the deep loess hills region of western Iowa from 1975 through 2002. Beginning in 1991, narrow rows of stiff-stemmed grass hedges were planted on 15.5-m spacings on the approximate contour throughout the watershed. The 28-year period was separated into a 17-year period without grass hedges and a 11-year period with established grass hedges. Three different approaches were used to estimate the impact of the grass hedges on sediment yield. Small plot research conducted in Holly Springs, MS and Columbia, MO indicates that grass hedges trap from 60 to 80% of sheet-rill erosion from <0.01-ha plots. From a watershed with ephemeral gullies and concentrated flow channels, the overall sediment trapping efficiency would be expected to be less than from small plots. A paired watershed in time approach comparing the 1975-1991 and 1992-2002 sediment yields indicated that the grass hedges reduced sediment yield by 56%. A regression approach where a linear relationship between sediment yield and measured runoff was developed for the 1975-1991 non-hedge period and used to estimate sediment yields from the hedge period indicated that grass hedges reduced sediment yield by 64%. For the third approach, the WEPP Watershed model was calibrated with data collected from 1975-1991. Runoff and sediment yields were predicted using measured climate data and dates of actual tillage operations (1992-1999 only). Data from the model indicated that grass hedges reduced sediment yield by 39%. Data will be collected from this watershed for several more years to better quantify the erosion control effectiveness of grass hedges and their impact on reducing sediment yield leaving the field. It is clear, however, that grass hedges on these steep slopes will work best in combination with soil conservation practices, such as, to minimize soil detachment and transport.

Acknowledgments

The authors appreciate the reviews of John Sadler and Allen Thompson.

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Simulating Channel Geomorphic Change in Semi-Arid Watersheds

Darius J. Semmens, David C. Goodrich, Waite R. Osterkamp

Abstract

An event-based distributed simulation model for geomorphic change in semi-arid watersheds has been developed for use on intermediate-scale watersheds. The model incorporates the general KINEROS design, including rainfall, infiltration, erosion, and sediment transport, with a new variable parameter Muskingum diffusion model for stream channel routing, and a geomorphic model for computing changes in channel geometry using the concept of stream power minimization. The model is also been adapted to run continuously, and track cumulative geomorphic change resulting from a series of rainfall-runoff events. Testing of the geomorphic model has been carried out on the Walnut Gulch Experimental Watershed in southeastern Arizona. Detailed rainfall records and repeat measurements of channel geometry were used to develop several multiple-year input data sets for separate calibration and validation of the model. The model is designed to address the need for evaluating distributed management impacts on sediment fluxes through and within large ephemeral channel networks typical of semi-arid environments. It thereby permits the assessment of local impacts as they relate to the larger watershed system, as well as cumulative regional impacts on the channel network. The model is also useful for the assessment of channel stability, and the classification of watersheds in terms of their susceptibility to channel destabilization, riparian habitat degradation, and increased sediment yields. These new assessment capabilities are currently in the process of being added to the Automated Geospatial Watershed Assessment Tool (AGWA).

Keywords: geomorphic, modeling, channel networks

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Subsurface Flow Effects on Soil Erosion in Watersheds

Mathias Römkens, Shyam N. Prasad

Abstract

Soil erosion in watersheds is routinely viewed as a surface phenomenon in which rainfall and surface flow detaches soil that is substantially transported to the drainage network for further disposition. Only rarely is the role of subsurface flow considered and then only in a qualitative manner. The lack of a quantitative assessment can in part be attributed to the highly complex nature of this process and is in part due to differences in time scale between erosion caused by surface flow during a storm event and subsurface flow-induced erosion. Subsurface flow induced erosion often occurs at specific locations in the watershed which commonly are associated with head cut and rill development. This paper presents a brief review of this phenomenon and discusses on-going theoretical research of soil erosion induced by changes in the soil water pressure in the head cut region of a moving head cut and recently started preliminary laboratory experiments of measuring rill growth by subsurface flow effects.

Keywords: soil erosion, subsurface flow, seepage, gully

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First Interagency Conference on Research in the Watersheds

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Watershed Modeling III

Influence of Sub-grid Variability on Snow Deposition and Ablation in North American Mountain Environments: Implications for Upscaling to Meso-scale Representations

Danny Marks, Gerald Flerchinger, Mark Seyfried

Abstract

A measurement and modeling evaluation of how snow distribution and melt are influenced by vegetation and topographic structure at three different experimental catchments in western North America (two in the western U.S. and one in western Canada) has been undertaken. This ongoing research investigates variations in the critical interactions between vegetation, topography and snowcover in different snow-dominated basins, how these variations impact upscaling site- and basin-scale processes for watershed and regional scale analyses, and how these differences are incorporated in transferable methods to account for the effects of sub-grid variability. By uniquely covering a transect of cordilleran research sites from 35 to 61°N and from 1,000 to 2,700 m asl, it provides a true 'Western Cordilleran baseline' for snowmelt runoff prediction for North America that will substantially benefit model development and testing.

Keywords: snow, watershed hydrology

Introduction

Forest snow scaling

The influence of forest canopy cover and variable melt energetics on depletion of snowcover was investigated following earlier work in open environments. The results can be stratified into that variability within the forest stand and that between forest stands. Within stands, Faria et al. (2000) found the frequency distribution of snow water equivalent (SWE) under boreal forest canopies fit a log-normal distribution. Within-stand covariance between the spatial distributions of snow water equivalent and melt energy promoted an earlier depletion of snowcover than if melt energy were uniform. This covariance was largest for the most heterogeneous stands (usually medium density). Stand scale variability in mean SWE and mean melt energy resulted in more rapid snow covered area (SCA) depletion for stands with lower leaf area. Because of the heterogeneity in the spatial distributions of SWE and melt energy in forest environments, it is necessary that these variations be included in calculations of SCA depletion (Faria et al. 2000).

Forest snowmelt energetics

Recent studies of energetics of forest snowmelt (Davis et al. 1997, Hardy et al. 1997, Pomeroy and Granger 1997, Link and Marks 1999a, Link and Marks 1999b, Hardy et al. 2000) have focused on sub-canopy radiative exchange. Sub-canopy insolation is roughly one order of magnitude less than that incoming to a mature pine forest during melt. Snow albedo below forest canopies is lower than that of typical open snowfields (Harding and Pomeroy 1996) and is subject to a premelt decay due to deposited leaf litter from forest canopies (Hardy et al. 2000). Melt simulations that include a litter decay algorithm are vastly improved over those with traditional albedo assumptions (Link and Marks 1999b, Hardy et al. 2000).

Few studies have considered in detail the contribution of sub-canopy longwave radiation, despite the relatively warm canopy temperatures (10 to 20 °C above that of snow) measured during melt.

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Measurements in a boreal pine forest by the authors of this paper (Figures 1 and 2) suggest that net sub-canopy longwave radiation can be a significant component of melt energy (Figure 1), and canopy air temperatures are poor predictors of sub-canopy longwave radiation (Figure 2). The implications of these suggestions are that research should focus on improving longwave radiation parameterizations for snowmelt under forest canopies.

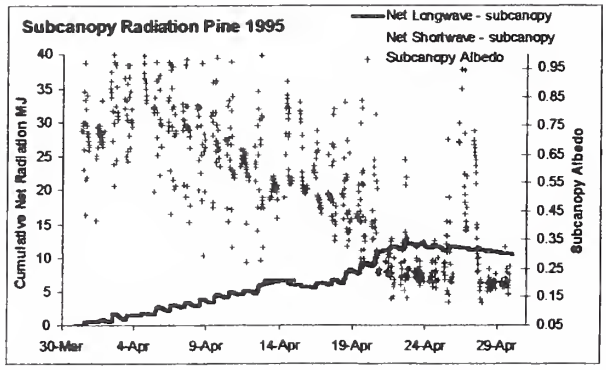


Figure 1. Radiation under a pine canopy 1995. Cumulative net longwave is within 2/3 of net shortwave until the snowpack is largely depleted (albedo < 0.25).

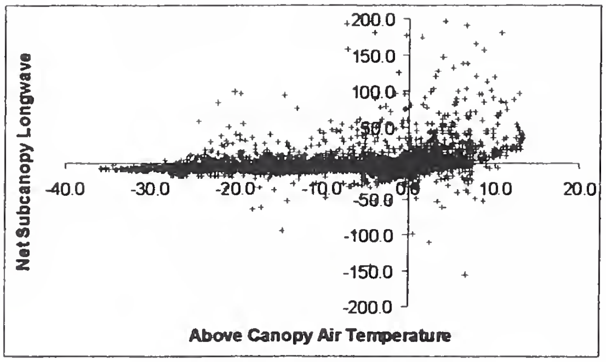


Figure 2. Sub-canopy longwave radiation vs. air temperature under pine canopy, 1995 showing poor predictive relationship.

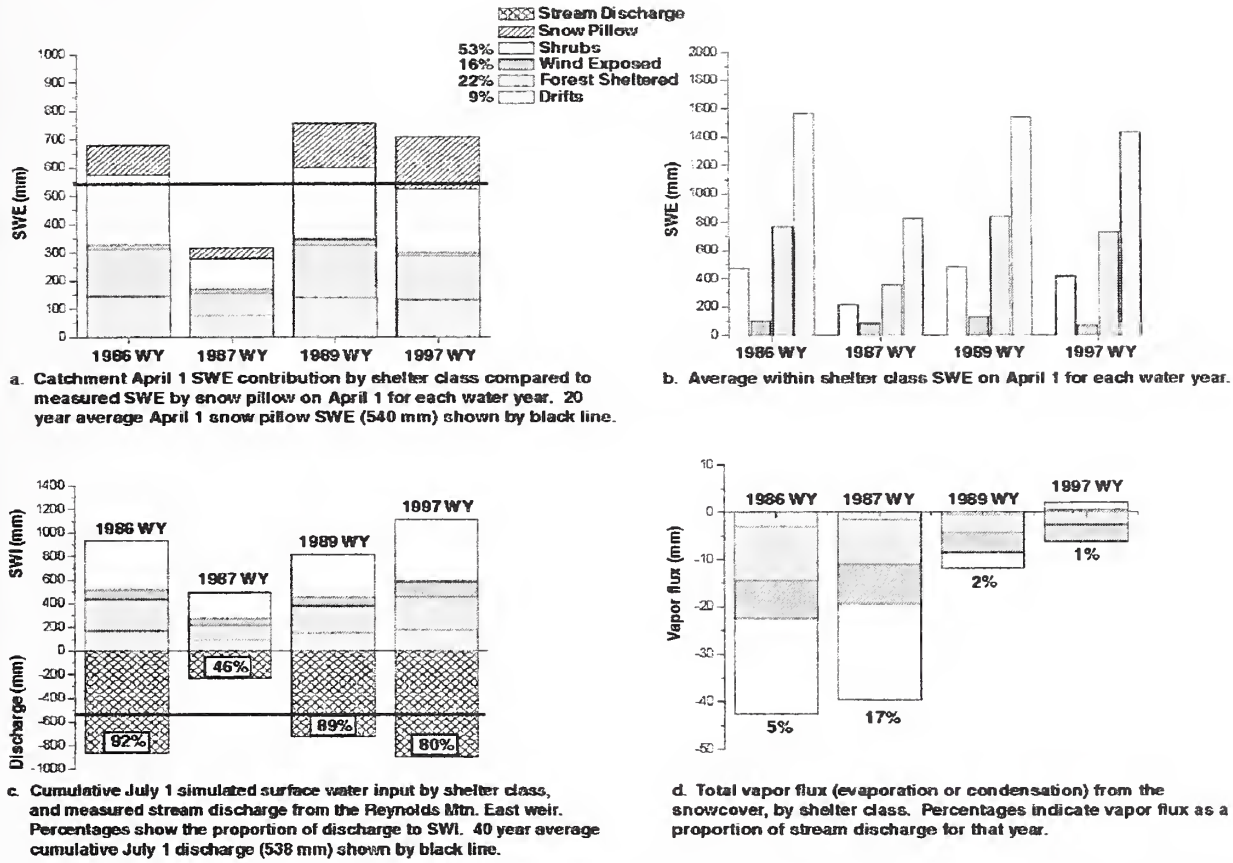
High mountain accumulation and ablation

Marks et al. (2001a, 2002) showed that in mountain catchments, the degree of vegetation and terrain shelter could account for patterns of snow deposition and melt. Figures 3a-3d show how the degree of shelter in a small headwater catchment in the Reynolds Creek Experimental Watershed, Idaho, impacts snow deposition, sublimation, melt and runoff generation. This work suggests that: 1) drift areas represent only a small portion of mountain catchments, but hold a significant portion of catchment SWE; and 2) wind scoured areas may account for a large part of catchment area, but hold hardly any catchment SWE. Furthermore, SWE stored in large drifts persists into late spring and early summer, providing water to ecosystems well into the growing season, while wind exposed areas melt early and are generally snow-free by late winter or early spring.

This prior research suggests that forest SWE mean and variability is controlled by canopy structure, that the below canopy solar and thermal radiation to snow is influenced by canopy type and structure, and that to account for these effects we must improve our understanding of litter, shading and thermal structure of different canopy types. It is also clear that the coupled effects of terrain structure and vegetation in mountain catchments significantly affect the degree of shelter from wind and solar radiation. The degree or shelter determines patterns of drifting, snow deposition, and scour, and alters snowcover energetics causing exposed regions to melt early, and more sheltered regions to melt later in the spring and early summer. In mountainous regions, all of these effects vary at scales much smaller than 1 km.

Background

The processes controlling the rates and magnitude of snow deposition and ablation over complex topography and in and under vegetation canopies remain one of the greatest uncertainties in the operation of land surface schemes and hydrological models over mountainous regions. For instance, very few hydrological or land surface models distinguish between snow intercepted in forest canopies, and the surface snowpack sheltered under forest canopies (Pomeroy et al. 1998). No climate or water model includes the effects of exposed shrubs in collecting wind-blown snow in the alpine zone or the development of large drifts in topographically sheltered areas; these effects transform shortwave and longwave radiative exchange above the snowpack and moderate turbulent exchange between the atmosphere and underlying snowpack. Land surface schemes have at best an ad hoc representation of snow cover development and depletion that does not well represent wind redistribution of snow or actual areal albedo decay during melt and results in significant errors in surface energy balance calculations (Pomeroy et al. 1998). Complex mountain



a. Catchment April 1 SWE contribution by shelter class compared to measured SWE by snow pillow on April 1 for each water year. 20 year average April 1 snow pillow SWE (540 mm) shown by black line.

b. Average within shelter class SWE on April 1 for each water year.

c. Cumulative July 1 simulated surface water input by shelter class, and measured stream discharge from the Reynolds Mtn. East weir. Percentages show the proportion of discharge to SWI. 40 year average cumulative July 1 discharge (538 mm) shown by black line.

d. Total vapor flux (evaporation or condensation) from the snowcover, by shelter class. Percentages indicate vapor flux as a proportion of stream discharge for that year.

Figure 3. Snow Water Equivalent (SWE), Surface Water Input (SWI) and Vapor Flux by Shelter Class, Reynolds Mountain East Catchment, Reynolds Creek Experimental Watershed.

terrain includes combined effects due to slope, aspect, terrain shelter, and vegetation structure (Marks and Winstral 2001) that largely control both snow redistribution and drifting during the development of the snowcover, and variable patterns of snowcover energetics during melt. These effects are either poorly described or ignored in regional and global scale climate and hydrological models.

For larger scale models the processes governing snowmelt energetics have been most fully considered for idealized 'plane' surfaces that meet the uniform, level fetch requirements for steady-state atmospheric boundary layer development and for straightforward radiation fluxes as a function of solar angle and latitude. Refinements to these considerations include the calculation of incoming direct and diffuse shortwave radiation on slopes (Garnier and Ohmura 1968), calculation of longwave fluxes from neighboring slopes (Marks and Dozier 1979) and consideration of the effects of vegetation on radiative transfer (Hardy et al. 1997, Link and Marks 1999b, Marks et al. 2001b). Recent progress has considered the important effect of discontinuous snow cover on advective turbulent exchange and snow energy balance (Shook and Gray 1997) but has still employed level terrain assumptions.

Most models of snow energetics, snow hydrology and snow-atmosphere interactions still employ the uniform planar snow cover assumption with the most sophisticated now including modifications for radiation based on solar angle, slope, aspect and skyview (e.g. Marks et al. 1999) and for advection (Shook and Gray 1997). At the catchment scale, over grids of 100m or less, the processes controlling both snow deposition and melt have been fully accounted for in the applications of the ISNOBAL model (Marks et al. 2002), and effectively accounted for over larger areas using grids as large as 250m (Garen and Marks 2003). In considering the usefulness of snowmelt calculations for mountain hydrology, it is important to realize that hillslopes form the most critical and distinctive part of the catchment contributing area. During snowmelt in alpine locations they are particularly important because snow drifts on lee-side slopes provide an inordinately large proportion of basin SWE accumulation (Woo and Marsh 1978, Marks et al. 2002) and contribute to streamflow generation for an extended period after other landscape types have become depleted of snow (Marsh and Pomeroy, 1996). The influence of slope and aspect are very important for hillslope snowmelt calculations as they affect snow accumulation, snowmelt energetics, the resulting meltwater fluxes and runoff contributing area. The impact of slope and aspect on the energetics of snowmelt may need to be considered comprehensively with snow accumulation and snow cover state for a discrete slope within a landscape type because the energy and mass balance processes are coupled through the evolving state of the snow cover and non-snow- covered surfaces. The variability in the microclimate of these slopes and of the atmospheric exchange occurring on slopes is of interest to better understand and describe land-snow-atmosphere interactions during snow deposition and melt in mountain environments. Critical parameters to consider are the spatial variability of snow mass, components of the snowcover energy balance, and the association between these terms. In particular, land cover characteristics and topographic shape of snow deposition areas where large drifts develop and scour areas where wind exposure and scrub height limit the maximum snow depth must be accounted for.

Approach

Research sites

This research effort relies on three world-class experimental basins: Wolf Creek Research Basin (WCRB) in the Yukon Territory (Canada), the Reynolds Creek Experimental Watershed (RCEW) in Idaho, and the Fraser Experimental Forest in Colorado (Fraser) (Figure 4). These sites form a continental-scale transect that is representative of northern cordilleran mountains, semi-arid mountainous rangelands, and high-elevation Rocky mountain regions that comprise the headwaters of western North American river systems. All sites are extensively instrumented, have a significant historical hydrometeorological and vegetation/ topographic data records, and are currently supported by intensive local experiments that provide the basic infrastructure and local collaborators with which to conduct research (USDA in RCEW, CLPX and USFS in Fraser, and CFCAS/Northern Affairs Canada in WCRB).

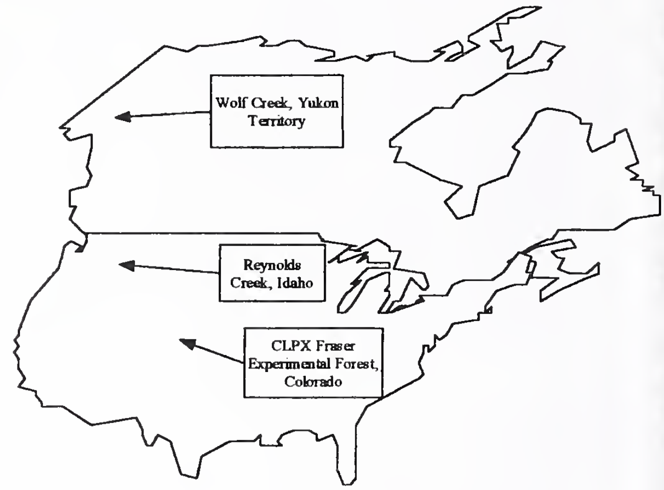


Figure 4. Field experiment sites in the Western Cordillera of North America.

Wolf Creek Research Basin (WCRB), Yukon Territory, Canada

The Wolf Creek Research Basin is located in the northern cordillera of northern Canada near the city of Whitehorse, Yukon Territory at 61° N latitude. The basin is 195 km^2, spans an elevation range from 800 to 2,250 m asl, and contains boreal forest, sub-alpine taiga, and alpine tundra ecosystems. The WCRB provides uniquely simple access to high-altitude sites with a distinct snow season, both forest and shrub-tundra and some degree of complex topography. Whitehorse Airport has a full reference climate and upper air sounding station. WCRB was established as a GEWEX research basin in 1993. WCRB has grown to host a number of hydrological, atmospheric and ecological studies (Pomeroy and Granger 1999) and its physical attributes are well documented with DEM, vegetation and soils maps, NDVI, and LAI. The proponents have established nine hydro-meteorological towers at various elevations/vegetation zones (alpine, shrub tundra, forest) in the catchment that continuously measure WMO standard meteorological variables as well as net and solar radiation, soil temperature, snow depth and soil moisture.

Reynolds Creek Experimental Watershed (RCEW)

The Reynolds Creek Experimental Watershed is located in the Owyhee Mountains of southwestern Idaho, in close proximity to Boise, ID (Marks 2001).

The RCEW is typical of semiarid rangelands that cover large regions of intermediate elevation lands in the western U.S. The RCEW is a 239 km^2 watershed ranging in elevation from 1,101 to 2,241 m asl. Mean annual precipitation ranges from about 230 mm at the lower elevations to over 1,100 mm at the higher elevations, where 75% or more of the annual precipitation occurs as snowfall. A mix of shrub, deciduous forest and coniferous forest communities are found at the more mesic, snow-dominated, higher elevation areas.

RCEW has been operated since 1960 by the USDA Agricultural Research Service, Northwest Watershed Research Center (NWRC). Intensive precipitation measurements are completed throughout the watershed with 28 precipitation sites composed of paired shielded and unshielded gauges. Snow cover and snow water equivalent (SWE) at RCEW is monitored bi-weekly. Meteorological data are collected at 11 sites and include solar and thermal radiation, air temperature, humidity, wind speed and direction, barometric pressure, and soil temperature. Hourly streamflow records are present for nested watersheds ranging from 1 ha to 23,866 ha. Spatial data sets for RCEW include a high-resolution (10 m grid) digital elevation model (DEM), vegetation coverage, soils, and geology.

Fraser Experimental Forest, Colorado

The Fraser Experimental Forest is located approximately 60 miles northwest of Denver, CO on the western slope of the Front Range. Fraser is a topographically complex, forested area, with a mean elevation is 3,066 m asl. Fraser is the site of the Cold Land Processes Field Experiment (CLPX) (Cline et al. 2001), designed to advance our understanding of the terrestrial cryosphere. CLPX is being conducted over an area 4.5° x 3.5° centered over the mountains of western central Colorado. Within this area, nine 25 km x 25 km regions have been selected for intensive snow surveys. During winter and spring of 2002, 2003, and 2004 pairs of surveys of depth and SWE will be conducted each time generating over 560 samples within each of the nine study areas, and over 5,000 over the CLPX region.

This data set will become the primary model testing and verification data set for snow simulation over large regions. Installed as part of the CLPX, the 1-ha Local Scale Observation Site (LSOS) was designed to investigate fine-scale snow processes and properties. The LSOS consists of a dense pine stand and an open pine stand. This site is comprehensively instrumented with full energy and mass balance meteorological stations by the USFS RMS and NASA GSFC and intensively snow surveyed at both fine and long scales.

Coordinated site data collection

Observational strategies at the three sites are harmonized according to an assessment of 'best practices' developed for a number of studies such as GEWEX-GAPP, MAGS, CLPX and ARS watershed hydrology programs at RCEW, to provide consistent, high quality datasets for model development and testing. Automated data collection efforts are coordinated so that key hydro-meteorological variables are measured using comparable equipment at vertical and lateral positions within vegetation canopies, and within similar terrain positions at all three sites. Specific variables measured at all sites include snow depth, snow water equivalent, total solar radiation, diffuse solar radiation, thermal radiation, air and soil temperature, sensible and latent heat fluxes, wind speed and direction, and relative humidity. Spatially intensive manual sampling of snow and vegetation properties is also coordinated at all three sites during a series of focused field campaigns.

Focused field campaigns

Focused field campaigns are designed to improve multi-dimensional process representations of snow dynamics for complex terrain, reduce uncertainty in model runs and assess the transferability of improved small to medium scale models of snow accumulation and ablation. The focused field campaigns provide the following key data: 1) continuous measurements of vertical and horizontal distributions of radiation and climate variables within and above forest and shrub canopies, 2) eddy covariance measurements of sensible and latent heat fluxes above forest canopies and from open snowfields, and 3) periodic, spatially distributed, measurements of snow properties over topographically complex catchments with a variety of shrub and forest covers.

Field experiments at RCEW focus on the shrub, coniferous and deciduous forest zones. Field experiments at WCRB focus on the forest and the shrub tundra zone, and field experiments at Fraser focus on the dense and open coniferous forests at the LSOS site.

Conclusions

This coordinated multi-watershed research effort will permit an assessment of the key fine-scale snow processes operating across the complex cordilleran environments, and the transferability of process upscaling methodologies in the various mountain environments. It will produce datasets that can be used for further basin-scale model validation, suggest methods to deal with the effect of sub-grid variability and provide the inputs needed to suggest faithful representations of snow deposition and ablation at larger scales.

Acknowledgments

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Studies of Scale and Processes in Hydrologic Modeling on the Lucky Hills Watershed

H. Evan Canfield, David C. Goodrich

Abstract

Hydrologic and sediment yield data collected from Lucky Hills 104 within the Walnut Gulch Experimental Watershed have been used extensively to study the effects of scale in the KINEROS model. These studies show that lumping of parameters derived at the small scale into increasingly less complex geometry has the effect of reducing runoff volume, peak and sediment yield. Model simulations mirror observations at the Lucky Hills watersheds showing a decrease in runoff per unit area as watershed size increases. Studies of rainfall variability on model response show that even at <5 ha scale, data from a single gauge cannot adequately describe rainfall input, and can lead to errors in runoff modeling. Studies of watershed representation indicate that runoff volume can be simulated using less complex geometries if equilibrium storage of runoff is maintained. Likewise sediment yield can be simulated using a simplified watershed representation by increasing the entrainment of sediment by raindrop impact on hillslopes even as entrainment by flowing water decreases as fewer channels are represented in the model.

Keywords: hydrologic modeling, semiarid rangeland watersheds, erosion, precipitation, spatial variability

Introduction

In order to develop, calibrate and validate watershed hydrology models, hydrologic data collected at instrumented watersheds are required. This paper presents an overview of data collection at the United States Department of Agriculture, Agricultural Research Service (USDA-ARS) Lucky Hills 104 watershed (Renard et al. 1986), and describes how these data have been used with the KINEROS2 model (Smith et al. 1995) to simulate hydrologic processes on small semiarid watersheds.

Lucky Hills Intensive Study Sites

In 1961, researchers identified two watersheds draining into stock ponds that were thought to be indicative of typical environments on the USDA-ARS Walnut Gulch Experimental Watershed near Tombstone, AZ. One of these, Kendall's pond 20, drained a 128 acre (52 ha) stable grassland watershed. The other, Lucky Hills pond 23, drained a 115 acre (46.7 ha) shrub-dominated, creosote bush and acacia watershed.

In 1961 and 1962, rainfall and runoff monitoring was initiated at these two stock pond watersheds, but data were of limited value in developing rainfall-runoff relationships. "Percentages of the rainfall that appeared as runoff in each of these ponds were so variable, however, that no comparison of the two areas could be made. Owing, probably, to unresolved characteristics of the drainage areas, storms of similar amounts and intensities on the same area produced differing amounts of runoff (Kincaid et al. 1964)."

Therefore, the researchers identified two smaller 'unit-source' watersheds at the upper end of the stock pond watersheds. "A 'unit-source' watershed is defined as a natural drainage area that has relatively homogenous soil and vegetation cover, that is subject to essentially uniform precipitation, and for which any geologic influences on the surface outflow are areally representative (Kincaid et al. 1966)." Two small upland areas were selected for more intensive study. In 1962 the first runoff was measured at Kendall Watershed 112 (4.6 acres, 1.9 ha) and Lucky

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Hills Watershed 101 (3.2 acres, 1.3 ha). In each of these small watersheds, runoff measuring weirs, rain gauges and soil moisture blocks were installed.

The location of Lucky Hills lent itself to paired watershed study, because an adjacent watershed had similar soil and vegetation characteristics. By 1963, a nested watershed had been instrumented at Lucky Hills Watershed 103 (9.1 acres, 3.7 ha.) which contained the Lucky Hills 101 watershed, and drained into the stock pond. A second nested watershed was instrumented at Lucky Hills 104 (11.2 acres, 4.5ha). While Lucky Hills 104 did not drain into a stock pond, the drainage network lent itself to establishing nested subcatchments on the northeast (Lucky Hills 106, 0.85 acres, 0.36 ha) and northwest (Lucky Hills 102, 3.6 acres, 1.46 ha) forks (Figure 1). The Lucky Hills watershed complex was fenced, and there has been no grazing by domestic livestock on this land since 1963 (Osborn and Simanton 1983), though rabbits and other small herbivores may graze on the watershed. Soils on the watershed are mapped as Luckyhills-McNeal Sandy Loam (Ustochreptic Calciorthid) (Breckenfeld et al. 1995).

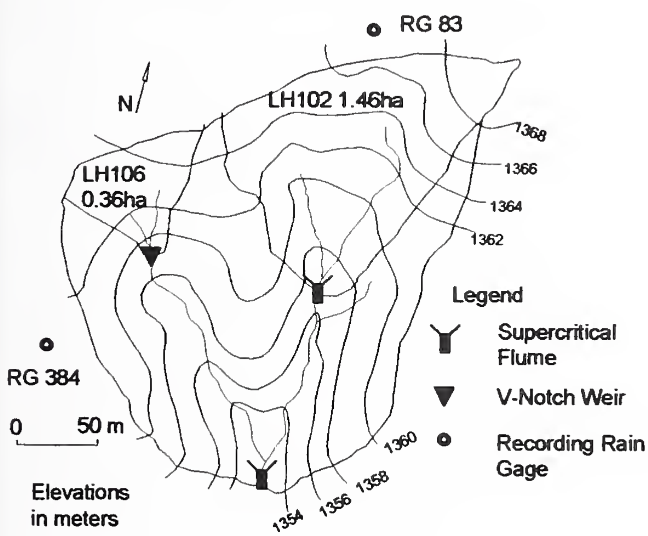

Figure 1. The Lucky Hills 104 showing the locations of the two nested watersheds, and location of current rain gauges and flumes.

Data Collection at Lucky Hills 104

Rainfall and runoff data have been collected at Lucky Hills 104 watersheds since 1963 when Rain gauge 83 and V-notch weirs 104 (LH104) and 102 (LH102) were installed. Rain gauge 384 was added in 1964, and a weir was installed on Lucky Hills 106 (LH106) in 1965.

In 1978 the wier at LH104 was replaced with a supercritical flume and traversing slot sampler (Renard et al. 1986). In 1977 a concrete flume replaced the weir at LH102. This concrete flume was subsequently replaced with a supercritical flume and traversings slot sampler in 1998. Since the instrumentation was installed in the early 1960s, rainfall and runoff data have been collected with only short interuptions for upgrading equipment, which generally occurred during the winter. However, there have also been periods of more intensive sampling, such as during Monsoon '90 (Kustas and Goodrich 1994), when a dense rain gauge and soil moisture monitoring network were installed. Soil moisture changes with depth were also monitored at six locations under shrub and bare conditions (Canfield and Lopes 2000, Hymer et al. 2000).

Sediment data are more difficult to collect and prone to sampling errors, so sediment data are not available for many events for which rainfall and runoff data are available. Intially, coarse particle load was measured after each event by removing and weighing the sediment in approach boxes behind the wiers. Integrated depth pump samplers were added in 1973 to collect suspended sediment samples. The traversing slot sampler, added to LH104 in 1978 and to LH102 in 1998, was developed to collect depth integrated samples for use in computing total load discharge through an event. The use of different sampling equipment has allowed reasearchers to asses the viability of different sampling methods using data from the Lucky Hills (Simanton et al. 1993)

There have been several efforts to characterize the topography of the watershed. A five-foot contour map, from field survey, was used in the first papers describing research at the watershed (e.g. Osborn and Lane 1969). A topographic map was prepared from a 1975 areal survey which resulted in a 1' contour map of the watershed (e.g. Faures et al. 1995), which was the basis for watershed characterization in many of the studies of watershed complexity and model response. A 2.5 m x 2.5 m DEM was prepared based on field survey, and used to relate to soil variability to topographic characteristics (Canfield 1998). The relationships between topographic characteristics and soil characteristics could then be used to parameterize the spatial variability of infiltration and soil erosion parameters (Canfield and Goodrich 2000, Canfield et al. 2001).

Plot scale studies have been conducted to determine the hydrologic impact of reduction of canopy cover and the increase of grass cover (Kincaid and Williams 1966, Schreiber and Kincaid 1967), a desirable outcome for management of rangelands. There was also a largely failed attempt to convert all of Lucky Hills 106 (0.36 ha) and Lucky Hills 102 (1.46 ha) from brush to grass (Woolhiser et al. 1990) using herbicides and physical treatments.

Hydrologic Process Studies and Model Development

Analysis of runoff data collected during the first two years of operation, 1963 and 1964, showed that anywhere from 50% to 150% more runoff was generated per unit area on 6'x12' plots than from the small watersheds (Kincaid et al. 1966). Researchers recognized that some of this difference could be attributed to watershed characteristics, and some could be attributed to spatial variability of precipitation. Therefore, early analyses of watershed data showed that even at the 'unit-source' scale, watershed chacteristics and spatial variability of precipitation affect hydrologic response.

Because of the length of record, and the density of the rainfall and runoff data, the Lucky Hills 104 data has become important for studying hydrologic process in the semiarid desert southwest. Early on, researchers were able to develop regression relationships to relate rainfall to runoff on the small watersheds at the Lucky Hills (Osborn and Lane 1969). Data from Lucky Hills have been used to validate USLE (Renard et al. 1974), as well as a distributed version of RUSLE using data in a GIS (Yitayew et al. 1999). In addition, the availability of these detailed data have made it possible to validate discriptive hydrologic models that describe the temporal and spatial variability of rainfall, runoff and sediment yield on semiarid watersheds. Specifically, the KINEROS2 model (Smith et al. 1995) has been used to describe, model and better understand hydrologic processes on the Lucky Hills 104 watershed.

The KINEROS2 Model

The KINEROS2 model is a distributed runoff-erosion model based on Hortonian overland flow theory, and, therefore, well-suited to describing the hydrodynamics of runoff and erosion on semiarid watersheds, where infiltration rates are low, and rainfall is infrequent but intense. The model allows for spatial variable rainfall input, channel transmission losses, and spatial variability of watershed characteristics such as soils, slopes and vegetation.

KINEROS2 is particularly well suited for modeling rangeland environments, because it can be parameterized to describe the variability of infiltration on hillslopes, which is best described as a distribution of values (e.g. Paige et al. 2002). The developers of KINEROS2 recognized that the distribution of infiltration can be described according to a mean and coefficient of variation (Woolhiser and Goodrich 1988). This distribution of infiltration more realistically describes the partial area response seen on semiarid rangeland watersheds such as Lucky Hills.

Runoff is treated in KINEROS2 with a one-dimensional continuity equation applicable to both overland and channel flow. Sediment entrainment and transport on hillslopes and channels is treated as an unsteady, one-dimensional convective transport phenomenon, using a continuity equation similar to that for runoff. Sediment flux on a hillslope has two independent sources, raindrop-induced entrainment and flow-induced entrainment. Watershed geometry is represented in KINEROS2 as a combination of overland flow plane and channel elements, with plane elements contributing lateral flow to the channels or to the upper end of first order channels (Figure 2).

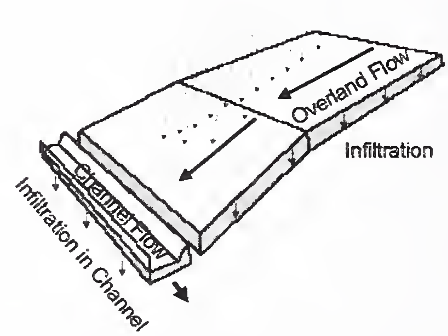

Figure 2. A schematic representation of the geometric representation of hillslopes and channels in the KINEROS2 model as well as the runoff processes simulated in the model.

Each plane may be described by its unique parameters, initial conditions and precipitation inputs. Each channel element may be described by its unique parameters as well. By allowing parameters and rainfall to vary spatially, KINEROS2 can take advantage of the wealth of spatial and temporal data from Lucky Hills to better describe and understand the effects of rainfall and watershed characteristics on runoff and sediment yield.

Results of Modeling Studies

The KINEROS2 model has been used to study the effect of rainfall spatial variability on runoff; the effect of soil moisture on runoff; the effect of geometric complexity on model response for runoff and erosion; and the effect of changes in vegetative cover on runoff.

Measuring rainfall and modeling runoff

Researchers recognized early that multiple rain gauges were necessary in order to capture the spatial distribution of rainfall (Kincaid et al. 1966). As such, there have always been at least three recording rain gauges to measure precipitation at the Lucky Hills 103 and 104 watersheds. While monsoon rainfall had been known to be highly spatially variable, the experiments performed during Monsoon '90 experiment (Kustas and Goodrich 1994) allowed hydrologists to better understand the spatial variability of monsoon rainfall at scales less than 5 ha, and its effect on infiltration and runoff. A monitoring network was devised to better sample the spatial variability of rainfall on Lucky Hills 104 using a series of non-recording and recording rain gauges (Goodrich et al. 1995). Forty-eight of the non-recording gauges were located on a 30 m grid across the watershed. Nine recording rain gauges with collocated non-recording gauges were located in areas of homogenous slope and orientation. Three vectopluviometers were employed to determine the orientation of rainfall. Analysis of these data using statistical and geostatistical methods indicated that:

- Total error for point rainfall measured from a non-recording gauge is 4-5% for storms greater than 15 mm, and greater than this for storms smaller than 15 mm.
- There were gradients in total rainfall depth which represented 4-14% error over 100 m. Therefore, a single gauge is inadequate for monitoring total rainfall depth, even on a watershed as small as Lucky Hills 104.
- The rainfall intensity variation did not change greatly across the catchment, so a single recording rain gauge is adequate for capturing temporal variability at the < 5 ha scale.

A follow-up study to the rainfall monitoring determined the relative impact of changes in rainfall on runoff modeling using the KINEROS model (Faures et al. 1995). This study found that:

- Assuming rainfall is uniformly distributed in space can lead to large errors in modeled runoff peak and volume.
- Using a single recording rain gauge and four well-distributed non-recording gauges, the sampling resolution approached the resolution of the 60-gauge dense rain gauge network, and model-estimated peak and volume were within 10% of the dense network values.

These two studies indicate that non-recording rain gauges can sample the spatial variability of rainfall at the < 5 ha scale, and that a single recording gauge can capture the temporal variability adequately for model estimates.

Soil moisture monitoring data collected in Monsoon '90 showed that soil moisture is spatially and temporally variable on the watershed (Whitaker 1993). A sensitivity analysis of soil moisture measurement methods showed that runoff estimates from KINEROS on the Lucky Hills are relatively insensitive to different methods of soil moisture estimation (Goodrich et al. 1994). This implies that model predictions will not likely be improved by improving the soil moisture estimate.

Watershed representation and its effect on runoff and erosion modeling

Because so much spatial data are available for the Lucky Hills, the data set lends itself to the study of spatial complexity and model response. Early researchers attributed the reduced runoff per unit area noted at the watershed, in part, to the impact of channels and depression storage (Kincaid et al. 1966). In model terms, the reduced runoff per unit area will depend both on the model parameters and the complexity of the channel network used in the model. KINEROS2 simulations at Lucky Hills 104 showed that runoff peak and volume decrease systematically as the watershed is represented with increasingly fewer channels (Lopes and Canfield (in press)). Goodrich et al. (1997) were able to show that equilibrium storage of runoff on a watershed can be maintained as the watershed is represented with increasingly fewer channels. These researchers assumed an impermeable surface, and showed how roughness could be increased as subcatchments were represented as a single overland flow plane. Using this method, he was able to show that runoff volume could be predicted without significant degradation (Nash-Suttcliffe statistic >0.9), from a complex representation with 263 elements draining hillslopes of a few hundred square meters, to a representation of 17 elements draining hillslopes of larger than 5000 square meters.

In contrast, as fewer channels are represented in KINEROS2, sediment yield is largely a function of event characteristics. For events with approximately a 2 year return period sediment yield was approximately the same for watershed representations with from a 312 element model draining hillslopes of about 200 square meters to an 18 element model draining hillslopes greater than 5,000 square meters (Lopes and Canfield, (in press)). However, runoff decreased systematically with less complex model representations for these events. In contrast, for events of longer than two-year return periods, sediment yield is affected by watershed representation with less complex representations underpredicting sediment yield, presumably because channels are a more important source of sediment for large events. Studies of model representations of erosion in the KINEROS2 model on the Lucky Hills 104 have shown that as the model representation becomes more simplified, sediment entrainment can be maintained by increasing the contribution from splash erosion at the expense of entrainment by flowing water (Canfield et al 2002). However, model predictions become less accurate for less complex model representations.

Conclusions

While scale effects and spatial variability in hydrology are widely recognized as problems that need to be addressed, there are limited data sets for studying these problems. Studies from the USDA-ARS Lucky Hills 104 Watershed at Walnut Gulch have been used to better understand the spatial variability of rainfall, the spatial variability of soil moisture, partial area response, and spatial variability of soils and infiltration characteristics. By using the data from the Lucky Hills 104 to parameterize the KINEROS2 model, researchers have been able to better understand these hydrologic processes and the effects that sampling may have on model predictions.

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Rangeland Ecological and Physical Modeling in a Spatial Context

S.M. Skirvin, M.S. Moran

Abstract

The Simulation of Production and Utilization of Rangelands (SPUR) model has been in use and revision since 1987 in diverse rangelands including Texas and the Great Plains. The model's applicability to semi-arid rangelands of the southwest is under evaluation at the USDA-ARS Southwest Watershed Research Center in Tucson, Arizona. As part of this effort, a spatially explicit implementation of SPUR (SESPUR) has been developed with the long-term goal of incorporating distributed hydrologic information from GIS-based hydrologic models such as the Soil and Water Assessment Tool (SWAT) and KINEROS, as well as remotely sensed soil moisture. Ease of model use has been improved with the ability to display output with a GIS, and linkage to a database of input parameters will improve the user's experience. Validation of the updated model is in progress with data obtained at the Walnut Gulch Experimental Watershed near Tombstone, Arizona. The original focus of SPUR included prediction of hydrologic and erosion changes resulting from management decisions, as well as simulation of forage growth and its utilization by grazing animals. Its usefulness for other biophysical and ecological modeling has been substantially enhanced with the SESPUR spatial implementation.

Keywords: SPUR, spatial modeling, rangeland ecology, hydrologic modeling

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Introduction

Rangelands form an important part of the natural resources of the western United States. There is growing interest in detailed inventories of rangeland ecosystems and physical processes, such as carbon sequestration and release (Hanson et al. 2001). Computer simulation of rangeland systems provides a means to integrate existing data and understanding of these areas and processes and to extrapolate beyond present conditions, in the generation of "what-if" scenarios that can aid stakeholders in land management decisions.

The SPUR model (Simulation of Production and Utilization of Rangelands) has been under continuing development since its initial release in 1987 (Wight and Skiles 1987). SPUR contains components that model plant growth, carbon and nitrogen cycling, soil moisture flux, surface hydrology and erosion, foraging by wildlife, and economics of beef production. There have been ongoing efforts to incorporate improved understanding of landscape processes and to make model components interactive during simulation runs, allowing dynamic changes in system states (Carlson and Thurow 1992, Hanson et al. 1992, Foy 1993, Carlson and Thurow 1996, Foy et al. 1999, Pierson et al. 2001).

SPUR climate driving variables include daily precipitation and temperature maximum and minimum. Additional driving variables that are more difficult to obtain include incoming solar radiation, wind velocity and direction, and dew point. These parameters can be simulated using a routine such as CLIGEN, a stochastic weather generator developed for the WEPP hydrological model (Flanagan and Livingston 1995). SPUR runs on a daily time-step, with output summaries available monthly, annually and at other user-set intervals. Model results comprise text reports of vegetation, soil, hydrologic, and grazing animal numeric outputs (Carlson and Thurow 1992).

Because SPUR is point-based, it is applicable only to small homogeneous areas at the scale of a pasture or smaller, with a basic modeling unit of one soil and one vegetation type. In regions such as the Great Plains these conditions may be applicable over larger areas, but in the southwestern U.S. climate and landscape are characterized by high spatial variability. Such variability suggests that point-based model results may not generalize well over areas of any significant size in this region. Even at pasture scale, differential use of areas by grazing animals can cause spatial heterogeneity that is not well represented by a uniform field (Foy et al. 1999, Teague and Foy 2002).

Recent work at the USDA-ARS Southwestern Watershed Research Center (SWRC) has included development of a raster-based, spatially explicit implementation of SPUR (SESPUR) that can simulate areas composed of mosaics of soils, vegetation, and topography. Model parameterization, calibration and validation will use the Walnut Gulch Experimental Watershed as a study area, with historical data sets and ongoing studies providing data to evaluate SESPUR under southwestern semi-arid conditions. Descriptions of changes to the original SPUR model and initial results from the prototype spatial implementation are presented here.

Methods

The most recently published SPUR upgrade, SPUR2000 (Pierson et al. 2001), was used as the basis for SESPUR. Extensive Fortran code modification was required to allow simultaneous simulation of areas with differing soils, vegetation, and topography including slope and aspect. Additional subroutines were added to handle spatial data processing. Spatial data for input are converted to ASCII text files outside SESPUR and read by the program, and output text files are formatted for import into a geographic information system (GIS) for display and further analysis.

A 70 by 70 meter test area within the Kendall sub-watershed of Walnut Gulch was selected for prototype application development. The Kendall area is primarily grassland, with scattered shrubs, forbs, and small mesquite trees. Some spatial input data for the test area were obtained from GIS layers developed at SWRC for Walnut Gulch, including the 1993 soil survey digital map (Breckenfield n.d.) and a 10 meter resolution digital elevation model (DEM). The test area had two soil units and a variety of slopes and aspects (Figure 1).

Detailed soils information, including layer structure and properties, were obtained from the soil survey and linked to the soils spatial data. Some soil parameters were estimated from textural information (Cosby et al. 1984). A mix of modeled vegetation types–including C4 mid-height grasses, C4 short grasses, annual forbs, perennial forbs, sub-shrubs and shrubs–was assumed to be distributed uniformly in the test area. Vegetation data were acquired in field surveys in 2003, as existing vegetation maps did not provide adequate detail for model needs. Vegetation parameters included species or functional group composition, abundance, cover, and species- or group-specific physiological parameters. Many physiological parameters had to be estimated from the literature (e.g. Carlson and Thurow 1992, Larcher 2003), as detailed information is not available for many southwestern species. Topographic data including slope and aspect were derived with GIS functions from the 10-meter DEM. Climate data were simulated using CLIGEN, based on a CLIGEN-formatted summary file of long-term means and averages of climate parameters for the Tombstone climate station. A one-year simulation was performed for the test area.

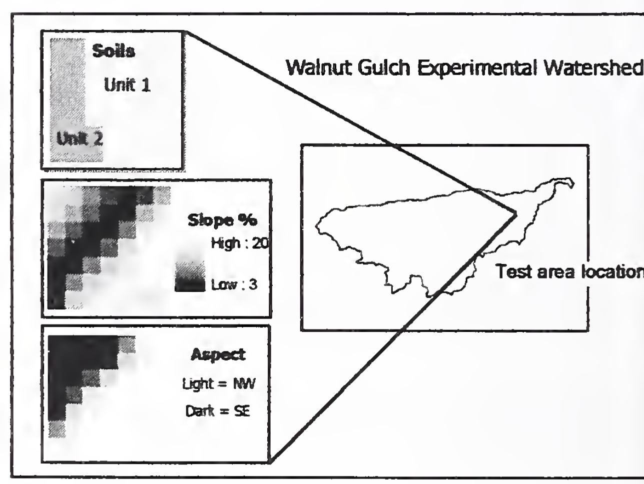

Figure 1. Spatial data for Kendall test area. Test area cell size = 10 meters; total size = 70 x 70 meters.

Results

SESPUR results for one vegetation variable, total monthly biomass production, are shown as a time series in Figure 2. Sensitivity of this variable to soil properties is suggested by the coincidence of higher production with the soil unit on the left side of the test area. This may indicate a need for adjustment of soil properties, as such large differences in production over a relatively small area are rarely seen in the field in this area. An array of additional outputs is available on a daily or monthly basis.

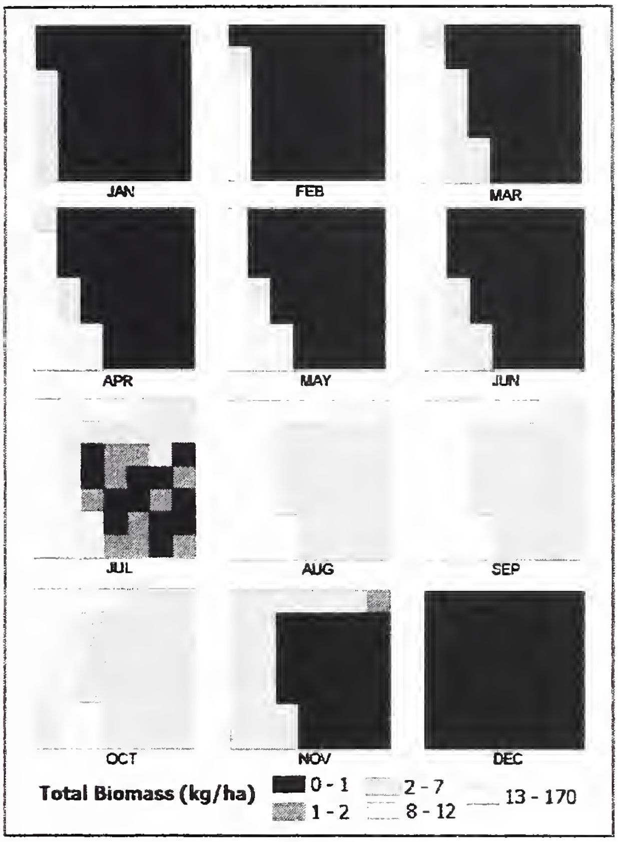

Figure 2. Time series of total monthly biomass production from SESPUR simulation.

The timing of net vegetation peak production in July (not clearly visible in Figure 2) shows a very strong influence of monsoonal precipitation on modeled vegetative growth. However, peak production generally occurs closer to September in this area, also indicating the need for parameter adjustment. SPUR was developed for areas with a well-defined growing season demarcated by last and first frost, and is known to perform less well in warm regions where extended or multiple growing seasons are possible (Carlson and Thurow 1992). This may be an area for future software modification, perhaps using a degree-day and soil moisture combined index to define growing season(s) during the year. Comparison of biomass production results with limited field data from May 2001 suggest model underestimation by at least one order of magnitude, but more field data will be needed to test this result.

Conclusions

The spatial implementation of SPUR opens up many possibilities for more accurate simulation of rangeland processes. The spatial distribution of daily vegetation growth and its impact on soil moisture may provide a valuable adjunct to a variety of physically based models. These include hydrologic models that focus on spatial routing of precipitation runoff, such as SWAT and KINEROS; and soil-vegetation-atmosphere (SVAT) models that examine overall water, carbon and energy balances. Increased understanding of other issues such as the shift to woody plant dominance in southwestern rangelands, and its implications for carbon cycling (Archer et al. 2001), will be facilitated by this kind of spatially detailed modeling.

The SESPUR model's potential is associated with a significant burden of data needs for parameterization, calibration and validation that may be complicated by the addition of spatial variability. It is anticipated that the model can be simplified and customized for southwestern rangelands, thus reducing data requirements. Immediate research data needs include a map of vegetation structure or composition with fractional cover estimates (i.e. percent woody vegetation cover and percent grass cover). Field data compiled during the 2003 growing season will be compiled with existing vegetation production data for further calibration and validation. Modeled soil moisture will be validated against in-situ measurements from the Walnut Gulch sensor network and estimates from remotely sensed data (see Bryant et al., this volume).

Significant effort is also presently required simply to assemble data for model runs. A user-friendly GIS-based interface is under development, which will assist in model parameterization and visualization of results. The improved interface and simplification of data requirements will support our efforts to make SESPUR a decision support tool that can be used by southwestern rangeland stakeholders as well as researchers.

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Combined Geomorphic and Numerical-Modeling Analyses of Sediment Loads for Developing Water-Quality Targets for Sediment

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Abstract

The principle objective of the study was to determine sediment loads for James Creek, Mississippi and for similar, but stable "reference" streams to develop water-quality targets for sediment. "Reference" sediment-transport loads were determined from stable streams with historical flow and sediment-transport data in the Southeastern Plains Ecoregion. Using the discharge that occurs, on average every 1.5 years ($Q_{1.5}$) as the "effective discharge," an initial "general reference" of 0.31 T/d/km^2 was obtained. This value, however, is skewed towards streams with sand beds and does not accurately reflect conditions along James Creek. A refined "reference" condition was developed for stable silt/clay-bed streams in the Southeastern Plains resulting in a "reference" suspended-sediment yield of 3.23 T/d/km^2 at the $Q_{1.5}$. A weighted-reference condition based on the percentage of the drainage area encompassed by the various bed-material types results in a reference yield at the $Q_{1.5}$ of 2.2 T/d/km^2. Similarly, a weighted-reference concentration of 160 mg/l was obtained. "Actual" sediment-transport loads were obtained by: simulations of flow and sediment transport using the model AnnAGNPS and by simulations of channel flow and sediment transport by the channel-evolution model CONCEPTS. Average sediment loads at the mouth of James Creek over the 35-year period are about 250,000 T/y with 88% emanating from channels and 12% from upland sources. This loading value, however, is somewhat misleading in that severe channel erosion occurred between 1967-1968 following channel clearing and snagging over the lower 17 km. Since this time, sediment loads attenuated and the contribution from channels and uplands over the period 1970-2002 shifted to 70% and 30%, respectively. "Actual" simulated suspended-sediment loads at the $Q_{1.5}$ show a 35-year average of 675 T/D/km^2; 155 T/D/km^2 over the past 10 years. Following the installation of low-water crossings in 1999 loads decreased to about 39 T/D/km^2. This value is more than an order of magnitude greater than the "reference" yield.

Keywords: sediment transport loads, reference conditions

Introduction

The 1996 National Water Quality Inventory (Section 305(b) Report to Congress) indicates that sediments are ranked as a leading cause of water-quality impairment of assessed rivers and lakes. The maximum allowable loadings to, or in a stream that does not impair designated uses has been termed the "TMDL" (total maximum daily load). Three segments along James Creek, Mississippi are listed as having impaired conditions for aquatic life support due to sediment. The Mississippi Department of Environmental Quality (MDEQ) seeks a percent

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reduction in sediment loads such that the James Creek watershed is producing sediment at rates commensurate with those in a biologically unimpaired stream. This unimpaired stream is thus termed a "reference" stream or reach.

Because there are no historical sediment-transport data or "reference" reaches for James Creek, alternative methods are required. Additionally, the sediment-transport data must be expressed in the same form as those developed for reference conditions. To accomplish these tasks a combination of empirical and numerical techniques are used. Suspended-sediment loads from typical streams in the region with historical data can be analyzed by relating the geomorphic conditions at those streams with the conditions along James Creek (Simon et al. 2002). Water and sediment contributions from uplands areas can be obtained with the watershed simulation model AnnAGNPS (Cronshey and Theurer 1998). These data are supplied as boundary conditions for the channel-evolution model CONCEPTS (Langendoen 2000), used to determine channel contributions from main stem streambeds and banks.

A "reference" sediment loading condition can be defined as a concentration (in milligrams per liter; mg/l), load (in metric tons per day or year; T/d or T/y) or yield (in tons per day per square kilometer (T/d/km^2) representative of "natural," stable, or non-impaired conditions. For James Creek and in the absence of a stable channel analog within the watershed, data from similar watersheds in the Southeastern Plains (Ecoregion 65) must be used. "Reference" sediment-transport conditions are established by (1) empirically determining sediment loads for the Southeastern Plains streams using historical flow and sediment-transport data; (2) determining the relative stability of each site where historical data is available; and (3) determining sediment loads for stable and unstable sites segregated by dominant bed-material size class.

"Actual" sediment loading in James Creek can be defined as the amount of sediment that is being transported through and out of the watershed outlet. Because no historical data on sediment transport is available for James Creek empirical and numerical-simulation models are used. To characterize the "actual" sediment load in James Creek, field and digital data are required as inputs to run the simulation models AnnAGNPS and CONCEPTS. The simulation period 1967-2002 was selected because this period coincides with periods of measured channel cross sections.

General description of AGNPS modeling technology

The Agricultural Non-Point Source Pollutant (AGNPS) watershed simulation model (Bingner and Theurer 2001) is a tool to evaluate pollutant loadings within a watershed and the impact farming and other activities have on pollution control. To run AnnAGNPS, daily climate information is needed to account for temporal variation in the weather. The spatial variability within a watershed of soils, landuse, and topography, is accounted for by dividing the watershed into many homogeneous drainage areas. These simulated drainage areas are then integrated together by simulated rivers and streams, which route runoff, sediment and pollutants from each area downstream. Flow and sediment generated by AnnAGNPS can then be input into CONCEPTS as a boundary condition.

The type of landuse assigned to each AnnAGNPS cell was determined using the AGNPS ArcView interface procedure. This procedure assigned a landuse to each cell based on the predominant land use from the land use GIS layer. There are 41 soil types identified from the soil GIS layer in the James Creek watershed. Silty-clay and silt-loam soils dominate the basin. Most of the soils information was derived from the NRCS Soils 5 database. Individual event information describing measured precipitation for the years 1967-2001 was obtained from the Aberdeen, Mississippi National Weather Service climate station located within the James Creek watershed.

General description of CONCEPTS modeling technology

CONCEPTS simulates unsteady, one-dimensional flow, transport of cohesive and cohesionless sediments in suspension and on the bed selectively by size class, and bank-erosion processes (Langendoen 2000). Hence, it can predict the dynamic response of flow, sediment transport and channel form 'channel evolution' to disturbances including channelization, altered hydrologic regime, or in-stream hydraulic structures. The model assumes streamflow to be one-dimensional along the centerline of the channel.

The model calculates total-load sediment-transport rates by size fraction from a mass conservation law, and by taking into account the differing processes governing entrainment and deposition of cohesive and cohesionless bed material (Langendoen 2000). For graded bed material, the sediment transport rates depend on the bed material composition. Following Hirano (1971), CONCEPTS divides the bed into a surface or active layer and a subsurface layer. For cohesive materials, the erosion rate is calculated by an excess shear-stress approach (Hanson and Simon 2000) while the deposition rate is calculated following the method of Krone (1962).

Channel-width adjustment is simulated by incorporating the physical processes responsible for bank retreat: (1) fluvial erosion of bank-toe material, and (2) mass-failure by gravity (Simon et al. 1999, Langendoen 2000). CONCEPTS accounts for streambank stratigraphy by allowing variable geotechnical properties to be assigned to bank materials. Bank stability is analyzed via the limit-equilibrium method. CONCEPTS performs stability analyses of planar slip failures and cantilever failures of overhanging banks by dividing the bank into slices, and evaluating the balance of forces on each slice in vertical and horizontal directions. The slope of the failure surface is defined as that slope for which the factor of safety is a minimum.

Methods

Channel surveys

Channel-geometry data surveyed in 1967 are available at 10 cross sections along James Creek and were used as inputs for the initial 1967 CONCEPTS simulations. Additional sections were synthesized between those surveyed in 1967 based on the average top width and channel depth of the adjacent measured cross sections. A total of 47 cross sections were surveyed along James Creek in 2002 to establish current channel geometry and to provide a means of directly comparing 1967 channel geometries. Bed erosion of about 2 m occurred between the mouth of James Creek and about rkm 10.5 over the 35-year period. This implies that tributaries entering the main stem in the lower 10.5 km experienced up to a 2 m overfall and much steeper slopes at their mouths resulting in re-incision. Bed erosion attenuated from 2 m to negligible amounts from rkm 10.5 upstream to the structure at rkm 12.3.

Streambed erodibility and composition

CONCEPTS requires information on the relative resistance of streambed materials for calculations of sediment entrainment and transport. For cohesive streambeds, a submerged jet-test device is used to estimate erosion rates due to hydraulic forces (Hanson 1990, Hanson 1991, Hanson and Simon 2001). A critical shear stress ($\square_c$) for the material is calculated from field data as that shear stress where there is no erosion. The rate of erosion $\square$(m/s) is assumed to be proportional to the shear stress in excess of $\square_c$ and is expressed in terms of an erodiblity coefficient (k). k is obtained in the field or can be estimated as a function of $\square_c$ (Hanson and Simon 2001). $\square_c$ for sites along James Creek characterized by streambeds of sand and gravel are based on the Shields criteria derived from streambed samples and particle counts.

Streambank stability

Bank-toe materials are composed predominantly of cohesive materials inter-mixed with sand. The submerged jet-test device (modified to operate on inclined surfaces) is used to determine values of $\square_c$ and k. To determine the resistance of cohesive materials to erosion by mass wasting, data is acquired on those characteristics that control shear strength; cohesion, angle of internal friction, pore-water pressure, and bulk unit weight. Cohesion and friction angle data are obtained with a borehole shear-test (BST) device (Lohnes and Handy 1968, Thorne et al. 1981, Simon 1989). The BST provides, direct, drained shear-strength tests on the walls of a borehole.

Texture of bed and bank materials

CONCEPTS uses information on sediment texture to determine sediment routing and sorting processes. Bulk samples of streambed and bank materials were collected at the 17 sampling sites to be analyzed for particle-size distributions. Although James Creek is considered to have a fine-grained streambed several sub-reaches are dominated by sand and gravel. Downstream reaches are dominated by gravel transitioning to sand through rkm 12 to 13, indicating depositional conditions. The reach between rkm 13 and rkm 23 is erosional with beds dominated by fine-grained materials. Average composition of the bank materials is 12% sand, 37% silt, and 51% clay.

Developing a "reference" sediment-transport condition

To determine the amount of sediment that impacts a given stream, one must first determine the sediment load in an un-impacted stream of a given type and location. To define this "reference condition" the scheme used in this study relies on the channel evolution framework set out by Simon and Hupp (1986) and Simon (1989), with stages I (pre-modified) and VI (re-equilibrated) used as stable morphologies.

Analysis of the impacts of suspended sediment requires a database of suspended-sediment concentrations with associated instantaneous water discharge. Data of this type permit development of rating relations (Glysson 1987). The USGS has identified more than 2,900 sites nationwide with at least 30 matching samples of suspended sediment and instantaneous flow discharge have been collected (Turcios and Gray 2001); 148 sites in nine states are in Ecoregion 65: the Southeastern Plains region James Creek is in. A suspended-sediment transport rating is developed for each of the 148 sites by plotting discharge versus concentration in log-log space and obtaining a power function by regression.

Because the "effective discharge" is that discharge or range of discharges that transport the most sediment over the long term it serves as a useful indicator of regional suspended-sediment transport conditions. In many parts of the United States, the effective discharge is approximately equal to the peak flow that occurs on average, about every 1.5 years ($Q_{1.5}$; e.g. Andrews and Nankervis 1995).

Suspended-sediment yields at the $Q_{1.5}$ were calculated for each site in the Southeastern Plains, and geomorphic assessments were carried out at 97 sites in the ecoregion. "Reference" stage I sites were found at 15 while 33 sites were characterized as stage VI. Data from the 48 "reference" sites were separated from those characterized as unstable to create sediment-transport distributions representing unstable and "reference" sites. The median value for stable sites is termed the "general reference" (0.3 T/d/km^2; 48 mg/l at the $Q_{1.5}$).The distributions are heavily influenced by sand-bed streams, representing the majority of the studied sites.

The central 50% of the reference distribution provides a "general reference" load at the mouth of James Creek of between 18.9 and 114 T/d at the effective discharge. The central 50% of the distribution for unstable sites in the Southeastern Plains ranges from 0.34 to 17 T/d/km^2 at the effective discharge.

Refinement of estimates of "reference" sediment discharge

The data set for both unstable and stable sites was sorted by dominant bed-material size class: gravel, sand, and silt-clay. "Reference" suspended-sediment yields for gravel-, sand- and fine-bed streams are 0.27, 0.42, and 3.2 T/d/km^2, respectively. The best estimate of the "reference" suspended-sediment yield or concentration for James Creek should be based on weight-meaning of the reference- parameter values. Utilizing the particle-size data we can identify those reaches that are dominated by the major textural size classes (gravel, sand and fines) and determine the percentage of the drainage area that is encompassed by those reaches. By assuming that tributaries entering the main stem have the same bed-material characteristics as the trunk stream we find that 65% is silt and clay, 21% is sand and 14% is gravel. The resulting "reference" values are 2.2 T/d/km^2 and 160 mg/l at the $Q_{1.5}$ or the effective discharge. Again if we multiply the reference yield by the drainage area of James Creek we obtain a "reference" load at the outlet of about 250 T/d at the $Q_{1.5}$.

Results from evaluations of "actual" sediment loading

Results from AnnAGNPS provide loadings data from gullies, fields and tributaries. Direct comparison of measured cross-sections between 1967 and 2002 provide strong evidence of channel contributions over the period. These data are compared with simulated channel contributions over the same reach and time period by CONCEPTS. Together, the AnnAGNPS and CONCEPTS simulations provide total loadings values for the James Creek watershed. AnnAGNPS simulations using a scenario of unstable tributaries indicated bed and bank erosion nearly seven times greater than the sediment produced from fields. The total sediment load simulated by AnnAGNPS at the outlet of James Creek for reduced tillage and indicated unstable reaches was 110,000 T/y.

Channel erosion 1967-2002: Measured changes in channel geometry

The area between the 1967 bed profile and the 2002 bed profile represents the amount eroded from the channel bed in m^2. On average, about 12% of the materials eroded from the channel came from the channel bed, with 88% coming from the banks. Over the period about 624,000 m^3 of channel sediments were eroded from James Creek between river kilometers 0.27 and 17.2. This converts to 1,136,000 tons (T) using the average saturated density of 1,820 kg/m^3 and an average-annual load of eroded channel materials of 32,500 T/y over this reach or 1,910 T/y/km or 1.91 T/y/m of channel.
To compare measured channel erosion with that simulated by CONCEPTS a shorter reach is used: river kilometers 7.29 to 17.3. Using the same techniques and conversion factors as previously we obtain the following erosion values for this approximate 10 km reach: 252,000 m^3; 459,000 T; 13,100 T/y; 1,320 T/y/km; and 1.31 T/y/m.

Combined AnnAGNPS and CONCEPTS simulation of main channel evolution and transport rates: 1967-2001

CONCEPTS was used to simulate channel hydraulics and morphology of James Creek between rkm 7.29 and 24.02. In the first 10 years, large amounts of sediment were eroded above at the upper end and upstream of the 1967 clearing and snagging work. The simulated thalweg profile is in good agreement with the 2002 measured profile ($r^2 = 0.99$). Up to three meters of incision in the upstream reach initiated mass-bank instabilities. Average sediment loads at the mouth of James Creek over the 35-year period are about 250,000 T/y with 88% emanating from channels and 12% from upland sources. This loading value, however, is somewhat misleading in that severe channel erosion occurred between 1967-1968 following channel clearing and snagging over the lower 17 km. Since this time, sediment loads attenuated and the contribution from channels and uplands over the period 1970-2002 shifted to 70% and 30%, respectively.

Simulated sediment-transport rates and yields

CONCEPTS provides detailed concentration and load estimates at 10-minute time intervals that are associated with discharge values to produce sediment-transport relations for each of the modeled cross sections. To provide a comparison between "actual and "reference" suspended-sediment loads we subtract the gravel portion from the total transport rate. Sediment-transport results are provided where simulated results are validated with measured flow and channel geometry data and, where reference sediment-transport conditions are established. CONCEPTS sediment-transport output was sorted by year and individual sediment-transport relations were derived for each year of simulation. Suspended-sediment loads and yields were calculated by substituting the $Q_{1.5}$ (86.8 m^3/s) into each equation. Yields are shown to approach the "reference", and the minimum values shown by the last data points in the time series represent attenuation of sediment yields, and are representative of relatively consistent sediment transport rates over the last three years (1999 – 2001). These lower rates of sediment transport beginning in 1999 can be attributed to the installation of several LWCs that year whose effects are reflected in the CONCEPTS simulations. Thus, we can state with greater certainty that the "actual current" suspended sediment yield at the $Q_{1.5}$ is in this range. Taking the average for the most-recent three-year period gives an "actual current" yield of 38.9 $T/d/km^2$ at the $Q_{1.5}$, which is still an order of magnitude greater than the "refined-reference" yield.

Conclusions

A combination of geomorphic and numerical-simulation analyses (AnnAGNPS and CONCEPTS) are shown to be powerful tools in evaluating the severity of sediment-transport conditions in James Creek, Mississippi. To develop water-quality targets for sediment in James Creek and in the absence of sediment-transport data in the watershed, historical flow and sediment transport data from similar streams in the Southeastern Plains were used to develop "reference" or un-impacted sediment-transport rates. These values are expressed in terms of the $Q_{1.5}$, or effective discharge. A "refined reference" yield was obtained by sorting the data by dominant bed-material size class, obtaining the median "reference" value by bed-material size class and by taking the weighted mean based on the percentage of drainage area encompassed by channels of particular bed-material types. The resulting "weighted-reference" values for James Creek are about 2.2 $T/d/km^2$ and 160 mg/l at the $Q_{1.5}$.

"Actual" suspended-sediment yields at the $Q_{1.5}$ as simulated with AnnAGNPS in combination with CONCEPTS show a 35-year average of 675 $T/d/km^2$.

However, the average over the past 10 years is 155 $T/d/km^2$ and, following the installation of additional low-water crossings in 1999 further reduced yields to about 39 $T/d/km^2$, still an order of magnitude greater than the calculated "reference" yield.

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Integrating a Landscape/Hydrologic Analysis for Watershed Assessment

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Abstract

Methods to provide linkages between a hydrologic modeling tool (AGWA) and landscape assessment tool (ATtILA) for determining the vulnerability of semi-arid landscapes to natural and human-induced landscape pattern changes have been developed. The objective of this study is to demonstrate the application of ATtILA and AGWA to investigate the spatial effects of varying levels of anthropogenic disturbance on runoff volume and soil erosion in the San Pedro River Basin. Results were particularly useful for assessing the effects of land cover change in the watershed and highlighting subwatersheds that require careful management.

Keywords: watershed assessment, landscape analysis, hydrologic models, sediment yield

Introduction

Empirical studies have established the significant causal relationship between watershed characteristics and sediment loads (Yates and Sheridan 1983). Agriculture on slopes of greater than 3% increases the risk of soil erosion (Wischmeier and Smith 1978), and this can lead to increases in sediment loadings to surface waters. A decrease in natural vegetation indicates a potential for future water quality problems (Likens et al. 1977, Hunsaker and Levine 1995, Jones et al. 2001).

This study presents an integrated approach to identify areas with potential water quality problems in particular high sediment loadings as a result of land cover change. Landscape metrics describing spatial composition and spatial configuration were computed using the Analytical Tools Interface for Landscape Assessments (ATtILA) (Ebert et al. 2002). These landscape metrics were used along with the Automated Geospatial Assessment Tool (AGWA) (Miller et al. 2002) to examine the contribution of land cover type to sediment yield and identify subwatersheds with high sediment production for the period 1993 to 1997.

Study Area

The San Pedro Basin is located in the northern portion of Sonora, Mexico and southeastern Arizona. The basin is traditionally divided into two sections, the Upper and Lower San Pedro Basins, which are separated by the geologic formation known as "The Narrows." This study includes the Upper San Pedro Basin and a portion of the Lower San Pedro Basin to the Reddington stream gauge. For convenience, all references to the Upper San Pedro Basin in this text refer to the entire study area (Figure 1).

The Upper San Pedro Basin contains approximately 7598 km^2. The Upper San Pedro Basin is bounded by generally north-northwest trending mountains, which range in elevation from 1524 m to nearly 3048 m. The San Pedro River enters the basin at the International Boundary near Palominas, Arizona, and

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flows northwest for about 120 km before leaving the basin at Reddington. The San Pedro River is mostly ephemeral and only flows in response to local rainfall. The river does have a perennial stretch of about 29 km between Hereford and a point just south of Fairbanks (Putman et al. 1988). The Upper San Pedro Basin represents a transition area between the Sonoran and Chihuahuan deserts and topography, climate, and vegetation vary substantially across the watershed. Annual rainfall ranges from 300 to 750 mm. Biome types include riparian forest, coniferous forest, oak woodland, mesquite woodland, grasslands, desertscrub, and agriculture.

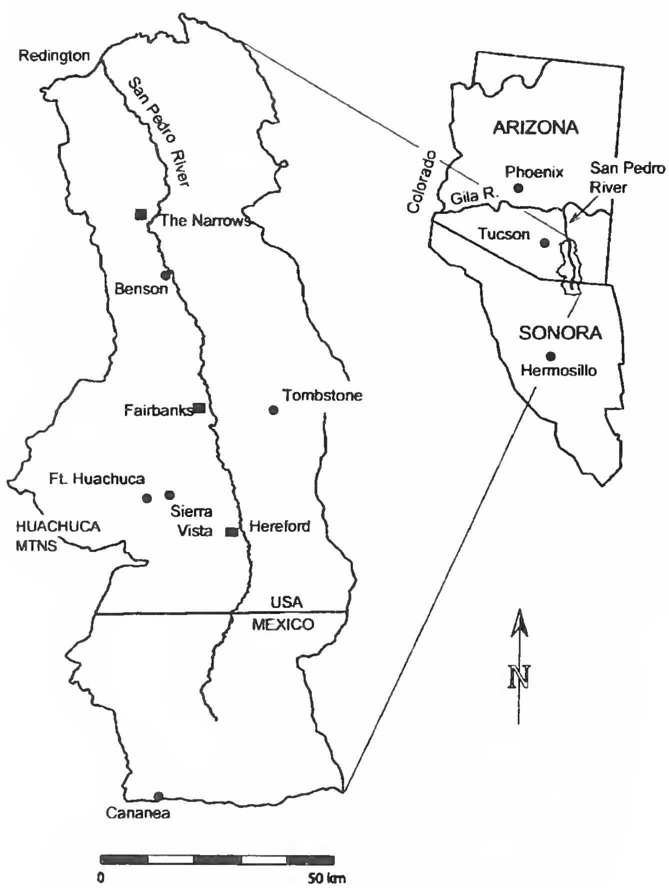

Figure 1. Location of the study area.

Methods

The general approach used in this study was carried out in three steps. The first step consisted of subdividing the Upper San Pedro Basin into subwatersheds or reporting units and computing landscape metrics using ATtILA to quantify the percent cover and spatial pattern on each subwatershed. The second step consisted of applying the AGWA tool to parameterize the Soil Water Assessment Tool (SWAT) (Arnold et al. 1994) and calibrate it using the USGS stream flow gauge at Reddington. The third step consisted of identifying subwatersheds with high potential of water quality problems based on sediment load for the period 1993 to 1997.

Description of ATtILA & AGWA

The U. S. Environmental Protection Agency, Landscape Ecology Branch has developed a user-friendly interface (ArcView extension) ATtILA to compute a wide variety of landscape metrics for categorical map patterns. Four families of metrics are included in the software: landscape characteristics, riparian characteristics, human stressors, and physical characteristics. Each group has a dialog box to accept user input on which metrics to calculate and what input data to use. Landscape characteristics are related to land cover proportions and patch metrics. Riparian characteristics describe land cover adjacent to and near streams. Human stressors are concerned with population, roads, and land use practices. Physical characteristics provide statistical summaries of such attributes as elevation and slope. Once metrics have been calculated ATtILA has three types of output display available. The first displays areas ranked by individual metric value, the second ranks areas by a weighted index made up of two or more metrics, and the third displays a bar chart of selected areas and metrics.

The AGWA tool uses widely available standardized spatial data sets to develop input parameter files for two watershed runoff and erosion models: KINematic EROSion (KINEROS) model and SWAT. Using digital data in combination with the automated functionality of AGWA greatly reduces the time required to use these two watershed models. The user selects an outlet from which AGWA delineates and discretizes the watershed using the Digital Elevation Model (DEM). The watershed elements are then intersected with the soil, land cover, and precipitation (uniform or distributed) data layers to derive the essential model input parameters. The model is then run, and the results are imported back into AGWA for visual display. AGWA is an ArcView extension designed to provide qualitative estimates of runoff and erosion relative to landscape change. Managers can use it to identify problem areas where management activities can be focused, or to anticipate sensitive areas in association with planning efforts.

Landscape metrics computation

Spatial analyses were carried out to (1) describe structural landscape patterns; and (2) relate overall land use changes to hydrological processes. Kepner et al. (2002) used remote sensing techniques for detecting change by analyzing multi-date imagery. Landsat-MSS 1973 was used for the baseline condition. They computed land use change between time intervals 1973, 1986, 1992, and 1997. Digital land cover maps were developed separately for each year using 10 classes: Forest, Oak Woodland, Mesquite Woodland, Grassland, Desertscrub, Riparian, Agriculture, Urban, Water, and Barren. The delineation of the subwatersheds or reporting units was carried out using AGWA dividing the basin into 68 subwatersheds.

Landscape metrics for each patch and cover class within a subwatershed on the 1997 analysis map were calculated using the ATtILA extension. All metrics included in the analysis are listed in Table 1.

Table 1. Landscape metrics included in the analysis

Category	Index Name
Spatial Composition	Land use proportions
	Shannon's diversity index
Spatial Configuration	Number of patches
	Patch density
	Largest patch index
	Average patch size
	Connectivity

Hydrologic simulation

The purpose of the simulation model was to assess the contribution of different land cover types to surface runoff and sediment yield for the period 1993 to 1997. The modeling was based on the subdivision of each of the 68 subwatersheds or reporting units into smaller units by generation of the so-called "Hydrological Response Units" (HRUs) (Leavesly et al. 1983, Maidment 1991).

In general, HRUs are defined by combining spatial attributes relevant to the model into discrete spatial features. The definition of HRUs varies depending on the model's conceptualization. In the case of SWAT, HRUs are response units that have similar hydrological response characteristics and lie within a subwatershed element but need not be contiguous. Runoff contributions from similar areas (HRUs) such as forest, grassland, desertscrub, agriculture, and urban, etc. within a subwatershed element are calculated separately and then summed before routing in the stream and river network.

In this study, the characterization of each HRU within each subwatershed was established based on the landscape metrics computed with ATtILA. In particular we used proportion of land use, slope, number of patches, and average patch size. The total number of HRUs was 384; the HRU mean area was 19.78 km^2 and the maximum and minimum areas were 275 km^2 and 0.0035 km^2, respectively. Sixty five percent of all HRUs have areas less than 12 km^2. The hydrologic parameter most affected by the characteristics of the landscape metrics was Curve Number.

Calibration

The SWAT model was calibrated separately against observed surface runoff and base flow for the period 1993 to 1997. Base flow was separated from the total observed stream flow according to the USGS HYSEP fixed-interval method (Sloto and Crouse 1996). For the calibration we assumed stationary land use conditions based on the 1997 land use and land cover characteristics. The curve number and Manning's roughness coefficient were adjusted to provide better comparisons between mean annual measured and simulated surface runoff. Similarly, for mean annual base flow, the values of initial depth of water in the shallow aquifer and the threshold depth parameter that controls the amount of groundwater flow into the stream were adjusted. Eight rain gauges were used in the calibration process. Daily rainfall data were available from the National Climatic Data Center.

The calibration results show that average annual total water yield at the USGS Reddington stream flow gauge was calibrated to within 12 % of the observed flow. SWAT was calibrated to within 13 % and 4% for surface runoff and base flow, respectively. Based on these results, we argue that SWAT was able to represent the hydro-dynamics of the watershed at the annual scale. No attempt was made to calibrate the model against measured sediment concentration because insufficient data were available at Reddington. For instance, eight, ten, and thirteen mean daily values were available for 1993, 1996, and 1997, respectively. We recognized that these mean estimates might be low because larger events could have occurred on days where data were not recorded. Based on these values, measured mean annual sediment concentration estimates are as follows: 40 mg/L, 73 mg/L, and 48 mg/L for 1993, 1996, and

1997, respectively. Mean annual sediment concentration computed by SWAT are as follows: 115 mg/L, 37 mg/L, and 29 mg/L, for 1993, 1996, and 1997, respectively. SWAT computed sediment yield based on default parameters available in the STATSGO soil database.

The relationship between sediment yield and mean annual surface runoff for Agriculture, Desertscrub, Grassland, and Mesquite Woodland land cover classes is shown in Figure 2. Land use significantly affected the magnitude of sediment through its influence on the degree of protection afforded by the vegetation cover. Kepner et al (2000) presents land cover descriptions for the vegetative communities in the study area. Desertscrub vegetative communities are characterized as having significant areas of barren ground devoid of perennial vegetation. In contrast, Mesquite Woodland are communities described as dominated by leguminous trees whose crowns cover 15% or more of the ground and resulting in dense thickets. Therefore, areas with Mesquite Woodland and Grassland cover types may produce lower sediment yield estimates than desertscrub areas as shown in Figure 2.

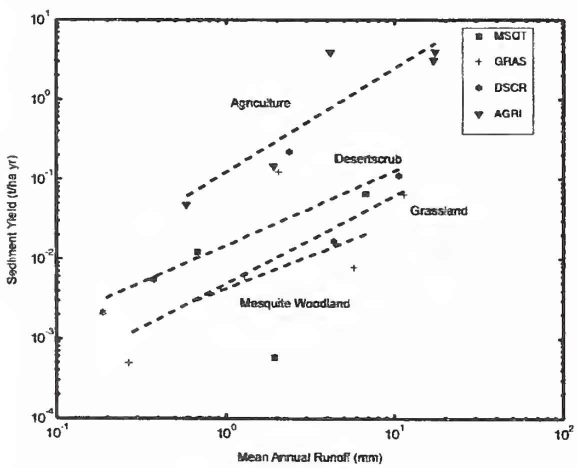

Figure 2. Relationship of sediment yield to mean annual surface runoff for four land use types for the period 1993 to 1997.

Agricultural areas are primarily found along the upper terraces of the riparian corridor and are dominated by hay and alfalfa. They are minimally represented in overall extent (less than 3% total cover) within the basin and are irrigated by ground and pivot-sprinkler systems. However, they may represent a potential source for water quality problem in the region. In addition, we investigated the rate at which sediment yield varies with mean annual surface runoff. We fitted straight lines to the data and computed the correlation coefficients and slopes for each land cover type. The correlation coefficient (r^2) and slope (s) are as follows: Agriculture 0.81 and 1.30; Desertscrub 0.65 and 0.93; Grassland 0.54 and 1.11; and Mesquite Woodland 0.16 and 0.83. From the analysis, Agriculture is the land cover type that produces the highest rate of sediment yield.

Because SWAT is a distributed model, it is possible to view model output as it varies across the San Pedro Basin. Figure 3 depicts the spatial variability of average surface runoff and average sediment yield for the period 1993 to 1997.

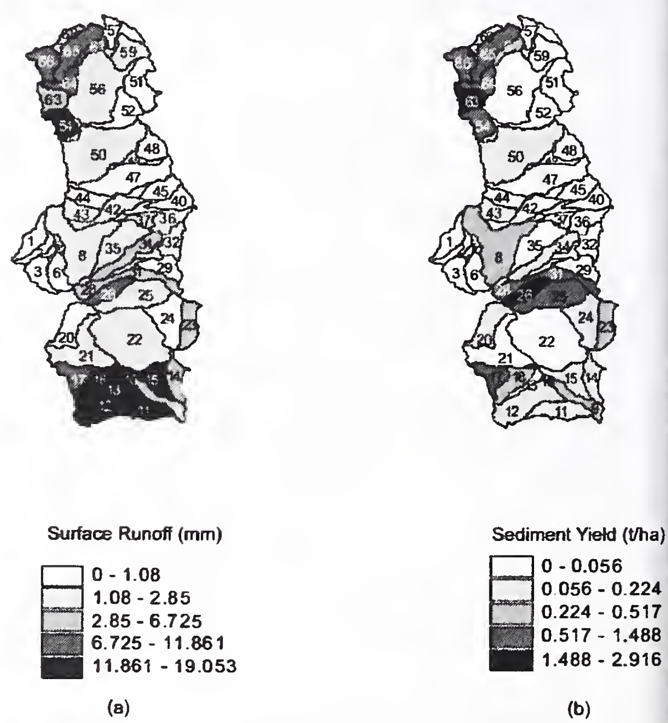

Figure 3. Spatially distributed (a) average surface runoff and (b) average sediment yield for the period 1993 to 1997.

At the watershed scale spatial variability of rainfall, partial area response, gully and alluvial channel densities and properties, and vegetation type largely determine sediment yield (Lane et al. 1997). This influence is apparently primarily through controlling the runoff generation process and channel detachment, transport, and deposition. The spatial variability of sediment yield shown in Figure 3(b) is being controlled primarily by the spatial distribution of surface runoff Figure 3(a).

Assessment

We ranked the HRUs according to high contributing sediment yield areas using the relationship between sediment yield to mean annual surface runoff as a function of land cover type, and the landscape metrics. We used as cutoff criteria the average slope (9%) and the average sediment yield (0.8 t/ha) of all HRUs for the period 1993 to 1997. The selection process yielded eight HRUs; six are classified as agriculture and two as desertscrub. The six agricultural HRUs are located within the subwatersheds 54, 61, 65, 28, 52, and 20. The two HRUs with desertscrub land cover are located within the subwatersheds 63 and 66. Only one subwatershed (20) crosses the boundary with Sonora, Mexico and its main contribution to sediment yield comes from agricultural lands. It is important to point out that the proportion of agricultural land in this subwatershed is 1% compared to 50% of forestland. This indicates that a small area can be the major source of sediment and, consequently, a problem to water quality. The ranking of the eight subwatersheds was carried out based on the average sediment load produced during the period 1993 to 1997. We computed the average sediment load based on the average patch size computed with ATtILA and the average sediment yield computed with AGWA. The outcome of the ranking process is listed in Table 2 and depicted in Figure 4.

Table 2. Sensitive areas with high sediment loads.

Rank	Sub (Id)	Slope (%)	Syld. (t/ha)	Average patch size (ha)	Sed. load (ton)
1	54	15	24.87	13.30	330.84
2	61	19	14.01	8.10	113.48
3	65	19	19.23	4.94	95.10
4	28	18	1.44	33.61	48.41
5	52	13	0.84	47.70	40.07
6	20	13	2.21	8.37	18.51
7	63	24	0.94	5.07	4.77
8	66	21	0.82	3.67	3.01

Conclusions

Landscape pattern analysis was conducted on a subwatershed basis to characterize the heterogeneity of land cover and land use. ATtILA was used to compute metrics associated with landscape characteristics for 1997. Since spatial variability of land cover alters the hydrological structure within the watershed, we used AGWA to examine the watershed response relevant to surface runoff and soil erosion at each subwatershed. We used the concept of HRU to examine the contribution of land cover type to sediment yield for the period 1993 to 1997. The hydrologic model was calibrated against total water yield, surface runoff and base flow using measured stream flow records at Reddington. The highest contribution to sediment yield is produced in areas with agriculture and desertscrub land cover types. We used the average slope steepness, the average annual sediment yield, and the average patch size to identify and rank the subwatersheds that require careful management.

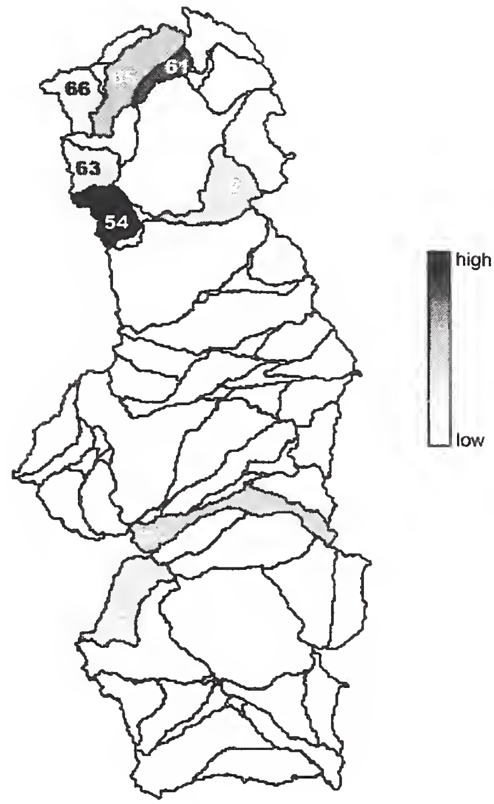

Figure 4. High sediment load subwatersheds based on land cover type, slope steepness, and average patch size for the period 1993 to 1997.

Methods for developing integrated planning and management strategies need to be spatially explicit, refer to specific areas, and utilize basic biophysical information together with assessments of both potential uses of individual land units and the potential levels of primary threats in each. The integrated approach presented here allows resource managers to integrate landscape spatial analysis with hydrological modeling to identify problem areas.

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First Interagency Conference on Research in the Watersheds

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Hydrology III

Exchange of Water, Solutes, and Nutrients at the Sediment-Water Interface Affects a Northern Minnesota Watershed at Multiple Scales

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Abstract

Flow of water, solutes, and nutrients across the sediment-water interface at two lakes, a stream, and a wetland (fen) in a northern Minnesota watershed has variable influence on their hydrology, geochemistry and ecology, depending on the scale that is considered. Ground water provides chemicals to a closed-basin lake that are sequestered by biological processes in the lake, but calcium, removed from the water column as a result of photosynthesis, is not present in profundal lake sediments. Biological and chemical processes deposit and then dissolve calcium in the near-shore margins, and it can leave the lake basin by advection in areas where lake water flows to ground water. At a nearby lake, inputs of dissolved inorganic carbon from a river and from ground water increase fluxes of carbon from the lake to the atmosphere. Ground-water discharge of iron and manganese also sequesters phosphorus in the profundal sediments. Much of the ground-water discharge to that lake is focused at near-shore springs, which are conspicuously absent of aquatic vegetation. Upstream of the lake, strong ground-water discharge reduces the thickness of the hyporheic zone, where microbes consume ammonium and reduce the amount of nitrogen that reaches the stream. Ground-water discharge also provides nutrients and a stable environment for establishment of aquatic vegetation in the streambed. At a nearby fen, ground-water discharge provides a stable environment for rare and protected plants. Locally, most of the discharge occurs at the break in slope along the fen margin, where other aquatic plants thrive. These results indicate that exchanges between ground water and surface water need to be well understood at multiple scales if the watershed as a whole is to be managed effectively.

Introduction

Increasing demands on our natural resources require greater understanding of ecosystem-scale processes that interact to generate landscape characteristics. Water-resource managers typically manage on a watershed or even regional scale, whereas field scientists typically conduct research at much smaller scales. Resolving this scale conflict remains a significant challenge to scientists and managers alike. Watershed-scale research has evolved to include scientists from numerous disciplines. Many hydrologists, biogeochemists, and ecologists have adopted an ecosystem perspective and now work together to solve site-scale and watershed-scale research questions (Likens and Bormann 1995). Ecohydrology is a rapidly growing subdiscipline that often is conducted at a watershed-scale (Wassen and Grootjans 1996, Baird and Wilby 1999, Gurnell et al. 2000, Nuttle 2002).

Quantification of hydrological and chemical fluxes, and biological processes, commonly is scale dependent. Scale-dependence problems are well documented and exist in many disciplines from ground-water hydrology (e.g. Rovey II and Cherkauer 1995) to chemistry (e.g. Capel and Larson 2001) to biology (e.g. Angermeier and Winston 1998). Interpretations resulting from a local-scale study can be greatly different from those resulting from a larger-scale study. Better understanding of the effects of scale on interpretation of research results can be achieved by studying processes at multiple scales, within a single watershed. This paper presents results from the Shingobee River headwaters area in northern

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Minnesota that demonstrate how interpretations can differ depending on the scale of the study. Flow of water and solutes between ground water and surface water is a common emphasis for all of the studies described herein. A better understanding of the effects of this exchange on ecosystem processes at a watershed scale results from taking a comprehensive view of these studies conducted at sub-watershed scales.

Study Area

The Shingobee River headwaters area (Figure 1) is the site of long-term research initiated by the U.S. Geological Survey in 1989. The purpose of the study is to examine the linkages that allow atmospheric water, surface water, and ground water to function as an integrated system (Averett and Winter 1997). Initial study was focused on two lakes within the watershed, one (Williams Lake) with no surface-water exchange and a relatively long water residence time, and one (Shingobee Lake) with a relatively short water residence time because it exchanges water with the Shingobee River. Processes in lakes integrate many physical, chemical, and biological processes that occur in watersheds; therefore, lakes were a natural focal point for concentrating investigations. Research also has focused on a reach of the Shingobee River upstream from Shingobee Lake, and at a wetland fen that discharges into the Shingobee River.

Figure 1. Shingobee River headwaters area.

The watershed boundary for the Shingobee River headwaters area encompasses 28 km^2 of hummocky glaciated terrain situated in a humid continental climate where evaporation and precipitation are about equal. Precipitation averages 640 mm yr^{-1} with about 12 percent falling as snow (Siegel and Winter 1980, Rosenberry et al. 1993). Discharge from Shingobee Lake to the Shingobee River has averaged 0.4 m^3 s^{-1} since 1989 and is relatively stable except following large rain events and during spring snowmelt. The watershed is covered primarily with mixed deciduous and coniferous forest, interspersed with pasture and fallow fields.

Exchange Between Ground Water and Surface Water in a Closed-Basin Lake

Williams Lake (39 ha, 9.8 m maximum depth) receives diffuse ground-water discharge along the south and east shoreline, but loses water to ground water along much of the west and north shoreline (Siegel and Winter 1980). Ground water dominates the water budget for Williams Lake, providing 58-76 percent of the annual water inputs (LaBaugh et al. 1995).

From a lake-basin perspective, ground-water discharge to Williams Lake supplies silica, calcium, and alkalinity that are removed from the lake water column by biological processes (McConnaughey et al. 1994). Those constituents are then deposited in the lake sediment. However, data from sediment transects indicate that very little calcium carbonate is present in sediments near the sediment-water interface beyond about 5-m depth (Dean and Bradbury 1997).

Local-scale, near-shore investigations indicated that calcium carbonate precipitate was deposited in the near-shore littoral sediments of the lake (Schuster et al. 2003). Sediment pore-water samplers were used to collect sediment-water chemistry data beneath the lakebed at 5-cm intervals. Calcium concentrations in pore waters of the near-shore, littoral sediments were substantially greater than lake-water or ground-water concentrations. In addition, calcium concentrations in the pore waters of these surficial sediments varied seasonally. Data indicated that plants sequester calcium in the littoral zone and deposit it in shallow sediment following senescence. Calcium deposited in the littoral sediments is dissolved by the acidic, anoxic pore water. It then mixes with the more dilute lake water or is lost to ground water, depending on the local hydraulic gradient.

Dean and Schwalb (2002) reached a similar conclusion through analyses of lake-sediment cores collected from near the center of the lake. They indicated that dissolution of calcium carbonate, caused by organic acids created by decomposition of organic matter, constitutes a "carbon pump" that has significant implications regarding cycling of other elements. The sediment record also indicates that very little calcium carbonate has been deposited in the lake sediments for the past 4000 years (Schwalb et al. 1995). However, older, deeper sediments contain abundant calcium carbonate, indicating that the lake had a different hydrologic setting prior to 4000 years ago. The carbon and oxygen isotopic composition of that older carbonate material tracks the hydrologic evolution of Williams Lake (Schwalb and Dean 2002). These results also have been extended from a watershed to a global scale (Dean and Gorham 1998). Based on data and understanding gained from studies of Williams and Shingobee Lakes (as well as from nearby Elk Lake and several other sites), lakes, reservoirs and peatlands, which collectively cover less than 2 percent of the Earth's surface, bury organic carbon at an annual rate that is three times the carbon burial rate in all oceans, which cover 71% of the Earth's surface.

Influence of Ground-water Discharge to a Lake Dominated by Streamflow

Shingobee Lake (66 ha, 10.7 m maximum depth) is hydrologically dominated by exchange with the Shingobee River (Rosenberry et al. 1997) and its chemistry is greatly affected by that of the river. Major-ion concentrations are 2-4 times greater, total phosphorus is 2.3 times greater, and ammonium concentration is about 3.6 times greater than at Williams Lake (LaBaugh 1997). Total kjeldahl nitrogen and nitrate plus nitrite concentrations are about the same at both lakes. In spite of the surface-water dominance, Shingobee Lake receives four times as much ground-water than does Williams Lake (Rosenberry et al. 1997). Because much of the water that discharges to Shingobee Lake has a ground-water origin, the water chemistry of Shingobee Lake has a surprisingly strong ground-water signature for a surface-water dominated lake. This leads to some interesting chemical and biological responses.

Dissolved inorganic carbon (DIC) in Shingobee Lake, expressed as alkalinity, is more than twice that in Williams Lake, and is also greater than DIC in the Shingobee River upstream from the lake. About 60-80 percent of the carbon input to Shingobee Lake is from DIC in ground water (Striegl and Michmerhuizen 1998). The large concentration of DIC results in large partial pressure of carbon dioxide, which generates carbon fluxes from the lake to the atmosphere that are about double the fluxes that would occur without ground-water discharge to Shingobee Lake. Therefore, ground-water discharge may be of potential significance to studies of global climate change.

Ground-water discharge also may indirectly sequester phosphorus in the lake sediments, which could reduce lake photosynthesis. Ground water supplies iron and manganese that reach concentrations in the oxygen-depleted hypolimnion of Shingobee Lake that are hundreds of times greater than in the epilimnion (Dean et al. 2003). The large concentrations of iron and manganese precipitate iron and manganese oxyhydroxides at fall turnover. These oxyhydroxides efficiently adsorb phosphorus and sequester it in the profundal sediments of the lake, thus removing it from the lake-water column.

Ground-water discharge to Shingobee Lake has local-scale effects as well. A significant portion of the ground water discharges at springs, many of which are situated near the shoreline of the lake (Rosenberry et al. 2000, Kishel and Gerla 2002). Nearly 30 percent of the estimated 57 L s^{-1} of ground water that discharges to Shingobee Lake originates from the near-shore springs located along the south and west shoreline of the lake. These springs provide stable habitat for aquatic plants where they discharge immediately landward of the shoreline. However, they create conspicuous "dead zones," lakebed areas where no macrophytes are present, where they discharge immediately lakeward of the shoreline (Rosenberry et al. 2000).

Influence of Ground-water Discharge on Nutrient Flux in a Gaining Stream

Flow in the Shingobee River increases by about 20 percent, from 180 to 216 L s^{-1}, along a 1200-m reach upstream from Shingobee Lake (Jackman et al. 1997). This large gain in flow is caused mainly by ground water that discharges to the river at rates varying from 0.006 to 0.06 L s^{-1} per meter of river reach. Large hydraulic-head gradients beneath the streambed limit surface-water exchange and restrict the thickness of the hyporheic zone. During summer, microbes in the relatively thin hyporheic zone reduce the amount of nitrogen that reaches the stream by decreasing ground-water derived ammonium through coupled nitrification-denitrification (Duff and Triska 2000).

Ground-water discharge is relatively high in nitrogen, phosphorus, and DIC compared to concentrations in the Shingobee River, which aids in the establishment of *Elodea canadensis* hummocks during spring following snowmelt. On a local scale, *Elodea* hummocks affect patterns of ground-water discharge (Duff and Triska 2000). Ground-water discharge is greater at the downstream end than at the upstream end of the *Elodea* hummocks because sediment that accumulates at the upstream end is finer and thicker than the surrounding streambed. Rates of nitrification and denitrification are greater beneath the *Elodea* hummocks, indicating that greater microbial activity occurs in those areas.

Influence of Ground-water Discharge to a Calcareous Fen

Little Shingobee Fen is a 13-ha wetland adjacent to and surrounding Little Shingobee Lake (Figure 1). The fen is vegetated primarily by a tamarack and black spruce forest with sphagnum covering the fen surface and a grass sedge zone adjacent to Little Shingobee Lake. Peat and marl deposits up to 14-m thick underlie the fen surface and sand and silt underlie the peat (Carter et al. 1997a). The sloping surface is saturated in most of the fen, and small pools occur throughout. Calcium carbonate concentration in the shallow ground water is about 10 times greater than that of typical fens (Puckett et al. 1997). Three streams discharge 360,000 m^3 yr^{-1} from the fen to Little Shingobee Lake (Winter et al. 2001). Ground-water discharge to the fen is remarkably constant, providing a stable habitat for several rare and protected wetland species, including pitcher plants, sundews, and orchids (Carter et al. 1997b). The steady ground-water discharge indicates that the source of water is distant from the fen and is little affected by seasonal or longer-term climate change.

Much of the ground-water discharge occurs along the eastern margin of the fen where the steeply sloping upland meets the gently sloping fen. Hydraulic-head gradients are strongly upward in this portion of the fen and heads in many of the wells are above land surface (Puckett et al. 1997). *Caltha palustris*, an indicator of areas of rapid ground-water discharge, grows in abundance along this break in slope and almost nowhere else in the fen (Rosenberry et al. 2000). It is likely that much of the central portion of the fen receives water that discharges near the break in slope along the eastern margin. This water then flows overland, down the sloping surface, until it is intercepted by one of the small streams that drain the fen.

Implications for watershed management

Watersheds are appealing to many environmental managers because they provide well-defined boundaries that allow determination of relatively well-constrained water and solute budgets. The budget values of interest generally are for the watershed as a whole, as determined by the major inputs (largely precipitation) and outputs (a stream or lake) that receive the water and solutes derived from the watershed. As presented here, surface-water bodies in relatively flat glacial terrain can have substantial interaction with ground water, and the interaction can vary greatly locally. Ground-water input to a surface-water body can be diffuse, allowing major biochemical interactions to take place in the sediments, or it can be focused at springs, resulting in a different chemical signature and ecological response than the diffuse input. The differences in water and solute residence times in lakes within a watershed can result in the development of substantially different lake ecosystems. Thus, to manage watersheds such as the Shingobee River headwaters area, the small-scale processes such as those discussed herein need to be well understood if the watershed as a whole is to be managed effectively. The challenge is to be able to integrate and scale up understanding of the small-scale processes that are the focus of much research, into broader understanding of the watershed that is useful to environmental managers.

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Hydrogeology of the Alluvial Aquifer Along the Curtiss Reach of the San Pedro River

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Abstract

Remedial investigations supporting the Apache Powder Superfund project since 1990 have revealed examples of hydrogeologic features characteristic of heterogeneous alluvial aquifers. West of the San Pedro River, groundwater flow is strongly influenced by (1) narrow and varying width of the alluvial aquifer between the alluvial materials (Qal) and St. David Formation (Qsd) contact and the San Pedro River, and (2) lateral variations in lithofacies. These lithofacies changes relate to paleodrainages and isolate portions of the shallow aquifer, thereby affecting groundwater flow and solute transport. Additionally, the aquifer width and the meandering and incisement of the San Pedro River in some locations influences groundwater-surface water interactions.

Keywords: San Pedro River, groundwater-surface water interactions, nitrate, Superfund

Introduction

The Curtiss reach of the San Pedro River (the river) is located along the Apache Powder Superfund Site and west of the Town of St. David in Cochise County, Arizona (Figure 1).

The alluvial basin within the St. David area comprises a typical intermontane basin formed within a Basin and Range structural valley. Adjacent to the Apache Powder Superfund Site, the river courses along the western edge of the basin.

Basinfill alluvium comprises a thick, greater than 1,000-ft, sequence of fine- to coarse-grained late Cenozoic sediments derived from the surrounding mountain ranges. The geologic conditions were described in detail by Gray (1965).

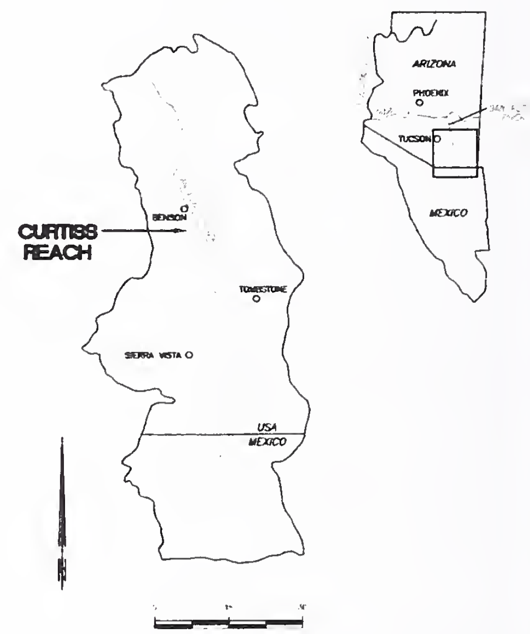

Figure 1. Location of the Curtiss Reach of the San Pedro River.

Two aquifers are present: (1) a shallow aquifer in the alluvial materials (Qal) overlying the St. David Formation (Qsd), and (2) a regional aquifer within sandy to gravelly units of the Qsd. A thick Qsd clay member serves both as the base of the alluvial aquifer and the overlying confining unit of the regional aquifer. The regional aquifer is under artesian pressure, as is the shallow aquifer in most places.

Groundwater and surface water monitoring, lithologic logging, geophysical surveys, and

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geochemical characterization primarily of the shallow aquifer have been performed since 1990 as part of a Superfund Site investigation. These investigations have focused on the source and fate of nitrate-nitrogen (nitrate-N) in groundwater and resulted in a conceptual hydrogeologic model for groundwater flow and solute transport in the shallow aquifer and for groundwater-surface water interaction with the river.

Summary of Investigations

Apache Nitrogen Products, Inc. (ANP) has been the lead Superfund investigator acting with the oversight of the U.S. Environmental Protection Agency (EPA). Formal Superfund studies began in 1990 and have involved the construction of 34 monitor wells, 12 piezometers, and 12 exploratory borings. These studies have been supported by various geophysical surveys employing seismic reflection and high-resolution resistivity. Groundwater levels and quality are monitored quarterly within this network of monitor wells and piezometers plus strategically located private wells. In addition, the river is sampled and gaged at several locations along the Curtiss Reach.

A remedial action featuring a pump-and-treat system was completed in 1997. This system is known as the Northern Area Remediation System (NARS), and comprises a high capacity extraction well, delivery piping, a treatment wetland, and an effluent return system. The NARS is designed to remove nitrate-N and ammonia-N from the groundwater via biological denitrification and aerobic nitrification, respectively.

Detailed studies of the river under baseflow conditions were recently completed (Hargis + Associates, Inc. (H+A) 2003). These studies involved cross channel surveys of the surface water including extensive sampling and analysis for nitrate-N and perchlorate; wellpoint sampling of shallow groundwater adjacent to the river; and flow gaging.

Conceptual Model

The cumulative information derived from the various stages of the remedial investigations have resulted in a conceptual hydrogeologic flow model, which in turn serves to explain important aspects of the dynamics of nitrate-N within the shallow aquifer. The geomorphic and sedimentary history of the shallow aquifer involves a degradation stage followed by a construction stage (Gray 1965, Deane 2000, Hargis + Associates, Inc. 2003a). The shallow aquifer is present in the alluvial materials deposited by the San Pedro River since the Pleistocene Epoch of geologic time. The shallow aquifer comprises a heterogeneous alluvial system emplaced upon the underlying Qsd clay unit, which serves as its base and acts as an effective hydraulic barrier between the shallow aquifer and the underlying regional aquifer.

Fundamentally, the most productive portions of the aquifer are situated near the present river channel and within areas where the ancestral San Pedro River and its tributaries once flowed. Lateral facies changes likely represent overbank deposits and un-eroded paleotopography. In most areas, the shallow aquifer is overlain by finer-grained silts and clays. These lower permeability sediments create semi-confined conditions in the alluvial aquifer. In proximity to the river channel, this confining unit may be absent, particularly where the channel has meandered over a significant distance.

Generally, the base of the shallow aquifer is found between 80 to 120 feet (ft) below land surface (bls). Depths to groundwater range from the base of the river channel to over 60 ft bls with increasing distance from the river. The general direction of groundwater flow is to the northwest, essentially subparallel to the direction of flow of the river.

The western boundary of the shallow aquifer is found along the Qal-Qsd contact and is highly irregular along the Curtiss reach. Several indentations are found where present day ephemeral channels have cut back into the Qsd or into the Granite Wash sediments (Qgw), which are largely reworked Qsd and alluvium from the surrounding mountain ranges (Whetstones and Dragoons). At least some of these areas were probably eroded by ancestral tributary streams that flowed along these same approximate alignments (Figure 2).

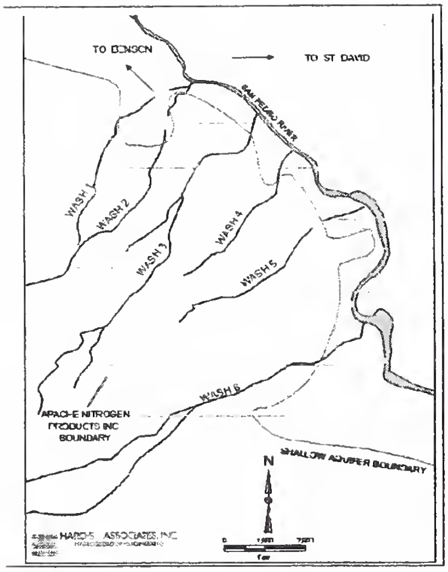

Figure 2. Shallow aquifer boundary and ephemeral washes along the Curtiss Reach.

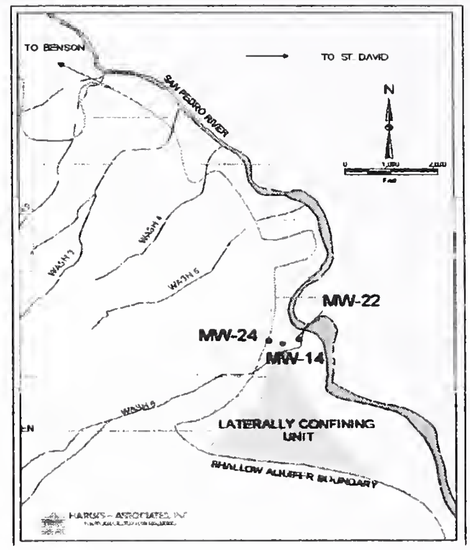

Figure 3. Features of conceptualized hydrogeology along Curtiss Reach.

It has been proposed that the source of the nitrate-N is related to historical surface runoff from the ANP plant site that recharged into the sediments above the shallow aquifer and also to water that recharged on the plant and migrated laterally eastward in the subsurface over the Qsd clay, eventually commingling with the shallow aquifer.

Another important feature of the conceptual model is the laterally confining unit (LCU). This feature was suspected as a result of significant differences observed along an east-west grouping of monitor wells (MW-22, -14, and –24) constructed as a transect normal to the direction of groundwater flow in that portion of the shallow aquifer (Figure 3). Along that transect, significant differences in hydraulic head, hydrochemical facies, water quality, and piezometric surface trends had been noted. This prompted speculation with regard to some lateral hydraulic barrier located between monitor wells MW-14 and –24. Subsequently, exploratory drilling and seismic reflection surveys were carried out to determine whether there was evidence for a lithostratigraphic change and/or elevation differences in the surface configuration of the underlying Qsd clay. The results of these field investigations confirmed both (Deane 2000, Hargis + Associates, Inc. 2003a). The lithology encountered during drilling of exploratory boring, EXB-3, drilled directly west of monitor well MW-14 confirmed the existence of a fine-grained unit, marking a sharp contrast in hydraulic conductivity as compared with typical shallow aquifer materials.

Another seismic survey, complemented by exploratory borings, was completed further to the southwest and west of the shallow aquifer boundary. This survey revealed a lower elevation alignment on the surface of the Qsd clay that projected to the location of monitor well MW-24 and beyond. This feature was interpreted by Deane (2000) as the paleochannel of an ancestral San Pedro River tributary, which was termed Molinos Creek. Thus, Deane (2000) was able to demonstrate that at that location, the shallow aquifer appeared to by hydraulically compartmentalized into two areas termed the Molinos Creek sub-Aquifer (MCA), which occupied the ancestral paleochannel, and the San Pedro Aquifer (SPA), situated along the present San Pedro River.

The lower hydraulic conductivity materials separating the MCA from the SPA were interpreted as overbank materials deposited between the two perennial ancestral streams. The presence and geometry of the LCU was further delineated by additional, recently drilled exploratory borings that showed essentially a triangularly shaped unit (Figure 3). The importance of the LCU is that it has apparently served as a barrier to eastward migration of historically contaminated perched groundwater discharging from the southern, most heavily industrialized, portion of the ANP plant site from migrating eastward into the SPA and the river. This fact became even more evident with the 1998 discovery of perchlorate in perched zone and MCA groundwater. Fortunately, and apparently owing to the hydraulic isolation provided by the LCU, perchlorate has never been detected in the SPA or the river.

Groundwater-Surface Water Interaction

The interaction between groundwater systems and surface streams manifesting as alternating gaining and losing reaches has been studied extensively in southern Arizona (Usunoff 1984, Ellett 1994, Roudebush 1996). Along the Curtiss Reach, at least two gaining reaches have been identified as a result of extensive in-stream surveys involving quality sampling and analysis, gaging, and well points during extreme baseflow conditions. Specifically, an extensive survey involving water quality sampling and analysis over an approximate 3-mile distance incorporating the Curtiss Reach and involving installation and sampling of 27 wellpoints and collection of 75 surface water samples was completed in October 2001 (Figure 4). Additional wellpoint and surface water sampling was performed in 2002.

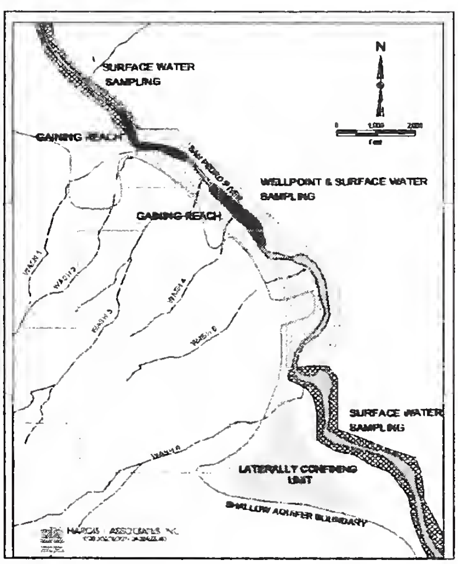

Figure 4. San Pedro River sampling network during October 2001 baseflow conditions.

On the basis of nitrate-N concentrations detected in the water samples, two gaining reaches were inferred (Figure 4). Further examination of these two reaches revealed common features in regard to topography and fluvial geomorphology. Both gaining reaches were located at or downstream from locations where the river meandered sharply westward from its generally northwestward course. Also, the river channel was deeply incised in both areas. This morphology appears to favor interception of the shallow aquifer groundwater flowing subparallel to the river in a northwestward direction. Additionally, the incisement and meandering of the river serve to remove the confining strata above the shallow aquifer allowing upward movement of groundwater in the vicinity of the channel.

Another interesting aspect of this regime was the alignment of these gaining reaches with the aforementioned indentations in the shallow aquifer boundary corresponding to the locations of Washes 4 and 5 (Figures 2 and 4). Again, inferences can be drawn with regard to the potential for such areas to have contributed historically to the presently observed nitrate-N discharges along these two gaining reaches (H+A 2003b).

Influence of Vertical Heterogeneities

Another interesting phenomenon has been observed in conjunction with the pumping of extraction well SEW-01, which withdraws groundwater with high concentrations of nitrate for treatment at the NARS. Extraction well SEW-01 is located about 50 ft from the MW-17 /18 monitor well pair. MW-17/18 are within close radial proximity, however, monitor well MW-17 is screened across only the upper 20 ft of the shallow aquifer, approximately 60 to 80 ft bls; whereas MW-18 is screened across approximately the lower 15 feet of the shallow aquifer, approximately 83 to 98 ft bls. By comparison, extraction well SEW-01 is screened across the entire shallow aquifer saturated thickness from approximately 62 to 110 ft bls.

The hydrographs and nitrate-N concentrations observed in MW-17/18 were nearly identical prior to the 1997 installation of extraction well SEW-01. Thereafter, greater hydrograph separation during periods of extraction well SEW-01 pumping is observed generally with MW-18 exhibiting greater drawdown than MW-17. During periods of non-pumping, however, both return to approximately the same static water level (Figure 5). Similarly the time-concentration plots for nitrate-N in MW-17/18 show separations with the concentration in MW-18 generally lower than in MW-17 (Figure 6).

These observations are believed to result from vertical heterogeneities in the shallow aquifer in the vicinity of monitor wells MW-17/18. Specifically,

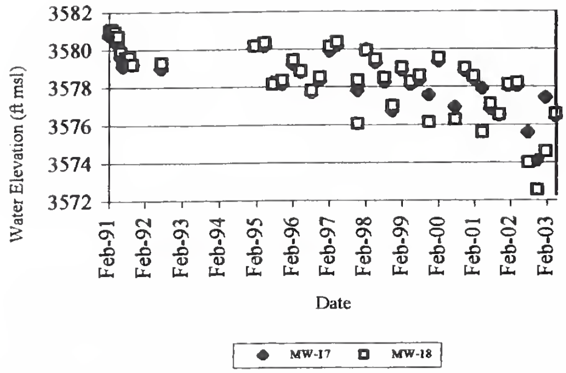

Figure 5. Hydrograph for monitor wells MW-17/18.

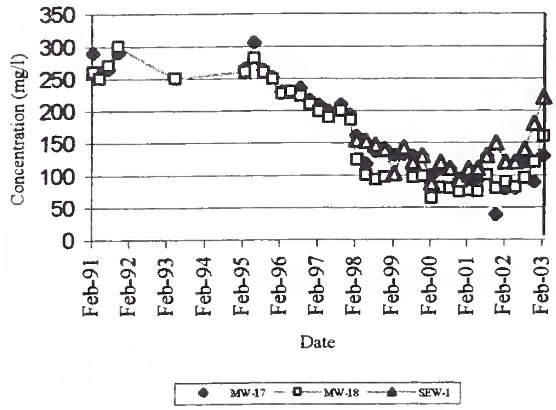

Figure 6. Time-concentration data for monitor wells MW-17/18 and extraction well SEW-01.

the lithologic log for an exploratory boring drilled and logged prior to the construction of monitor wells MW-17/18, shows a 7-ft clay unit 83 to 90 ft bls. Thus, it appears that the pumping of water from nearby SEW-01 differentially draws water from above and below the clay unit and hence from the intervals open to monitor wells MW-17 and -18, respectively. During periods of pumping stress, the water level in MW-18 is affected more than the water level in MW-17. The greater effect on the water level in MW-18 occurs either because downward vertical recharge through the clay is slow or because the pump in extraction well SEW-01 is set near the bottom of well, across from the monitor well MW-18 screened interval, or a combination of these factors (Figure 7).

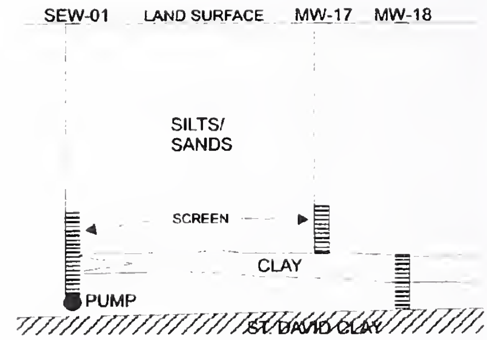

Figure 7. Schematic of monitor wells MW-17/18 and extraction well SEW-01.

Conclusions

The hydrogeology along the Curtiss reach displays a heterogeneous alluvial system with features that control the occurrence, availability, and quality dynamics of groundwater. The opportunity to discover these features arises from remedial investigations associated with Apache Powder Superfund Site. The results of these investigations provide a detailed understanding of the water quality dynamics and the relationships among the shallow aquifer, regional aquifer, and San Pedro River.

Acknowledgments

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Analysis of Long-Term Precipitation for the Central Texas Blackland Prairie: 1939 to 1999

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Abstract

Historical data on precipitation occurrence, amount, intensity, and spatial and temporal variability are vital in water resource management. These data are beneficial in adapting agricultural, industrial, ecological, and domestic water supply management to best utilize the occurrence of natural rainfall events because rainfall ultimately determines surface and groundwater supplies. Therefore, knowledge of historical rainfall patterns is necessary to make informed decisions and predictions about future water supplies. In the Texas Blackland Prairie, an important agricultural region with a large and increasing urban population, drought and excess rainfall can be experienced throughout the year. With the diverse demands placed on water resources in this region and an increasing demand due to population growth, water resource management will be an even more important issue in the future. Faced with these demands, the continuous precipitation record from the USDA-ARS Grassland, Soil and Water Research Laboratory watersheds near Riesel, TX should prove valuable in water resource planning and management by providing information on long-term trends in rainfall amount intensity, and frequency.

Keywords: precipitation data, climate change

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Introduction

Our objective in this paper is to present the results of long-term analyses of precipitation data for the central Texas Blackland Prairie (1939-1999). These results should provide useful information to projects involving design of hydrologic structures such as dams, culverts, and detention basins, water supply and water quality modeling, and other hydrologic and water quality issues relevant to the region. The lack of adequate hydrologic data has been recognized for some time as a cause for failure of hydrologic structures but more commonly as contributing to unnecessarily conservative safety factors in structure design (USDA-SCS 1942). More recently, the importance of precipitation data in hydrologic modeling has been demonstrated (Favis-Mortlock 1995, Chaubey et al. 1999, Harmel et al. 2000).

In addition to presenting results from the precipitation analyses, it is hoped that this paper will publicize the availability of the precipitation data from Riesel. Recent publications by Hanson (2001) and Nichols et al. (2002) provide valuable regional precipitation analyses based on USDA-ARS experimental watershed data from Reynolds Creek, ID and Walnut Gulch, AZ. This publication will provide similar regional precipitation analyses within the Texas Blackland Prairie.

Site Description

The Blackland Prairie encompasses 4.45 million ha and is a major agricultural region extending from San Antonio 480 km north to the Red River (Figure 1). The area also contains the major metropolitan areas of Dallas, Fort Worth, Waco, Temple/Belton/Killeen, Austin, and San Antonio. Houston soils are the most extensive in the region and are noted for their strong shrink/swell potential.

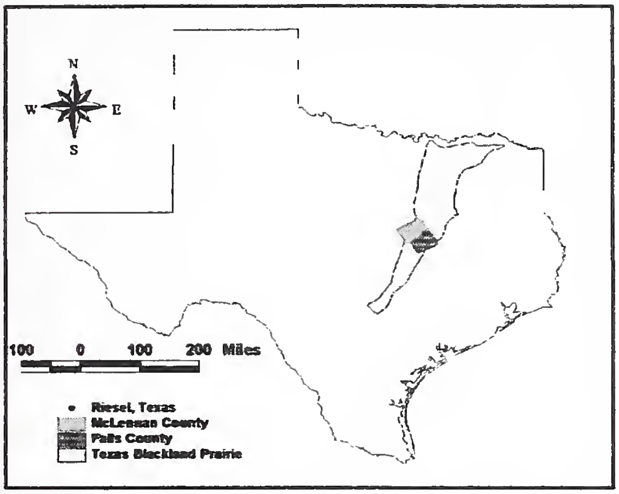

Figure 1. The Texas Blackland Prairie.

Long, hot summers and short, mild winters characterize the climate in the Blackland Prairie. The growing season lasts on average from mid March to mid November. Most rainfall occurs associated with the passage of Canadian continental and Pacific maritime fronts (Knisel and Baird 1971). Convective thunderstorms during the warmer months also contribute intense, short duration rainfall events. Tropical hurricanes can contribute substantial rainfall in rare occurrences. Frozen precipitation can occur occasionally but does not contribute significant moisture.

Riesel Precipitation Network

In the mid 1930s, the Soil Conservation Service (SCS) realized a need to analyze and understand hydrologic processes on agricultural fields and watersheds because of their impact on soil erosion, flood events, water resources, and the agricultural economy. As part of the SCS research program, the Hydrologic Division was created and a number of experimental watersheds were established across the United States. The primary functions of the facilities were to collect hydrologic data and to evaluate the hydrologic response from watersheds influenced by various agricultural land management practices (USDA-SCS 1942). One of those three original facilities, the Blackland Experimental Watershed, was established in 1937 in the heart of the Blackland Prairie near Riesel, TX (Figure 1). This experimental watershed facility later became part of the USDA-ARS Grassland, Soil and Water Research Laboratory with headquarters in Temple, TX.

Collection of rainfall data at the Riesel facility began in 1937 and continues through today. A total of 57 rain gauges were used at some time during the period of record. Historical data from these rain gauges (approximately 1400 rain gauge years) are available on the website: http://arsserv0.tamu.edu/hydata.htm. Rainfall estimates for individual watersheds can be calculated from Thiessen weights, which are also available. This site lists the stations and the years for which daily and sub-daily rainfall data are available. Currently, 15 rain gauges are in operation within the 340 ha watershed area (Figure 2). These operational gauges are instrumented with a tipping bucket rain gauge and datalogger to measure and record sub-daily precipitation. A standard non-recording rain gauge is also used at each site as a backup and calibration device.

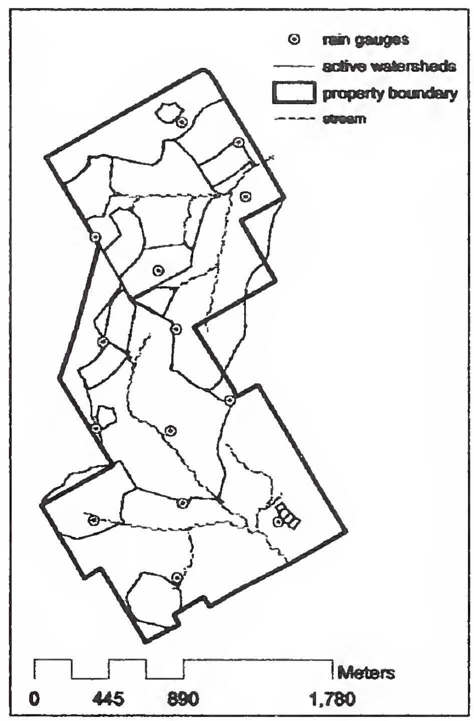

Figure 2: Active rain gauges at Riesel.

Methods

General rainfall characteristics

Descriptive statistics for measured monthly, seasonal, and annual rainfall were calculated for the period from 1939 to 1999 for the longest continually active gauges. Data from these four gauges were also used to compare the spatial variability of daily rainfall. Analyses of rainfall occurrence, amount, and variability were also conducted for a representative rain gauge.

Depth-duration-frequency relationships

To determine relationships between rainfall amounts, duration of rainfall events, and the expected number of events over a given time period, annual maximum rainfall amounts for 0.25, 0.5, 1, 2, 3, 6, 24, 48, and 72 hr durations were calculated. The return frequency for each of these depths and durations was then calculated as indicated by Haan (1977). The mean depth for each of these durations was in turn computed for 1 through 50 yr return frequencies for comparison with results calculated with the USGS depth-duration-frequency procedure for Texas (Asquith 1998) and estimates from the Rainfall Frequency Atlas of the United States or "TP-40" (Hershfield 1961). Return frequencies of seasonal and annual rainfall were also calculated.

Trend analysis

Monthly and annual means and standard deviations for all available rain gauges with continuous records were calculated to detect trends in rainfall amount and variability from decade to decade. Linear regression over time was conducted to evaluate long-term trends in monthly, seasonal, and annual rainfall totals. Trends in rainfall variability were examined with regression analyses on the absolute value of the residuals.

Results and Discussion

General rainfall characteristics

Mean annual rainfall for the four continuously active rain gauges was 880 to 900 mm. Typically, spring (defined as March, April, and May) is the wettest period with average rain of 270 mm. Fall (September, October, November) is also relatively wet averaging more than 220 mm. Winter (December, January, and February) and summer (June, July, and August) are relatively dry with approximately 190 mm of rainfall occurring each season. Average monthly rainfall ranges from 115 mm in May to less than 50 mm in July. In the wettest period, April through June, rainfall averages approximately 292 mm. In contrast the driest period, July through September, receives only 165 mm. On average, 90 days per year had measurable rain (greater than 0.25 mm), 72 days per year had rain amounts greater than 0.76 mm, and 11 days had rain amounts greater than 25 mm. It is interesting to note that the wetter months generally exhibit greater rainfall variability (Figure 3).

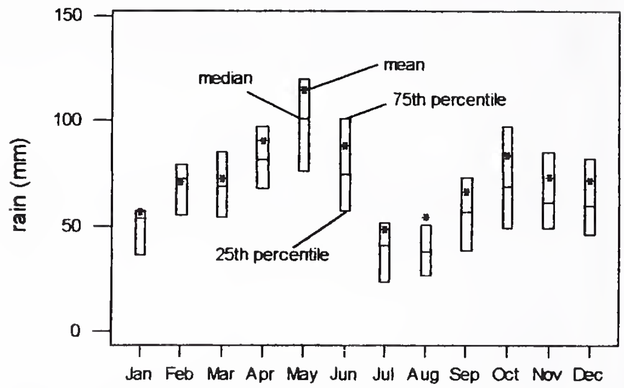

Figure 3: Monthly distribution of rainfall and rainfall variability.

Monthly, seasonal, and annual rainfall amounts and variability did not differ among the four rain gauges. This result was expected as the gauges are within 2500 m of each other with little change in topography. In terms of the maximum difference between the four rain gauges for each rainy day, relatively small differences were observed. On days with rain, spatial variability ranged from 0 to 54 mm with an average difference of 2.8 mm. However, 75% of values were within 3.3 mm and 90% were within 7.1 mm. Because of the small difference between these four rain gauges, rainfall characteristics and trends analysis are presented for a representative rain gauge.

Depth-duration-frequency relationships

Measured data and estimates derived from the USGS depth-duration-frequency procedure were similar at return frequencies greater than 1 yr for all durations. Although TP-40 results were similar to measured data for most durations for return frequencies greater than one year, TP-40 depths for the commonly used 24 hr design duration were considerably larger (approximately 25 mm). This difference represents a significant volume in hydrologic design and emphasizes the need for engineers to use the most up-to-date and extensive data sets and/or proven relationships when designing hydrologic structures.

These depth-duration-frequency results are valuable for hydrologic structure design, but seasonal and annual return frequencies are more important for water supply issues. Wet spring and fall seasons are much more frequent than wet summers and winters (Figure 4). Late fall through early spring rainfall is important because of increased likelihood of runoff into water supply reservoirs due to low evaporation and transpiration losses in this period.

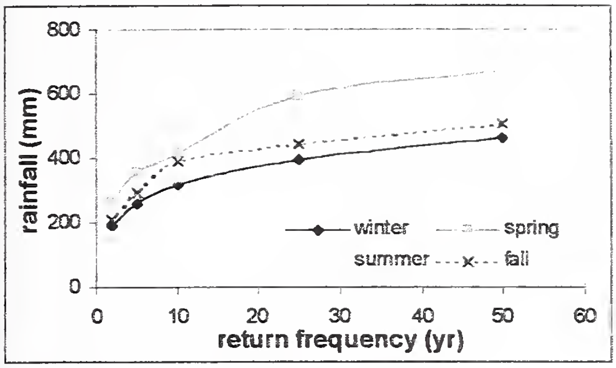

Figure 4: Seasonal return frequencies

Trend analysis

With two exceptions, the decadal monthly and annual rainfall totals from the 1940s to the 1990s seemed to fluctuate without any readily apparent trend. These two exceptions were detected through linear regression analyses and are presented in Table 1. The table also shows the general tendency of rainfall from the late 1930s through 1999. Regression analysis resulted in one statistically significant change in monthly, seasonal, and annual rainfall at $\alpha = 0.05$ and 0.10. The slope of the regression line (the change in monthly rainfall) for October rainfall was significant. October rainfall increased 0.89 mm (0.035 in) per year, which in terms of water supply is an important increase of over 50 mm in the 64 yr period of record. It is also interesting to note the relatively large increase in rainfall amounts for all seasons except spring and the overall annual increase of over 1.6 mm. When the influence of decreasing spring rains was removed, a significant increase in non-spring (summer, fall, and winter) rainfall of 2.19 mm per year was detected by regression analysis ($\alpha = 0.10$).

Because two significant trends in rainfall amount were determined, we wanted to determine whether the changes were due to changes in rainfall frequency and/or magnitude. Linear regression analysis resulted in statistically significant ($\alpha = 0.10$) increases in the number of summer and fall rainy days and also a significant increase in the total number of rainy days each year.

Table 1: Linear tendencies in rainfall amount from 1938 to 1999

time period	ave change (mm/yr)	time period	ave change (mm/yr)
jan	-0.03		
feb	0.15	winter	0.55
mar	0.43	spring	-0.57
apr	-0.67	summer	0.56
may	0.09	fall	1.07
jun	0.01		
jul	0.32	non-spring	2.19*
aug	0.24		
sep	0.01	annual	1.63
oct	0.89*		
nov	-0.20		
dec	0.39		

*statistically significant at a = 0.10.

A similar analysis was conducted on the frequency and magnitude of extreme rainfall events. From the 1940s to the 1990s, the number of days with rain exceeding 25.4 mm decreased for the spring (Figure 5). This observed decrease over time was statistically significant as determined with linear regression ($\alpha = 0.10$). This decrease in the number of spring days per year with rain greater than 25.4 mm did not, however, significantly affect the total annual number of days with excessive rain. No significant change in the number of days with rain greater than 50.8 mm of rain was evident from the 1940s to the 1990s. In terms of the magnitude of extreme rain events, linear regression determined significant decreases for the 75th percentile winter, spring, and fall rainfall events and the 95th percentile fall event.

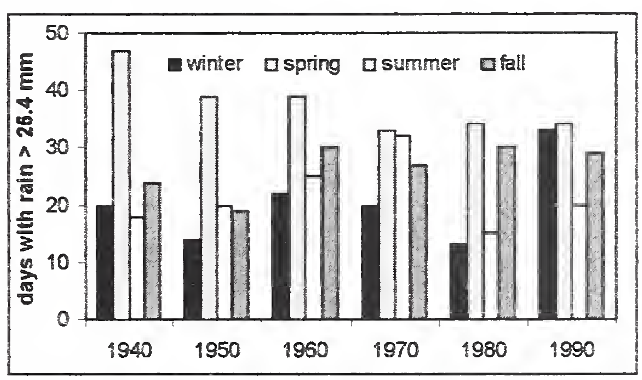

Figure 5: Number of days with rain exceeding 25.4 mm in each season.

The interaction of these changes in rainfall amount impact the overall trends presented in Table 1. The increases in October rainfall amount and the number of summer and fall rainy days contribute to increasing precipitation trends in summer and fall. This impact, however, is lessened by the decrease in extreme rainfall events in the fall. The decreases in both rainy days and days with extreme rainfall in the spring contribute to decreasing spring rainfall. When the influence of decreasing spring rainfall was removed, as was done for summer by Nichols et al. (2002), a significant increase in non-spring rainfall was determined. The lack of a significant trend in annual rainfall even though non-spring rains are increasing can be attributed to the decrease in spring rains, which contributed about 30 % of the annual rain total.

Trends of rainfall variability, as well as rainfall amounts, are important in water resource management, cropland and rangeland production, and other applications. Based on examination of plots of monthly and annual rainfall and residuals and decadal patterns, rainfall variability for certain months appears to be changing. The most noticeable observation was the pattern of decreases in the variability of November rainfall and increases in December rainfall variability from the 1940s to the 1990s. To test these initial observations, linear regression analysis was conducted on the absolute value of residuals of monthly rainfall. These tests indicated significant increases in February and August variability and a significant decrease in November variability ($\alpha = 0.10$).

Conclusions

This paper presents results of selected long-term analyses of precipitation data collected at the USDA-ARS Grassland, Soil and Water Research Laboratory, Riesel, TX since the 1930s. For this period, annual rainfall averaged 892 mm. In the wettest months (April, May, June, October) average monthly rainfall exceeds 85 mm, but for July, August, and January, rainfall averages less than 55 mm. On average 72 days per year have rain greater than 0.76 mm, and 11 days per year have rain amounts greater than 25 mm.

The depth-duration-frequency analysis yielded several notable results. The measured 24 hr storm depth, which is often used in hydrologic structure design, ranged from 89 mm for a 2 yr return period to 192 mm for a 50 yr return period. When the measured depth-duration-frequency relationship was compared to a USGS depth-duration-frequency procedure (Asquith 1998), results were similar for all return periods greater than 1 year for all durations. Measured depths were also generally similar to TP-40 results (Hershfield 1961); however, the 24 hr TP-40 depths were approximately 25 mm larger than measured depths. This difference can represent a significant volume in hydrologic design and emphasizes the need for engineers to use the most up-to-date and extensive data sets and/or proven relationships in design of hydrologic structures. Based on these results, the USGS depth-duration-frequency procedure (Asquith 1998) is a recommended alternative to measured data for hydrologic design in the central Texas Blackland Prairie region.

Although the general rainfall characteristics and depth-duration-frequency relationships are important, the analysis of possible changes in precipitation patterns due to possible global climate change has become a topic of intense speculation. In this study, we observed significant increases in October rainfall, non-spring rainfall, and the number of summer and fall rainy days; all which contribute increased rainfall. This impact, however, is lessened by decreases in the number of rainy days and extreme events in the spring and in the magnitude of extreme fall rain. These changes in rainfall are different from those observed in other regions. For instance, Nichols et al. (2002) reported increasing annual precipitation due to an increasing frequency in non-summer rains. These differing findings reinforce the need and value of regional, long-term precipitation analysis.

The long-term analyses included examination of general precipitation properties, depth-duration-frequency relationships, and trends in rainfall amount and occurrence. Results from these selected analyses, as well as additional analyses possible for the extensive database, should prove useful in hydrologic structure design, water supply and water quality management and modeling, and other hydrologic and water quality issues relevant to the Texas Blackland Prairie region.

Acknowledgments

We would like to recognize the efforts of many past employees that have contributed to the collection of data at Riesel. We especially want to recognize current staff members Lynn Grote, Steve Grote, James Haug, and Gary Hoeft for their outstanding efforts in equipment maintenance, data collection, and record keeping. Without their service the intensive hydrologic monitoring program at Riesel would not be possible.

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Multi-Watershed Evaluation of WSR-88D (NEXRAD) Radar-Precipitation Products

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Abstract

The National Weather Service (NWS) operates a network of Doppler-radar stations (NEXRAD, WSR-88D) that produce hourly-rainfall estimates, at approximately 4-km^2 resolution, with nominal coverage of 96% of the conterminous US. Utilization of these data by the NWS is primarily for the detection and modeling of extreme-weather events. Radar-precipitation estimates were compared with gauge estimates at six ARS watershed-research locations in Idaho, Arizona, Oklahoma, Georgia and Mississippi to evaluate the utility of these data for hydrologic and natural resources modeling applications. Radar precipitation estimates underestimated gauge readings for all locations except Tucson. In all cases, the total number of hours with measured-radar precipitation was much less than hours containing gauge-precipitation estimates. Additional modification of NWS precipitation processing procedures will be necessary to improve accessibility and utility of these data for hydrologic and natural-resource modeling applications.

Keywords: NEXRAD, radar, gauge, precipitation, watershed, calibration

Introduction

The National Weather Service (NWS) operates approximately 160 WSR-88D Doppler-radar stations as part of a Next Generation Radar (NEXRAD) program that began implementation in 1992. These radar stations provide spatial rainfall estimates, at approximately 4-km^2 resolution, with nominal coverage of 96% of the conterminous United States (Crum et al. 1998). This network was originally designed to support Departments of Defense, Transportation and Commerce objectives for detection and mitigation of severe weather events (Crum and Alberty 1993, Baeck and Smith 1998, Crum et al. 1998, Whiton et al. 1998, Winchell et al. 1998, Witt et al. 1998a, Witt et al. 1998b, Anagnostou and Krajewski 1999, Fulton 1999, Warner et al. 2000). The primary hydrologic applications have been modeling of river flow and flood forecasting by thirteen NWS River Forecast Centers (RFC). Data processing and quality control of NEXRAD precipitation products is optimized for a relatively large spatial domain (>100,000 km^2) (Anagnostou and Krajewski 1998, Seo et al. 1999).

Utilization of NEXRAD data in agricultural and natural-resource management applications has been slow to develop (Brandes et al. 1991, Nelson et al. 1996). Studies that evaluate NEXRAD datasets for

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input into non-NWS hydrologic models focus on parameter sensitivity and variability rather than spatial accuracy of the data (Winchell et al. 1998, Koren et al. 1999, Carpenter et al. 2001). NWS-RFC applications for NEXRAD data occur in real-time within a custom software system that is inaccessible to most external users (Anagnostou and Krajewski 1998). Digital, distributed-precipitation radar products can be downloaded directly from the NWS but hourly precipitation files are stored in binary-coded format. Georeferencing tools for comparing NEXRAD and ground-based measurements are relatively difficult to obtain and must be adapted for use outside of the NWS-RFC application domain. The ARS Northwest Watershed Research Center has developed software tools to decode, process and evaluate NEXRAD precipitation products. The purpose of this study was to use data from the ARS-watershed gauge network to evaluate the utility of NEXRAD Stage-1, Level-3 spatial-precipitation data for agricultural and natural-resource management applications.

Methods

NEXRAD precipitation-processing protocols consists of three processing Stages (Anagnostou and Krajewski 1998, Fulton et al. 1998). We are concerned with Stage-1 processing which produces spatial rainfall estimates for a single radar domain. Within Stage-1 processing there are three processing levels. Level 1 consists of the raw analog output from the radar scanning process. Level-2 processing produces reflectivity estimates for every radial scan (5-10 minutes), in a polar grid of 82,800 bins representing 1° of arc and 1-km distance out to a radius of 230 km. Reflectivity is measured at multiple beam angles between 0.5 and 20° for each bin (Young et al. 1999). The processing program selects an appropriate beam angle for every bin based on a map of potential beam blockage for a given site location (Fulton et al. 1998).

The Level-2 processing program conducts a number of error checking procedures, and estimates precipitation within each bin using an algorithm that relates reflectivity (Z) to precipitation (R). The default equation relating Z and R is the power function, $R = aZ^b$, where $a = 0.017$ and $b = 0.714$. Z-R coefficients have been shown to vary as a function of many factors and previous studies have shown that it is not possible to derive a single equation that is accurate at every point in a given radar domain, and for every storm-type and storm intensity (Austin 1987, Hunter 1996, Glitto and Choy 1997, Anagnostou and Krajewski 1999, Ciach and Krajewski 1999, Ulbrich and Lee 1999). Level-2 processing also involves selection of a precipitation detection function (pdf) which establishes a threshold reflectivity, below which radar-rainfall estimates are set to zero (Anagnostou and Krajewski 1998, Fulton 1999, Kingsmill and Huggins 1999).

Level 3 processing aggregates and re-maps Level-2 precipitation estimates into hourly values within the Hydrologic Rainfall Analysis Project (HRAP) grid (Reed and Maidment 1999). The HRAP grid is a polar stereographic projection that covers the 48 contiguous states in the continental United States (Reed and Maidment 1999). HRAP cells are approximately 4 km x 4 km in size and have a row (1-861) and column (1-1121) designation relative to a reference cell (1,1) located just west of Baja California (Fulton 1998). Reformatting the data to fit the HRAP grid facilitates utilization of data for locations with overlapping radar coverage. Files from individual radar locations, however, are georeferenced relative to a local HRAP grid that contains only a 131-row x 131-column (17,161 cell) subdomain of the national-HRAP grid.

Figure 1. Radar (circle) and watershed (square) locations.

Hourly Level-3 radar data were obtained for the closest radar location to each of the ARS-watershed locations listed in Table 1 (Figure 1). Individual hourly-precipitation files were decoded using a computer program (decode.pl) which we modified from the original source code obtained from the NWS Hydrologic Research Laboratory in Silver Springs MD. This program converts binary-coded files into ASCII-formatted files that contain a precipitation estimate, in mm, for every row and column within the 17,161-cell local-HRAP grid domain. We also developed a computer program (gauges_lh.exe) using code from existing NWS algorithms to geo-reference radar and gauge data relative to both the local and national-HRAP grids. We encountered 3 versions of level-3 files that differ significantly in structure. The most current files are labeled DPA (for Digital Precipitation Array) and have a different structure than older files that are labeled HDP (for Hourly Digital Precipitation). A transition period also occurred when files with the HDP format were labeled as DPA files. The program "decode.pl" decodes older, newer, and transition files by calling specific subroutines depending upon the nature of the file that it is tasked to decode. If no precipitation is detected during a given hour, DPA and HDP-file headers contain a flag indicating that all of the precipitation values are zero.

Precipitation-gauges within each ARS watershed were georeferenced relative to the local-radar HRAP grid. A single mean-precipitation estimate was calculated for all gauge measurements that fell within a single HRAP-grid cell in a given hour. Significant periods in which NEXRAD data were not available were excluded from the analysis. Comparative statistics concerning radar and gauge estimates were averaged across all instrumented-grid cells.

Results

NEXRAD radar-precipitation records obtained from the NWS contain significant gaps in which no data are available. For the periods in which the radar data were evaluated, the NEXRAD system underestimated gauge precipitation for all location except Tucson (Table 1). The estimate improved somewhat when the period of consideration was limited to those hours in which the radar data were both present and flagged as operational. The ratio of radar to gauge data changed significantly when a comparison was made only for those hours in which radar data showed positive precipitation (Table 1). For hours with positive radar-precipitation estimates, the radar overestimated rainfall for 4 locations and underestimated at both Georgia locations, relative to gauge readings (Table 1). In all cases, the number of hours showing positive radar-precipitation estimates was very much less than the number of hours showing positive gauge-precipitation estimates (Table 1).

Discussion

Radar-data products are subject to three types of error that affect the accuracy of precipitation estimates: mean-field systematic bias, range dependent systematic error, and random error (Hunter 1996, Seed et al. 1996, Smith et al. 1996, Anagnostou and Krajewski 1998, Anagnostou et al. 1998, Anagnostou and Krajewski 1999, Ciach and Krajewski 1999, Steiner et al. 1999, Young et al. 1999, Young et al. 2000). Range dependent error in this study should be similar among gauges within a given location as the watersheds in question are all an order of magnitude smaller than the individual radar domain. Error detection and optimization or radar-precipitation products is almost always conducted by comparing radar-precipitation estimates with ground-truth gauge data (Smith et al. 1996, Anagnostou and Krajewski 1998, 1999, Seo 1998, Anagnostou et al. 1998, Anagnostou et al. 1999, Fulton 1999, Seo et al. 1999, Steiner et al. 1999, Seo et al. 2000a). Gauge data can be used to improve the accuracy of WSR-88D radar-precipitation estimates in two ways: development of more accurate Z-R relationships and pdf values for Level-2 data processing; and in post-estimate bias correction of Level-3 DPA data (Glitto and Choy 1997, Anagnostou and Krajewski 1998, Anagnostou et al. 1998, Seo 1998, Ciach and Krajewski 1999, Seo et al. 1999, Steiner et al. 1999, Ulbrich and Lee 1999, Ciach et al. 2000, Seo et al., 2000a).

Additional analysis of the Boise data indicated that the majority of gauge-precipitation, during hours where the radar reported zero precipitation, occurred at a rate below the precipitation detection function for NEXRAD. Post-processing bias adjustment with gauge data may be inappropriate for the Boise-radar data as the procedure assumes that all precipitation occurs at a rate that is higher than the precipitation detection function. Errors associated with low-rainfall rates may be under-represented in the literature as most radar-gauge comparisons focus on storm totals for higher intensity events or specifically ignore lower intensity events that occur during the test periods (Austin 1987, Anagnostou and Krajewski 1998, Baeck and Smith 1998, Brandes et al. 1999, Fulton 1999, Steiner et al. 1999, Ulbrich and Lee 1999, Seo et al. 2000a). Recently detected programming errors in the WSR-88D precipitation processing system have also been

Table 1. NEXRAD and gauge information by watershed location. Numbers in parentheses represent 1 standard error of the mean across all HRAP grid cells. Values for total precipitation are in mm. POR = period of record.

ARS location **Radar location**	**Boise, ID** **Boise (CBX)**	**Tucson, AZ** **Tucson (EMX)**	**El Reno, OK** **OK City (TLX)**	**Oxford, MS** **Memphis (NQA)**	**Tifton, GA** **Tallahassee (TLH)**	**Watkinsville, GA** **Atlanta (FFC)**
Start date	1-1-98	9-21-99	2-1-98	1-1-99	1-1-98	7-17-98
End date	12-31-00	12-31-00	12-31-00	12-31-00	11-28-00	3-30-01
HRAP cells	4	12	34	7	18	3
# gauges	4	50	34	28	28	5
Total hours in POR	26304	11232	25559	17548	25517	23690
Non-zero hours (NEXRAD)	207 (18)	154 (11)	386 (12)	483 (19)	442 (17)	491 (8)
Non-zero hours (Gauge)	845 (100)	248 (34)	719 (21)	1065 (296)	1116 (172)	1516 (215)
Precipitation (mm):						
Mean radar total (POR)	233 (41)	502 (67)	1326 (96)	1673 (43)	1242 (101)	1303 (62)
Mean gauge total (POR)	728 (114)	435 (51)	2315 (395)	2071 (122)	3158 (183)	2511 (83)
Mean gauge total (NEXRAD functional)	593 (95)	424 (54)	1933 (392)	1899 (121)	2618 (165)	2166 (68)
NEXRAD/gauge (mm/mm):						
N/G (POR)	33 (8)	116 (9)	58 (7)	81 (6)	39 (3)	52 (1)
N/G (NEXRAD functional)	40 (9)	119 (10)	70 (9)	89 (6)	44 (3)	60 (2)
N/G (NEXRAD Non-Zero)	123 (21)	165 (21)	128 (14)	116 (6)	59 (4)	94 (2)

determined to cause truncation errors that may systematically lower cumulative radar-precipitation estimates (Seo et al. 2000b). There were significantly fewer hours of NEXRAD-precipitation detection relative to gauge-precipitation detection for every radar evaluated in this study (Table 1). This indicates that the various radar locations were not detecting many events and their data would, therefore, also be unsuitable for bias adjustment with gauge data.

Our analysis indicates that NEXRAD radar data may not be suitable for long-term water balance and natural-resource modeling applications that require estimates of total annual rainfall. The utility of the data for modeling extreme-event flooding, runoff and erosion requires more detailed analysis of radar and gauge data for individual storms within a given watershed location. These data should also be evaluated for potential errors associated with beam blockage (Andrieu et al. 1997, Creutin et al. 1997, Maddox et al. 2002). Furthermore, additional research should be conducted to compare radar and gauge estimates in watershed locations with multiple, overlapping radar coverage.

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Substrate and Dendrochronologic Streamflow Reconstruction

David E. Grow

Abstract

Two piñon (*Pinus edulis*) tree-ring chronologies developed on each of three substrates (sandstone, shale, and alluvial fan deposits) in southern Utah for the period 1702 to 1997 demonstrate that geologic substrate affects dendrochronologic streamflow reconstructions. Chronologies from alluvial fan deposits explain the most variance of winter streamflow reconstruction (October 1 to May 31) with an adjusted coefficient of determination (R_a^2) equal to 0.59. Chronologies from sandstone deposits account for 52 percent of the variance, while those on shale deposits account for 45 percent. Correlation coefficients among the three substrates are significantly different at the 95% confidence level.

The highest single-site annual discharge reconstruction (October 1 to September 30), $R_a^2 = 0.25$, is provided by chronologies from shale deposits. The highest substrate-pair annual discharge reconstruction, $R_a^2 = 0.27$, is provided by chronologies from alluvial fan deposits. The highest summer reconstruction (July 4 to September 3), $R_a^2 = 0.14$, is provided by chronologies from sandstone. Over 90 percent of the summer reconstructions are below $R_a^2 = 0.10$.

The different substrate response is attributed to varying amounts of clay in each substrate affecting infiltration and available water for tree growth. [5]

Keywords: streamflow reconstruction, substrate, dendrochronology

Introduction

Dendrochronological streamflow reconstructions are a valuable tool to assess the long-term discharge behavior of a river. The long-term behavior can provide insights into the management of discharge, and is useful for planning and restoration projects.

Dendrochronological streamflow reconstructions have been performed since the mid 1930s. Early 1900s streamflow studies (Hardman and Reil 1936, Hawley 1937, Schulman 1945, Schulman 1951) were not strict reconstructions as the term is used today. These early studies generally compared tree-ring records with streamflow, and made estimates for wet and dry periods for pre-gauged streamflow.

Tree-ring growth is directly related to precipitation (Fritts 1976, Loaiciga et al. 1993). Streamflow reconstructions represent precipitation less water lost to evapotranspiration and storage (Jones et al. 1984, Meko and Stockton 1984). Therefore, the climate and vegetation peculiar to a specific basin will directly influence the dendrochronologic streamflow reconstructions for that basin. Fritts (1976) reports that substrate and soil differences affect tree-ring width. The substrate controls infiltration, local drainage, and nutrient supply to the tree. A tree is therefore an integrator of the local environment, and the tree-ring record reflects not only precipitation but also the substrate on which the trees are growing.

The objective of this study is to address the effects of substrate on dendrochronological streamflow reconstructions. Geological substrate controls local hydrological systems. Drainage characteristics peculiar to different substrates are reflected in the tree-ring record, and trees on a particular substrate produce a chronology that provides improved streamflow reconstructions over trees on other substrates.

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The area chosen for this study is the Paria River basin in southern Utah and northern Arizona (Figure 1). The widespread presence of piñon and exposure of geologic strata provide an opportunity to address the effects of substrate on tree-ring chronologies and streamflow reconstructions.

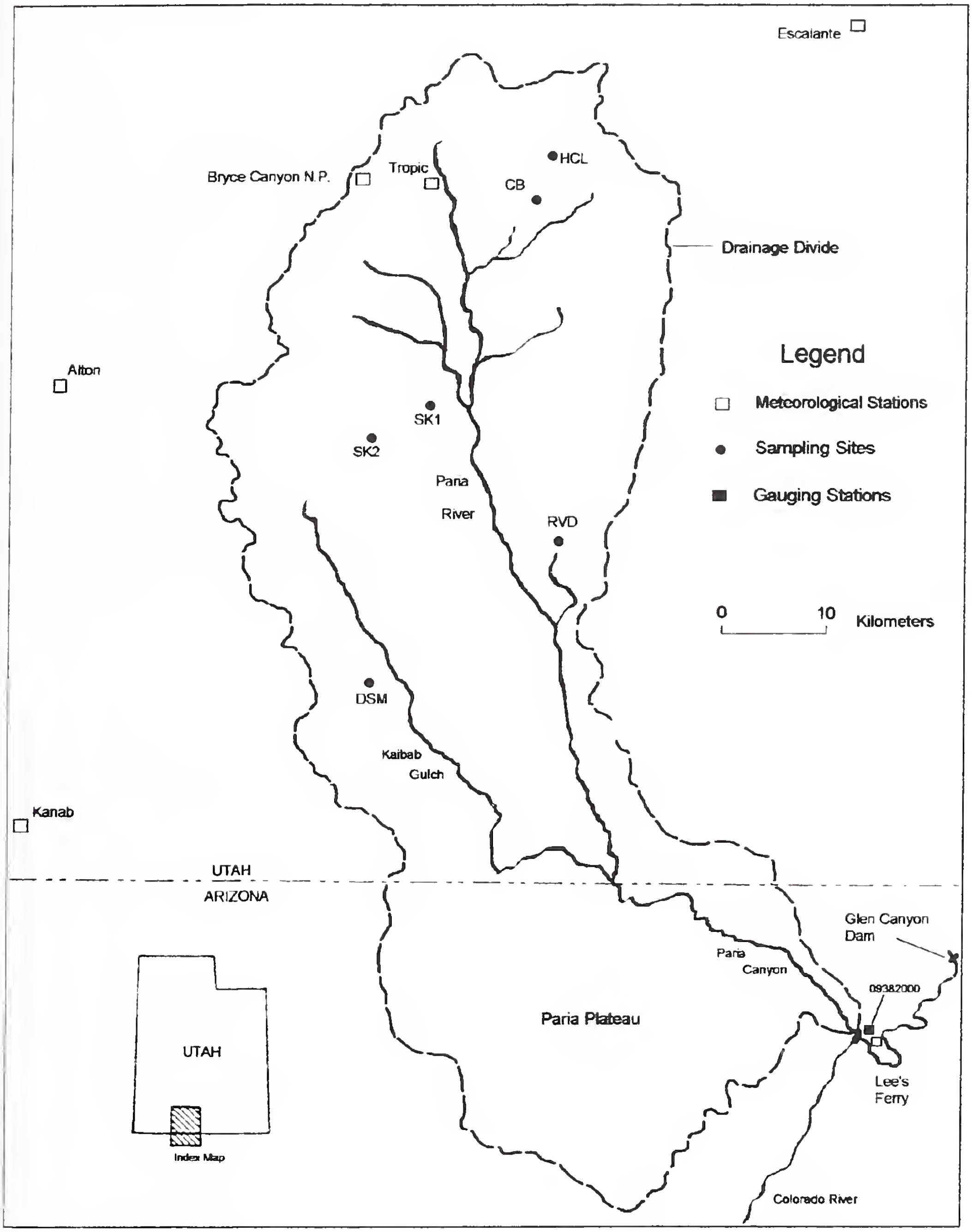

Figure 1. Paria River Basin in southern Utah showing locations of tree-ring sampling sites. Coal Bench (CB) and Henderson Canyon Lower (HCL) are located on alluvial fan deposits. Skutumpah Road site 1(SK1) and Skutumpah Road site 2 (SK2) are located on shale. Round Valley Draw (RVD) and Deer Springs Mesa (DSM) are located on sandstone.

Methods

Six tree-ring standard chronologies (indices) were developed for this analysis. A minimum of 10 trees, 2 cores per tree, was sampled at each site. Samples were prepared and mounted according to procedures described by Stokes and Smiley (1996). Cores were crossdated using skeleton plots, and crossdating was verified by Laboratory of Tree-Ring Research personnel. Ring widths were then measured to within +/- 0.01 mm. A standard chronology was created by removing differential growth trend among trees using a cubic smoothing spline. To obtain tree-ring indices of equal length for comparison, the six different chronologies were truncated so that each chronology spanned the period from 1700 to 1998.

Substrate characteristics

Soils throughout the basin are predominantly fine, sandy loams, very deep, and well drained (Swensen and Bayer 1990). The tree-ring sample sites are located on three different soil series (Table 1). Sites HCL and CB are located on the Hernandez-Clapper Series; DSM and RVD on the Podo Series; and SK1 and SK2 on the Cannonville Series. The Hernandez-Clapper series is formed in alluvium from sandstone and limestone. The Podo series is formed from sandstone residuum and alluvium. The Cannonville series is formed from shale residuum. The clay content of the soil series ranges from 5 to 50%, and permeability ranges from 0.15 to 15.24 centimeters per hour. These features affect the infiltration capacity, hydraulic conductivity, transmissivity, and available water capacity of the different substrates (Birkeland 1984, Ritter et al. 2002, Brooks et al. 2003). All samples were taken on relatively flat aspects of each substrate.

Table 1. Sampling site soil summary (Swenson and Bayer 1990).

Site	Clay Content (%)	Permeability (cm/hr)
CB	18-27	1.52 - 5.08
HCL	18-27	1.52 - 5.08
DSM	5-25	5.08 - 15.24
RVD	5-25	5.08 - 15.24
SK1	40-50	0.15 - 0.51
SK2	40-50	0.15 - 0.51

Streamflow discharge records for the period from 1924 to 1998 were obtained from U.S.G.S. gauging station 09382000 located at Lee's Ferry, Arizona. The total streamflow discharge for a year is based on the water year, October 1 through September 30. The water year was partitioned into three sub-periods: 1) October 1 through March 31 (Winter 1), 2) October 1 through May 31 (Winter 2), and 3) November 10 to April 17 (Winter 3). The annual and the Winter 2 partitions are the subject of this study.

Streamflow reconstructions

Multiple linear regression was used to estimate past streamflow. The chronologies were segregated by substrate: CB and HCL are on alluvial fan deposits, DSM and RVD on sandstone, and SK1 and SK2 on shale. Models of pre-gauged streamflow were developed by comparing the gauged discharge for each year with tree-ring indices for each year, with up to +/- 2 year lags.
The coefficients of determination were adjusted to account for the loss of degrees of freedom due to the addition of predictors (Weisberg 1985). The validity of each model was determined by examining the estimated model coefficients, the residuals from modeling, the root-mean-square-error (RMSE) of calibration and verification, and the reduction of error (RE) statistic of calibration and verification. Each model was verified using the PRESS statistic (Weisberg 1985).

Results and Discussion

The highest adjusted coefficients of determination (R_a^2) show that the Winter 2 partition provides the highest R_a^2 values, with the paired sites CB/HCL providing the highest discharge reconstruction ($R_a^2 = 0.59$). The differences in correlation coefficients are statistically significant at the 95% confidence level (□ = 0.05) (Table 2).

Table 2. Discharge reconstruction summary (R_a^2) for the annual (October 1 - September 30) and Winter 2 (October 1 - May 31).

Site	Annual	Oct – May
CB	0.22	0.46
HCL	0.24	0.54
DSM	0.11	0.43
RVD	0.18	0.48
SK1	0.23	0.43
SK2	0.25	0.45
CB-HCL	0.27	0.59
DSM-RCD	0.18	0.52
SK1-SK2	0.25	0.45

Clay content and permeability are highest for sites SK1 and SK2, ranging from 40-50% clay content and 0.15 to 5.0 centimeters per hour permeability. Sites DSM and RVD are located on sandstone residuum. Compared to alluvial fan and shale substrates, clay content is low, 5 - 25%, and permeability is high, 5.0 to 15.0 centimeters per hour. Sites CB and HCL are intermediate between the other two sites with clay content from 18-27%, and permeability from 1.5 to 5.0 centimeters per hour.

Substrate appears to play a major role in the streamflow reconstructions. The extremes of the infiltration rates of sandstone and shale, 5.08-15.24 cm/hr and 0.15-0.51 cm/hr, respectively, bracket the infiltration rate of 1.52-5.08 cm/hr for the alluvial fan deposits (Figure 2). The extremes represent end-points of water availability for tree growth. The lower infiltration capacity of the shale deposits may result in rapid surface runoff before the precipitation is recorded in the tree-ring record. The higher infiltration rates of the sandstone may result in water passing through the system vertically, again before being recorded in the tree-ring record. The alluvial fan deposits, being intermediate in infiltration, provide the substrate texture more conducive to water availability for tree growth, and is subsequently reflected in the tree-ring record.

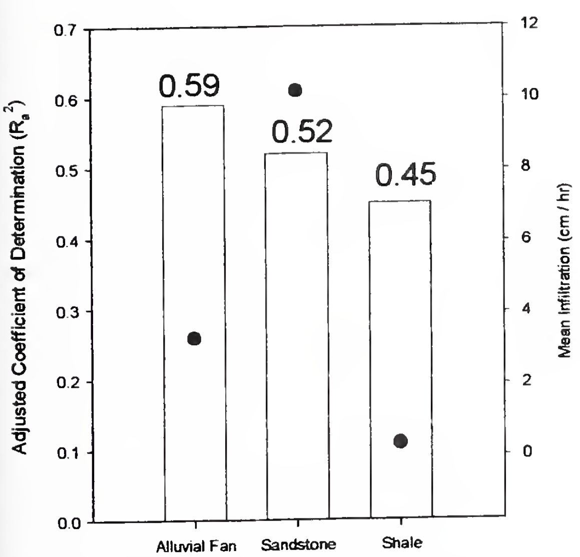

Figure 2. Substrate versus the adjusted coefficient of determination (bars) for the Winter 2 discharge reconstruction with mean infiltration (circles).

Conclusions

Several factors influence tree growth and chronology development. This study has successfully compared geologic substrates with respect to tree-ring chronology development and streamflow reconstructions using multiple linear regression. The alluvial fan deposits generally provide the highest coefficient of determination values for streamflow reconstruction. These results suggest that substrate affects the available water for tree growth, and subsequently affects streamflow reconstructions. This information may prove useful to land managers for planning and restoration purposes.

This study provides a foundation to expand the substrate/species component of dendrochronological streamflow reconstructions. Future work on this topic should include more species and substrate comparisons.

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Long-Term Rainfall and Runoff Characteristics of a Small Southern Piedmont Watershed

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Abstract

The long-term hydrologic response of a small (7.8 ha) zero-order Southern Piedmont watershed were analyzed from 1940-1984. Four land use phases occurred during this period: row cropping (5-yr), kudzu (5-yr), grazed kudzu mixed with rescuegrass (7-yr), and grazed coastal bermudagrass (28-yr). Land use and rainfall variability influenced runoff characteristics. Row cropping produced the greatest runoff, percentage runoff, and peak flows. Kudzu reduced spring runoff and almost eliminated summer runoff, as did a mixture of kudzu and rescuegrass; however, rainfall was reduced during these two phases. Peak flows were also reduced during these two phases. Bermudagrass reduced runoff more than row cropping but not as much as kudzu or kudzu mixed with rescuegrass. Peak flows increased during grazing of bermudagrass but stayed below those during the cropping phase. Monthly rainfall-runoff regression relationships were developed and R^2 greater than 0.60 were found for: cropping phase - summer, spring and winter; kudzu phase - spring and winter; kudzu-rescuegrass phase - summer, spring, winter; and bermudagrass phase - summer and spring. Long-term information such as this is needed for various environmental and management decisions faced by land managers today.

Keywords: experimental watershed, rainfall-runoff, Southern Piedmont, kudzu, bermudagrass

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Introduction

The USDA-Agricultural Research Service has been operating a network of geographically distributed experimental watersheds of various sizes since the early 1950s. Some were established in the 1930s by the Soil Conservation Service. These watersheds are resources of historic hydrologic data and knowledge (Goodrich et al. 1994). Part of this network is the 7.8 ha (19.2 ac) zero-order watershed known as W1 established in 1939 near Watkinsville, GA. The topography, soil, and land use characteristics of W1 are typical of many sloping fields throughout the Southern Piedmont, a 16.5 million ha (40.7 million ac) region in the southeastern United States (Carreker and Barnett 1953). The region is one of the most severely eroded parts of the United States as a result of over two centuries of row crop agriculture, frequent intense summer storms and easily erodible soils. The need to study the rainfall-runoff characteristics of typical Piedmont fields under varied cropping systems in order to develop predictive capabilities and recommendations for land management and conservation measures spurred the establishment of W1. In this paper we summarize 45 years of rainfall-runoff data under 4 different land uses lasting 5 to 28 years each, and focusing on monthly, seasonal and annul relationships.

Methods

Data sources

Data were acquired for the 1940 to 1984 period from various sources, including ARS Water Data Center, Hydrology Laboratory, Beltsville, MD, annual reports (e.g., SPCRU 1970), and several miscellaneous USDA-ARS annual publications (e.g., Burford et al. 1982).

Watershed description and history

The zero-order W1 watershed is pear shaped with slopes from 3 to 10% and an average of about 7%. Moderately eroded Cecil and Pacolet soils occupy about 69 and 31% of the watershed, respectively (R.R. Bruce, unpublished report). The Pacolet soil is confined to the upper part of the watershed. Both soil series have developed in residual felsic igneous and metamorphic parent material such as granite, granite gneiss, mica gneiss, and mica schist (Radcliffe and West 2000). The Pacolet soils have less thickness than those of the Cecil but the properties of the two soils are similar otherwise. The soils generally have brownish-gray sandy loam to red clay loam surface horizons overlaying red clayey argillic horizons. The Bt horizon at W1 begins less than 36 cm from the surface in the Pacolet, but is 36 to 66 cm from the surface in the Cecil (R.R. Bruce, unpublished report). The subsoil in both soils is underlain by a C horizon, several meters thick, of porous decomposed rock material resting on solid rock at 5 to 30 meters from the surface.

When established in 1940, W1 had residual vegetated bench terraces constructed several decades earlier by farmers and used for row crop farming. Terraces were removed in 1957 by spreading the spoil over the immediate area. Land use was as follows: 1940-44, cotton-oats-cowpeas on a two year rotation; 1945-49 kudzu with corn in the first year; 1950-56 kudzu mixed with rescuegrass with light controlled summer kudzu and winter rescuegrass grazing; and 1957-84 grazed coastal bermudagrass and winter annuals.

Rainfall-runoff measurement

Rainfall was measured with a chart-based Fergusson-type weighting and recording rain gauge (Carreker and Barnett 1953). A 2 to 1 concrete broad-crested V-notch weir fitted with a chart-based Friez-type Fw-1 water-level recorder was used to record runoff. Calibration for the stage-flow relationship of the weir included correction for ponding. Data acquisition was discontinued after 1984. However, in June 1998, the weir was rehabilitated and data collection resumed with an automated system.

Data analysis

We used Excel, SigmaPlot, SAS and TableCurve3D to summarize statistical properties and establish relationships between parameters.

Results

Rainfall

Mean annual rainfall for 1940-84 was 1257 mm varying from a low of 853 in 1954 to a high of 1838 mm in 1964. Mean monthly rainfall was least in October (71 mm; CV 73%) and highest in March (149 mm; CV 45%). Seasonal rainfall decreased from winter (mean monthly 116.5 mm) to spring (April/May mean 105 mm) to summer (mean monthly 103 mm) and fall (mean monthly 80 mm).

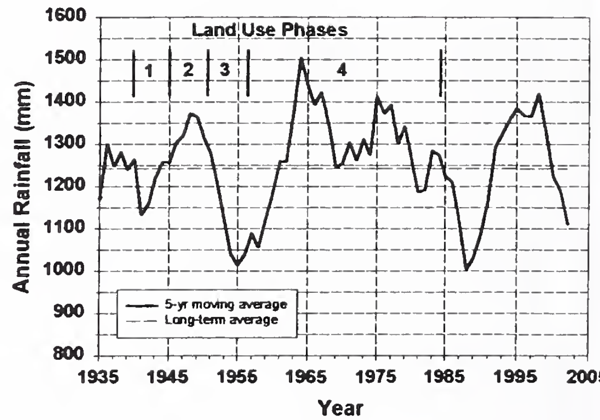

Figure 1. Five years moving average of annual rainfall starting in1935 for Watkinsville, GA.

During each of the four land use phases there were distinct rainfall patterns that influenced runoff as seen in the 5-yr moving averages (Figure 1). The 5-yr moving average of annual rainfall best reflected the trend in periodic drought and wet periods. The 1940-44 phase began with rainfall deficit compared to the long-term average but the deficit was eliminated by 1944. The next phase (1945-49) had above average annual rainfall, which led to rainfall surplus at the end of the period. Seven years of below average annual rainfall led to significant rainfall deficit during the next phase (1950-1956). The most severe drought in Georgia since official records began in 1892 occurred in 1954-55 (Plummer 1983). After 1957, a long period of average or above average rainfall ensued and lasted through 1984 with few years of below average annual rainfall.

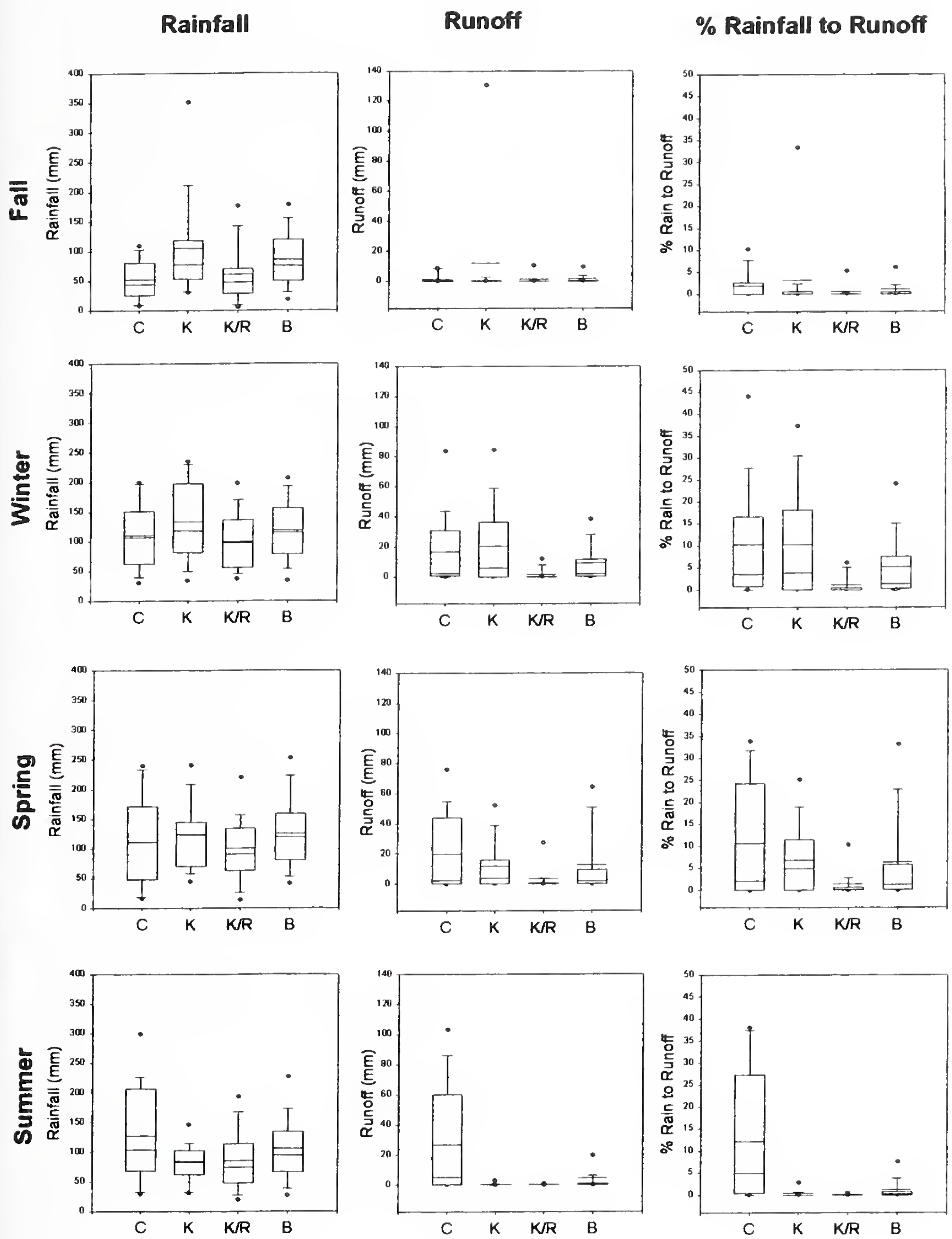

Figure2. Distribution of seasonal monthly rainfall, runoff, and percent runoff from 1940 to 1984, during each of four land use phases at W1 watershed. Land uses are designated as C for cropping, K for kudzu, K/R for kudzu-rescuegrass, and B for bermudagrass. September, October, and November are considered as fall season. December of a previous year, January and February are taken as winter season. Spring consists of March, April and May. Summer occurs in June, July and August. Each box shows the 25th percentile, median, and 75th percentile. Whiskers show the 10th and 90th percentile. Outliers up to the 5th and 95th percentiles are shown as dots. Means are shown as dotted lines inside boxes.

Monthly runoff

The distribution of seasonal monthly runoff for each land use phase (C = Cropping, 1940-44; K = Kudzu, 1945-1949; K/R = kudzu rescuegrass, 1950-56; B = bermudagrass, 1957-84) interacted with rainfall (Figure 2). Seasons were taken as: fall - September, October and November; winter - December of a previous year, January and February; spring - March, April and May; and summer - June, July, August.

The cropping-based first phase exposed the soil to high intensity spring and summer storms, which led to high runoff. Winter rains saturated the soil and also produced runoff. The fall rains were generally below the threshold for initiating and sustaining runoff. Runoff was highest in summer (mean 26.6 mm) followed by spring (mean 20.0 mm), winter (mean 16.8 mm), and fall (mean 1.4 mm) during the first land use phase. Runoff as a percent of rainfall was highest also during the row crop phase. Means in mm were: summer, 12.5; spring, 10.6; winter, 10.3; and fall, 1.9.

After kudzu (1945-49) became fully established, summer runoff was dramatically reduced and spring runoff was reduced by almost half (runoff means: summer-0.3 mm, and spring-11.7 mm; percentage runoff means: summer-0.3, spring-6.7). The winter runoff was similar to the earlier phase, with slightly increased runoff during phase 2 (means: cropping-16.8, kudzu-20.2; runoff percentage about 10.2 for both). Fall runoff during the kudzu period was almost non-existent except for one extreme event (Fall-Runoff, Figure 2), despite a doubling of rainfall compared to the first phase. The reduction in runoff while in kudzu production is partly attributed to a reduced summer rainfall (mean 127 vs. 83 mm). During kudzu-rescuegrass (1950-56) production, a combination of good ground cover, and below normal rainfall led almost to no seasonal runoff. Annual runoff accumulation was 207.8 mm/yr during the row cropping ($r^2 = 0.99$), 127.8 mm/yr during the kudzu ($r^2 = 0.96$), but only 19.3 mm/yr during the kudzu-rescuegrass period ($r^2 = 0.97$). Corresponding annual rainfall accumulations in mm/yr were 1265, 1315, and 1044, respectively. Runoff was 3% or less in fall, spring and summer, and 5% or less in winter in 90% of the events.

A transition period occurred at the beginning of the final land use phase when terraces were removed in 1957 and coastal bermudagrass established. Full grazing of the watershed started in 1960. Winter annuals were used to supplement the summer bermudagrass grazing. The terrace removal would have contributed to the potential for increased runoff. Runoff increased compared to the 1950 to 56 period, but was less than the first 2 land use phases (except that summer runoff was larger than kudzu summer period also). Annual runoff accumulation after 1957 was 83 mm/yr ($r^2 = 0.96$) with annual rainfall accumulation of 1344 mm/yr ($r^2 = 0.99$). Runoff was minimal in the fall. Mean runoff was: summer, 1.2%; spring, 6.4%; winter, 5.1%; and fall, 1.1%.

Annual runoff and peak flow rates

The median annual runoff was 5% the annual rainfall. Annual runoff was 10% or less 75% of the time and only about 10% of the time did it reach between 15 and 25% of annual rainfall. Peak runoff rate reached 1472 L/s (52 cfs) and occurred in 1945. However, the median peak flow rate was about 300 L/s (10.5 cfs). Peak flow rates were highest during the 1940-46 period with mean of 954 L/s (33.7 cfs) and the coefficient of variation of 36%. This period included the first two years of kudzu when the plant had yet not achieved full ground cover. Mean peak flow rates plummeted to 119 L/s (4.2 cfs) with coefficient of variation of 76 for 1947-56. Peak flow increased during 1957-84 when mean peak flow rate was 380 L/s (13.4 cfs) and the coefficient of variation was 81%.

Rainfall-runoff correlation

Non-linear regression analysis showed that except for fall, relationships between monthly rainfall and runoff had R^2 greater than 0.6 (Figure 3). Coefficient of determination was highest for the cropping phase. The spring and winter bermudagrass phase data showed a larger scatter, which led to R^2 of 0.49 for winter. The scatter is expected because of the long duration of this phase. The steep rising arm of the fitted line for the kudzu-rescuegrass-spring correlation is largely the effect of a single influential observation.

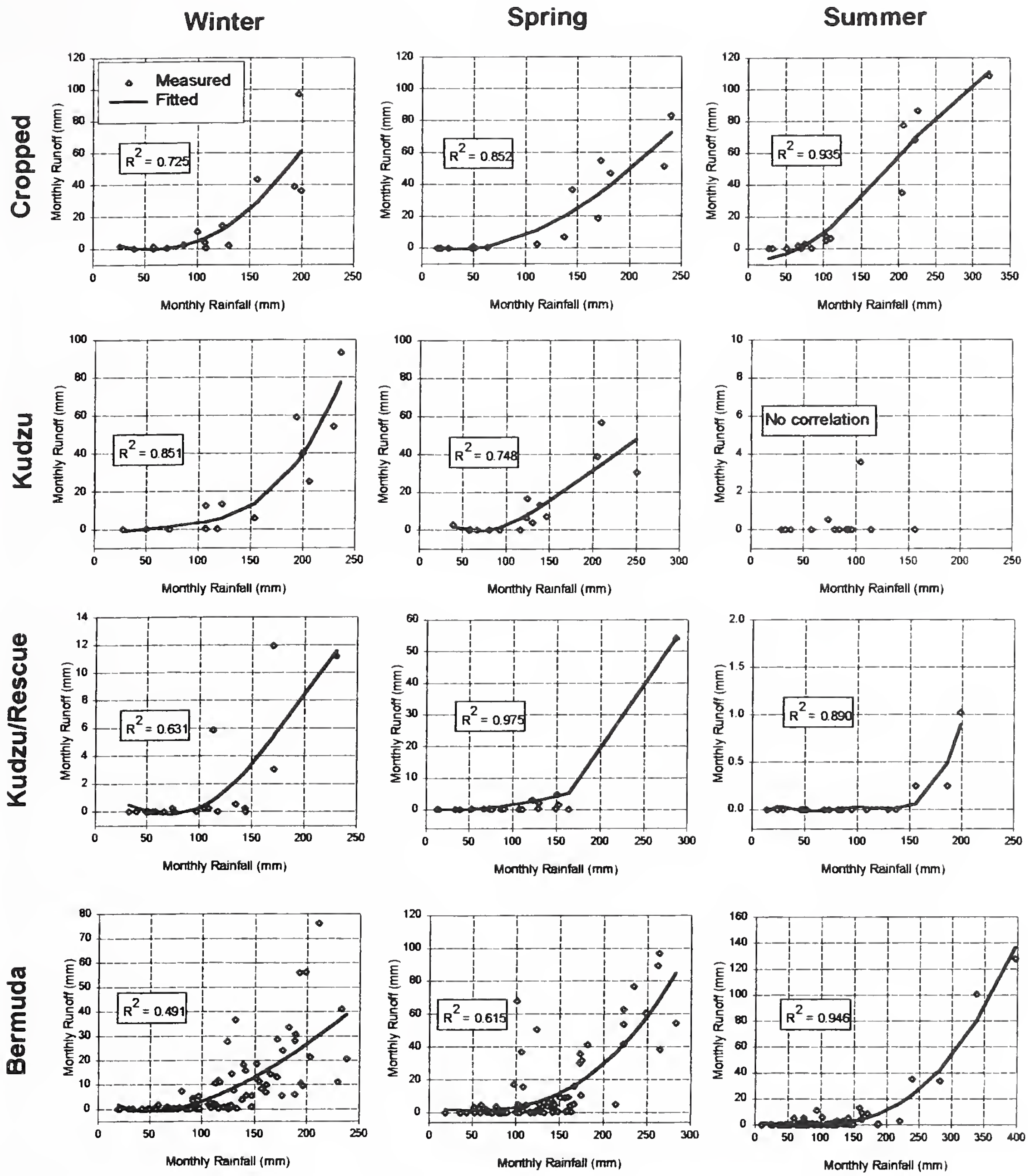

Figure 3. Relationships between seasonal monthly runoff and rainfall at W1 watershed from 1940 to 1984, during each of four land use phases. Coefficient of determination R^2 is given for each correlation. The diamonds and solid lines represent measured and fitted values, respectively.

Recent rainfall-runoff monitoring

Land use at W1 has continued as bermudagrass and winter annual-based grazing since 1984. Runoff has been limited since monitoring resumed in 1998, because of another relative drought in the Southeast from May 1998 through the end of 2002 (Figure 1).

Conclusions

Although this analysis is limited by the relatively short records for some of the land uses, it is clear that relatively subtle differences in land use have significant hydrologic ramifications at the small watershed scale. The data demonstrate how land use can be utilized as a management tool for resources protection. In Southern Piedmont, clean-tilled row cropping has clearly the potential for high runoff and consequently for land degradation. Year round ground cover offers the best protection for Southern Piedmont farm lands. This is particularly true in spring and summer when ground cover can be used to absorb part of the energy of the typically high intensity rainfall of these seasons and reduce potential runoff. Rainfall variability is an inherent part of the environmental attribute of Southern Piedmont. The seven years of below normal rainfall during the kudzu/rescuegrass phase contributed to the reduced runoff. What would have been the response had W1 been in row cropping? Hendrickson and Barnett (1963) reported on runoff research done from 1940 to 1954 on 6.32 m (20.74 ft) wide by 21.34 m (70 ft) long plots of 7% slope in continuous cotton located in close proximity to W1. They found that average annual runoff was 22% of the annual rainfall during the 15-yr period. Average runoff equivalent to phases 1, 2 and the first 5 of the 7 years of phase 3 of this paper were 22.7%, 21.3%, and 21.8%, respectively. There was no runoff reduction from these plots. While there are scale differences between the plots and W1, it is reasonable to conclude that runoff would have been much more from W1 during phases 2 and 3 had these phases been in cropping. Runoff however still occurs even under full ground cover as born out by the 28-yr data of the bermudagrass phase. The median annual percent runoff was 5% with 10% probability for 16% or higher runoff peaking at 23%. We analyzed only monthly, seasonal and annual rainfall and runoff data and did not go into individual storms. Nevertheless, long-term information such as this is needed for various environmental and management decisions faced by land managers today.

Acknowledgments

Special thanks are extended to the dedicated professionals and early pioneers led by A.P. Barnett and J.R. Carreker who established the Watkinsville W1 watershed, maintained, collected, analyzed and reported the data. We appreciate A.L. Dillard, C.R. Jackson, D.E. Radcliffe and M.H. Young for providing valuable suggestions during the preparation of the manuscript.

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Evaluation of Dielectric Constant-Based Soil Moisture Sensors in a Semiarid Rangeland

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Abstract

In winter 2002, nineteen Stevens Vitel Hydra soil moisture probes were installed at the USDA-ARS Walnut Gulch Experimental Watershed to provide surface soil moisture data for use in calibrating remote sensing instruments. At three sites, two additional probes were installed at depth to provide a profile of soil moisture. The probes accurately measure soil moisture after applying a linear regression to match Vitel volumetric water content with gravimetrically sampled VWC. Probes at 5 cm and 15 cm responded quickly to larger rainfall events, while the one at 30 cm showed a delayed and gradual response. The optimal sampling interval was about 5 minutes during a rainfall event at 5 cm and 15 cm and no less than 30 minutes at 30 cm depth. During dry periods, the probes may be sampled at longer intervals, 30 minutes or greater, with no loss in data quality. Soil water was redistributed from the surface to 30 cm depth during the summer rainy season, and to 15 cm depth during the winter rainy season.

Keywords: Vitel probe, soil moisture, sensor, dielectric constant

Introduction

Soil moisture can be an important factor for land managers to consider when making decisions concerning livestock grazing patterns, crop planting and irrigation scheduling, and soil stability for machinery traffic. Many methods of determining soil moisture have been developed, from simple manual gravimetric sampling to more sophisticated remote sensing and Time Domain Reflectometery (TDR) measurements. One common technique is to measure dielectric constant, that is, the capacitive and conductive parts of a soil's electrical response. Through the use of appropriate calibration curves, the dielectric constant measurement can be directly related to soil moisture (Topp et al. 1980).

Dielectric constant may be measured in a variety of ways. Soil moisture probes, designed to be buried and left in-situ, are commercially available. Satellites such as RADARSAT, using synthetic aperture radar, can indirectly measure the dielectric constant of the soil due to its direct effect on microwave backscatter (Henderson and Lewis ed. 1998). Because the soil probes and radar both measure dielectric constant, less error is introduced when comparing one to the other. Soil moisture may also be remotely sensed using a passive microwave radiometer such as AMSR-E on the recently launched Aqua satellite. AMSR-E covers a larger footprint than RADARSAT, and uses an algorithm based on a radiative transfer model, rather than dielectric constant, to determine soil moisture (Njoku 1999). Remote sensing instruments can produce measurements of surface (from a few mm to ~5 cm depth) soil moisture at a large spatial scale but only at occasional times, while in-situ sensors measure soil moisture at a point, can be installed at depth (> 5 cm) in the soil matrix, and can sample nearly continuously. Therefore, soil moisture probes are often used as calibration checks for remote instruments.

In this study, soil moisture was measured by soil moisture probes over a twelve month period, incorporating both winter and summer moisture regimes, the dominant precipitation periods for southeastern Arizona. Winter precipitation events (Nov

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– Apr) are characteristically frontal systems originating in the Pacific. These slow moving storms cover large areas, and produce low intensity precipitation (<25 mm/hr). Precipitation from these storms, which usually do not generate runoff, combined with low evapotranspiration (ET) demand during these months, increases soil water content in the near surface layers, which may remain elevated for months (Scott et al. 2000). Summer precipitation comes from convective storms mainly during the North American Monsoon. They are usually limited spatially and temporally, and of high intensity (>25 mm/hr). Due to high ET demand and Hortonian infiltration-excess generated surface flow, most precipitation runs off immediately, rapidly evaporates from the surface, or is transpired by plants. Therefore, the soil water content during summer can change rapidly and dynamically within soil layers in response to a precipitation event.

The primary objective of this study is to ensure that the data collected from soil moisture probes installed at WGEW is of sufficient quality and quantity to aid future research at the Watershed. Specifically, we seek to: (1) assess the accuracy of dielectric constant-based soil moisture probes through comparison with gravimetric samples, (2) optimize the sampling interval of each probe in order to maximize the collection of useful data, and (3) investigate soil water redistribution following precipitation events in winter and summer. Data collection and assessment are ongoing. Due to the lack of precipitation events, particularly winter events, during the study period, the results presented are preliminary and subject to revision.

Methods

In February 2002, 19 Stevens-Vitel Type A Hydra soil moisture probes (commonly referred to as Vitel probes) were installed at the USDA-ARS Walnut Gulch Experimental Watershed to provide in situ surface soil moisture measurements as part of the NASA-AMSR Aqua Project (http://www.nasda.go.jp/projects/sat/aqua/launch/index_e.html). Probes were co-located with established WGEW rain gages to facilitate data collection and provide reference rainfall data. To supply data representative of the soil moisture measured by the AMSR-E instrument, one probe was installed at a depth of 5 cm at each site. To assess the redistribution of water within the soil profile, additional probes were located at depths of 15 cm and 30 cm at three of the rain gage (RG) sites (46, 82, and 83). All probes were located at sites lacking canopy cover, with the exception of RG 46, which is grass-dominated. Sites were selected to provide large areal coverage and be representative of the soils present at the watershed. Bulk density measurements were made at each site at the time of installation.

From the time of installation until January 2003, soil moisture was sampled every five minutes, with the average logged at thirty minute intervals. In February 2003, the sampling rate for the three profile sites was modified to log data every five minutes. To provide reference soil moisture values, three gravimetric soil samples were taken from the top 5 cm at each site following most precipitation events. Samples were taken from an area representative of the probe location and in close proximity in order to minimize the effect of spatial variability, which may be significant beyond one meter (Whitaker et al. 1991). The average gravimetric water content was converted to volumetric water content using the measured bulk density at each site.

Volumetric water content (VWC) was derived from dielectric constant measured at each probe using calibration curves provided by Stevens-Vitel (1994) for sand, silt, and clay soils. Using a linear fit, gravimetric VWC samples were regressed on the output of each of the three calibration curves. The linear regression provided a means to correct the Vitel VWC to more closely match the gravimetric data.

For sampling rate evaluation, the response time of a probe was calculated as the time from the first measurable rainfall (greater than 0.254 mm) until an increase in VWC was observed. Evaluation of soil water redistribution was facilitated by transforming VWC to the volume of water per unit area. The probe at 5 cm depth was assumed to represent soil water from 0 – 10 cm, the probe at 15 cm depth from 10 – 22.5 cm, and the probe at 30 cm from 22.5 – 37.5 cm. Therefore, the volume of water contained in each depth interval is the thickness of section multiplied by VWC. The minimum water content at each probe, as recorded during the course of a year, was subtracted from the measured water content to represent the residual volume of water in a dry soil profile.

This paper focuses on soil moisture measurements and rainfall events at the three profile sites, 46, 82, and 83. It should be noted that there was virtually no precipitation in 2002 after the probes were installed until the onset of the monsoon in July 2002. Due to the homogenous nature of the soil in the top 30 cm of the soil profile at RG 83, located at Lucky Hills, additional analysis will focus on data collected at that site. The soil at this site is the Lucky Hills-McNeal complex, a very gravelly sandy loam comprised of mixed calcareous alluvium (Breckenfeld et al. 2000).

Results and Discussion

Vitel probe calibration

Three calibrations for the Vitel probe are provided by Stevens Vitel for sand-, silt-, and clay-dominated soils to transform the dielectric constant to soil moisture. Each calibration curve was applied to each sensor at the three profile sites. It was found that in nearly all cases the Vitel soil calibration under-estimated the volumetric water content, as determined by gravimetric sampling. Pending completion of on-going site-specific calibrations for each probe, a linear regression was used at the profile sites to transform the Vitel probe output to more closely match the gravimetrically determined volumetric water content values (RG 83, Figure 1).

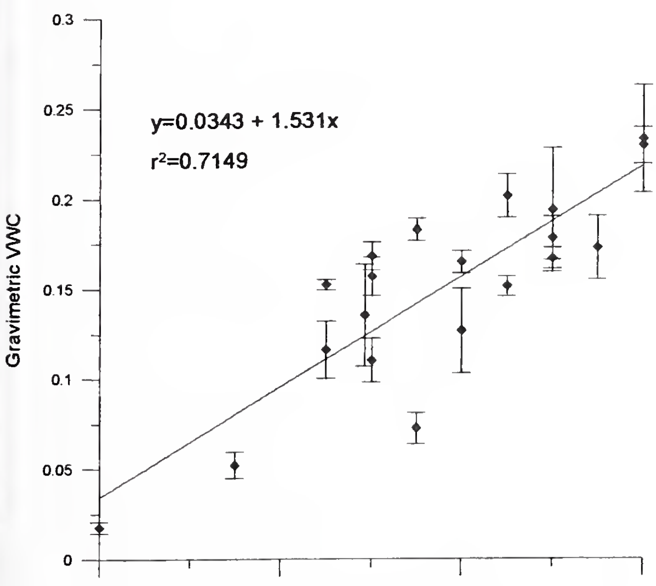

Figure 1. Linear regression of gravimetric VWC vs. Vitel VWC at RG 83. Error bars show root mean square error among sets of gravimetric samples.

At gages 82 and 83, the regression was applied to the sand-dominated calibration; at gage 46 the clay-dominated calibration was used. The regression was based on gravimetric sampling of the top 5 cm, and an assumption is made that bulk density is constant throughout the profile. At RG 82, the regression was similar to that at RG 83:

$$y = 0.0482 + 1.273x \qquad (r^2 = 0.750) \qquad (1)$$

A poor correlation was seen at RG 46 ($r^2 = 0.268$). This is likely due to the large variance among gravimetric samples and the shrink-swell properties of the soil. The largest root mean square error (RMSE) in VWC of a set of gravimetric samples at RG 46 was 0.18 $m^3\ m^{-3}$, with a mean RMSE of 0.06 $m^3\ m^{-3}$.

Soil moisture measurements

Differences in soil moisture response from winter and summer precipitation events were evident at all three of the profile sites. Figure 2 shows a representative response for the probes at each of the three profile sites for a summer and a winter event. Soil at RG 46 is a clay loam, which results in the highest measured VWC of any probe. During the summer event (top row), a rapid response to precipitation can be seen at 5 cm and 15 cm, typical of most summer events. No immediate response is seen at 30 cm during the summer; however, a delayed and gradual response at this depth to the precipitation event at RG 83 in Figure 2 is seen at another scale in Figure 3. This event, with a cumulative precipitation of 30.5 mm over 3.5 hours, produced the most rain of any event in 2002. It is nearly typical of the maximum 2-year return period storm at WGEW (Osborn et al. 1980). Therefore, the immediate active depth of infiltration for most individual summer events appears to be between 15 and 30 cm. This is similar to the response of TDR probes at a nearby site for summer precipitation (Canfield and Lopes 2000, Scott et al. 2000).

The limited number of precipitation events during winter 2003 (Figure 2, bottom row) show a gradual response at the 5 cm level, little or no response at the 15 cm level, and no response at the 30 cm level. However, interannual winter precipitation varies greatly, influenced by El Niño-La Niña episodes, and VWC is known to increase at deeper layers during El Niño winters (Scott et al. 2000).

Sampling interval

The soil moisture sampling interval is often a compromise between too much data during periods of little or no change in soil moisture and not enough data during periods of rapid changes within the profile. The initial sampling interval of 30 minutes was selected for

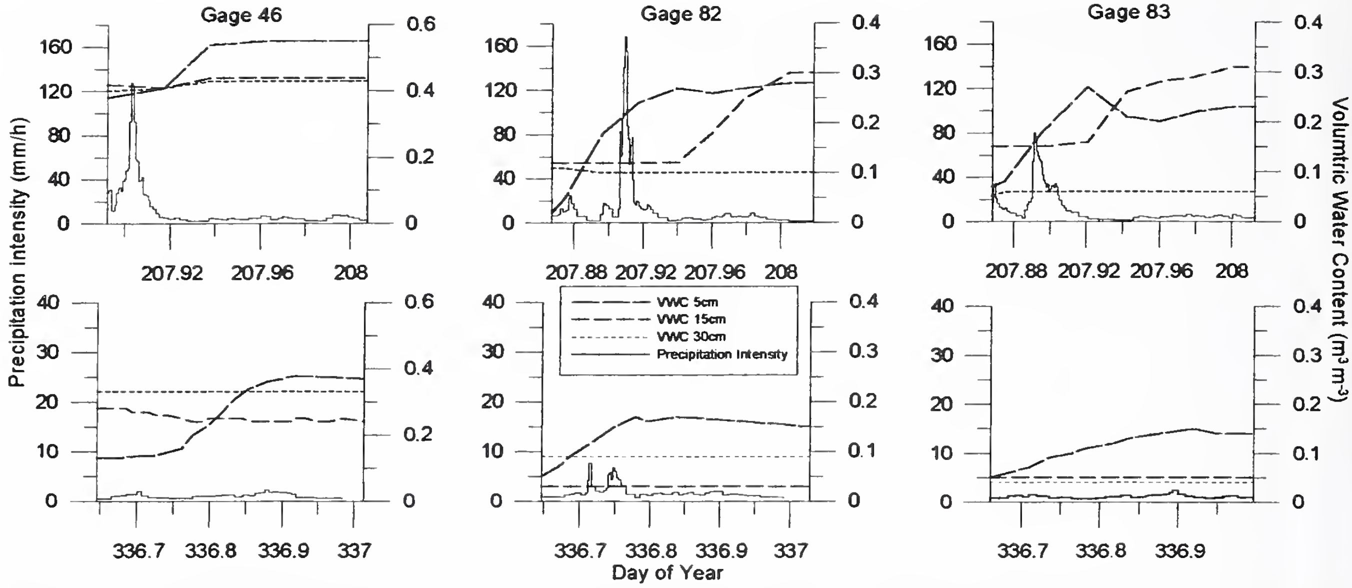

Figure 2. Precipitation intensity and VWC vs. time showing response to a single winter and summer event. Note differences in vertical scale.

several reasons, including consistency with other ARS locations in the AMSR project, and to minimize data logger storage and radio-telemetry transmission time yet still record sufficient data for use in various studies.

For a winter storm, the 30 minute average closely represents the 5 minute samples (Figure 3). However, some small changes are omitted. During higher-intensity summer rainfall events, when soil moisture is changing rapidly, it is likely that these discrepancies will be greater. Although a 30 min average soil moisture is probably acceptable for use with many longer time frame analyses (e.g., weeks to years), to maximize the usefulness of data for analyses at an event scale, it is desirable to log soil moisture at 5 min intervals. Because each probe is associated with a precipitation gage, and data is logged with a programmable data logger, the sampling interval may be varied based on rainfall patterns. This is desirable to reduce the amount of extraneous data, thereby minimizing the amount of storage space required, both in the data logger and database.

Poor correlation was found between maximum rainfall intensity and response time, or rainfall volume and response time (r^2 values of 0.214 and 0.370, respectively). However, response time could only be identified to the nearest 30 minute interval during the initial data collection phase. This supports the findings of Amer et al. (2000) that show an electrical resistance sensor in the same watershed had no correlation between rainfall intensity or volume and response time.

A general correlation between rainfall duration and response time was seen ($r^2 = 0.668$), but not sufficient to be an effective indicator of optimal sampling rate.

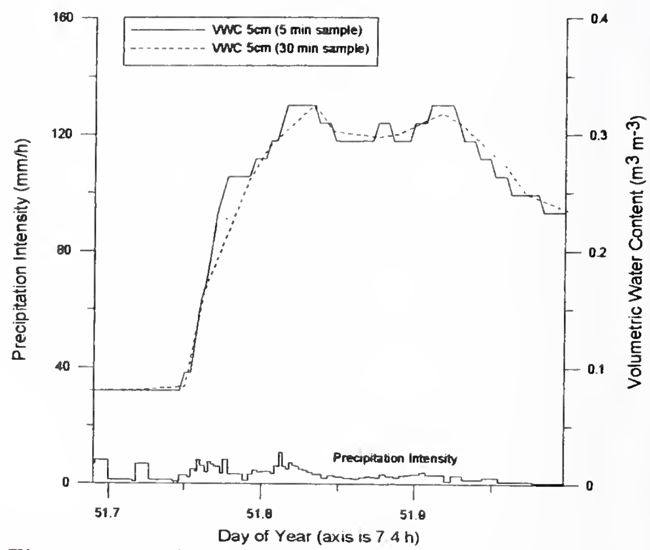

Figure 3. VWC vs. time comparing 5 minute sample and 30 minute average at RG 83.

Figure 4 shows typical precipitation and probe response over a ten day period during the summer monsoon season at RG 83. Periods of rainfall are shown by the shaded regions. Rapid changes in soil moisture occur during periods of rainfall at 5 cm and 15 cm. Changes in soil moisture during periods of no rain are more gradual.

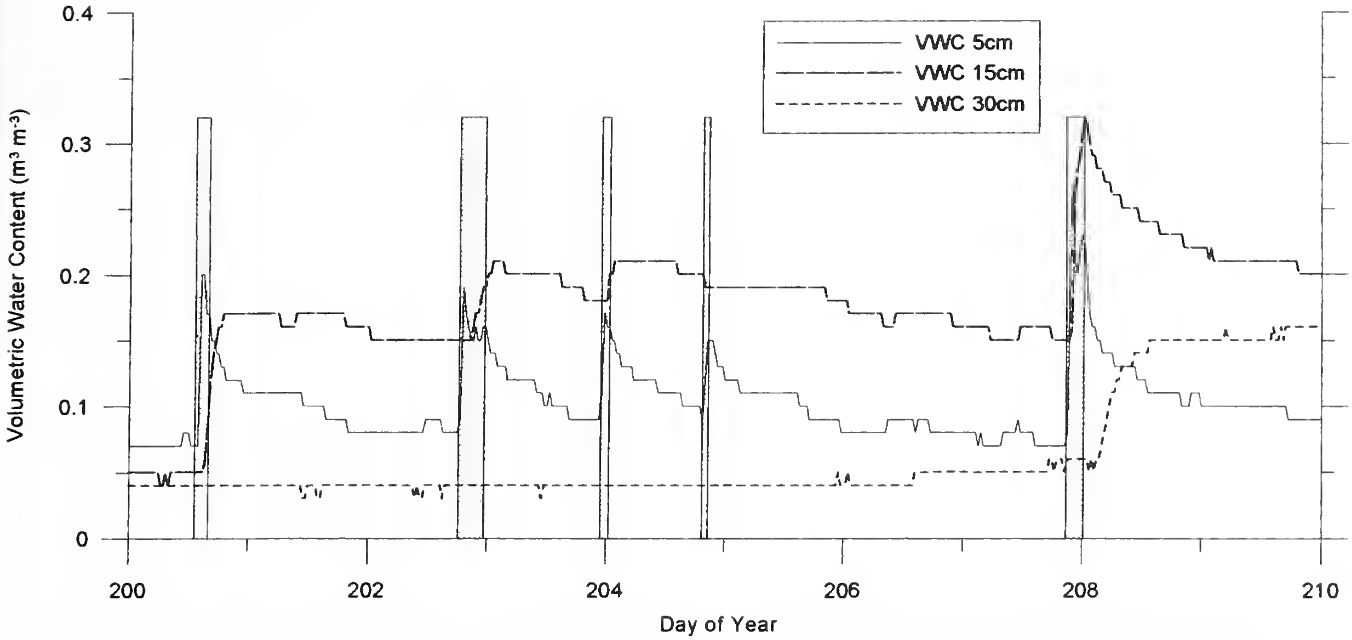

Figure 4. Vitel VWC during the summer monsoon season at RG 83. Shaded regions are precipitation events.

Therefore, it is reasonable to sample these probes at a 5 minute interval during precipitation events, and at longer intervals, such as 30 minutes, during periods of no precipitation. The response at 30 cm is more gradual, and based on the monsoon events of 2002, little or no gain would be realized by sampling more frequently. If data storage space is an issue, it would be possible to sample the 15 cm and 30 cm probes at longer intervals, upwards of two hours, during dry periods. In extreme cases, such as during spring and fall when the watershed may go weeks at a time with no rain, the probes could be sampled daily or even longer.

Soil water redistribution

Figure 5 shows daily soil water content values and cumulative event precipitation during the summer monsoon period of 2002 at RG 83. Because of the correction for minimum dry conditions (see Methods), DOY 189 shows the minimum water volume possible in the profile, at the end of the spring dry season. Following the onset of the summer monsoon, water content remained elevated throughout the year, until the end of the following spring (data not shown).

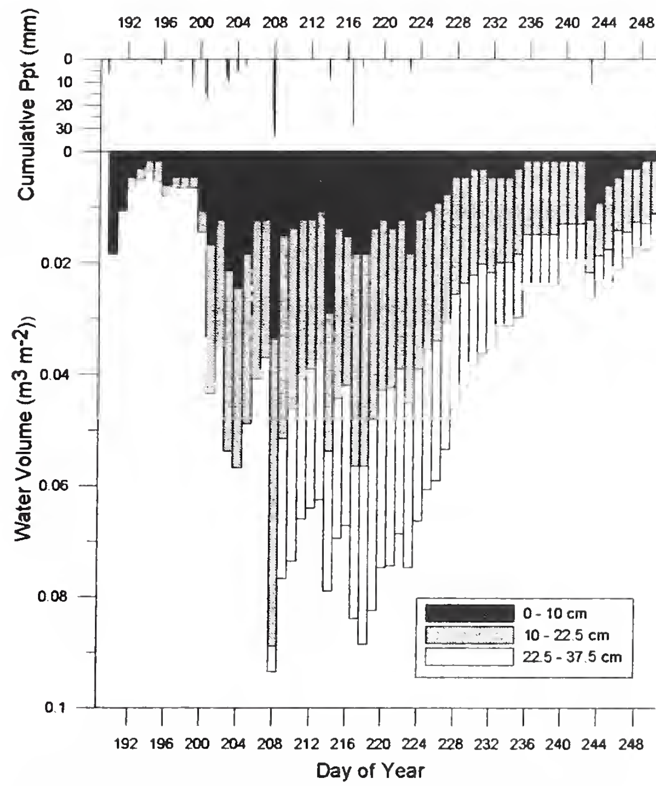

Figure 5. Soil water volume and precipitation at RG 83.

Dry down in the upper layer of the profile occurred much more quickly than in the lower two layers. The surface layer dries quickly and stays dry (less than 0.02% available water) following a rainfall event. Therefore, it is unlikely that matric potential is drawing significant water from deeper soil layers to the surface. Otherwise, a rise in water content at 5 cm would be seen. Water volume at 15 cm and 30 cm also decreases, but at a much slower rate than at 5 cm. At 30 cm, water volume increases and decreases gradually over the course of the monsoon. The largest increase occurs shortly after the largest rainfall event of the season, on DOY 208. However, antecedent rainfall was likely a contributing factor. Because infiltration beyond 30 cm is infrequent during the summer monsoon (Scott et al. 2000), it is presumed that water at the 30 cm depth is lost through root-uptake and transpiration or lateral infiltration. However, this question is still being evaluated. The results from Scott et al. (2000) are based on soil moisture measurements taken at approximately two week intervals. As can be seen in Figures 1, 4, and 5 many changes in soil moisture can potentially occur during a two week period.

Conclusions

In-situ dielectric constant-based soil moisture probes offer several advantages over other techniques for measuring soil water content, such as electrical resistance sensors, neutron probes, and gravimetric sampling. Most importantly, the probes allow near-continuous measurements to be made with a data logger, precluding the need for routine site visits. These probes are relatively low in cost compared with in-situ TDR systems, require minimal maintenance, and are easy to install.

Data collected during the first year since installation of the Vitel probes shows the probes are capable of quickly responding to changes in soil moisture, and with appropriate calibration and/or correction, accurately measure soil water content. After sampling the three profile sites at five minute intervals from February through May 2003, it is apparent that an abundance of extraneous data is being collected. Following the 2003 summer monsoon, data loggers at these sites will be re-programmed to respond to precipitation events, thereby minimizing data storage overhead while maintaining the ability to record small scale and rapid changes in soil moisture during and following precipitation events.

Acknowledgments

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An ARS Retiree Looks at USDA Water Resource Programs

Kenneth G. Renard

Abstract

ARS scientists and engineers have been developing natural resource models to assist action agencies with programs to manage and rectify environmental concerns at various time and spatial scales. Past efforts are reviewed and comments offered on future needs.

Keywords: natural resource models, water, watersheds, simulation, historical development

Introduction

The paper will be directed toward water initiatives needed in USDA but not including USFS programs because the author is not familiar with that work. As an 'outside observer,' a new emphasis to USDA water programs is long overdue. Although USDA was a strong participant (and past leader) in Federal water programs, this is no longer true (in my opinion). The reason for this change may include: 1) the populous is now urban and does not recognize that most water originates from agricultural and forest lands, 2) food is too plentiful to warrant additional funding, and 3) USDA leadership is production-oriented (food and fiber) with only minimal environmental concerns.

Historical Programs

PL 566 programs

Most past funding impetus came to USDA in response to problem needs. Certainly the small watershed programs in the 1930's and later associated with flood-control activities (funded under Public Law 566) on upland watersheds, set a strong precedent in USDA for conservation activity. The famous (or infamous) curve numbers produced by SCS personnel in 1957 for estimating flood peaks from rural watersheds is widely used even in the 21st century. This technology (curve numbers, design rainfall data and land uses) has been cussed/discussed/revised in the ensuing years but still remains a favorite design procedure for water problems and specifically flood peak estimation (Stewart et al. 1975, ASCE 1996).

Senate document '59

Following the creation of ARS in 1953-54, soil and water conservation research programs that had originated in SCS, were slow to expand. The US Senate Appropriations Committee in 1959 produced a report on needed research based on citizen input. The report stressed the urgent need to determine water (and soil) problems of regional and national need for protecting the nations natural resources. The hearings and input at them from a number of users, led to *new perspectives for USDA.*

The needs report stated that "Special attention should be given to hydrologic research on agricultural watersheds." Six regional watershed centers were identified as needed supplement original SCS watersheds and to quantify geographic and cultural differences in areas of the U.S. These six new centers funded over a few years and added to iriginal SCS watersheds have provided fundamental technology for hydrologic and erosion/ sedimentation problems. A national hydrology and an erosion and sedimentation laboratory complemented the regional centers.

Parallel to the aforementioned laboratories was the need for laboratories to research specific problems (e.g. Water Conservation Laboratory, etc.) which have helped the SCS/NRCS implement conservation programs with individual farmers and ranchers for the wide range of soil and water resource programs. Over the ensuing period, funding of these research locations has become a major concern as problems change and inflation erodes available resources.

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Specific collaborative efforts

With an established infrastructure to pursue research and provide needed data, ARS began a program of mathematical model development to assist SCS/now NRCS and others with program solutions for specific problems. The personnel assembled at ARS research locations represented differing scientific expertise to address specific problems. The earliest collaborative efforts addressed erosion problems associated with wind and water. The timeline for programs discussed subsequently is presented in Figure 1.

The USLE (Universal Soil Loss Equation) rcsulted from plot data sent to Indiana and analyzed by Wischmeier and Smith (1965). That technology was updated (Wischmeier and Smith 1978) and used by the SCS in numerous environmental assessments. The technology is essential to farm conservation plans involving water erosion and plans to reduce soil loss.

In somewhat of a parallel approach, the Wind Erosion Equation was published as a USDA Handbook by Skidmore and Woodruff (1968). The technology is widely used for wind erosion prediction and control. Although specific funding for both pieces of technology was part of continuing basic ARS funding, it soon became evident that such funding was insufficient to address emerging new concerns and that new monetary resources were needed.

Water Pollution (Stewart et al. 1975)

The EPA funded collaborative research with ARS to assess water pollution from cropland (Stewart et al. 1975). For this effort, ARS scientists were assigned to represent a disciplinc and diverse geographic location. The resulting reports were widely used to assess pollution from farmlands of the U.S. To my knowledge this effort was unique because it represented the first such federal interagency effort of this kind.

CREAMS/GLEAMS

The need for computer technology to assist with problem solutions led ARS to designate a team to assist with non-point pollution problems (Chemicals, Runoff, Erosion and Agricultural Management Systems). When the agency asked for volunteers in 1979, a large number of scientists responded. Computer software was developed to predict the hydrology-erosion-chemical losses from different land uses and physiographic regions of U.S. (Knisel et al.1980). This analytical model set a prescription for several computer efforts that followed. Again SCS, the technology user, adopted the model with ARS scientists and engineers to address specific USDA needs. Although specific **new resources** to support the effort were minimal, it encouraged ARS staff members for future such research. The model developed by Knisel

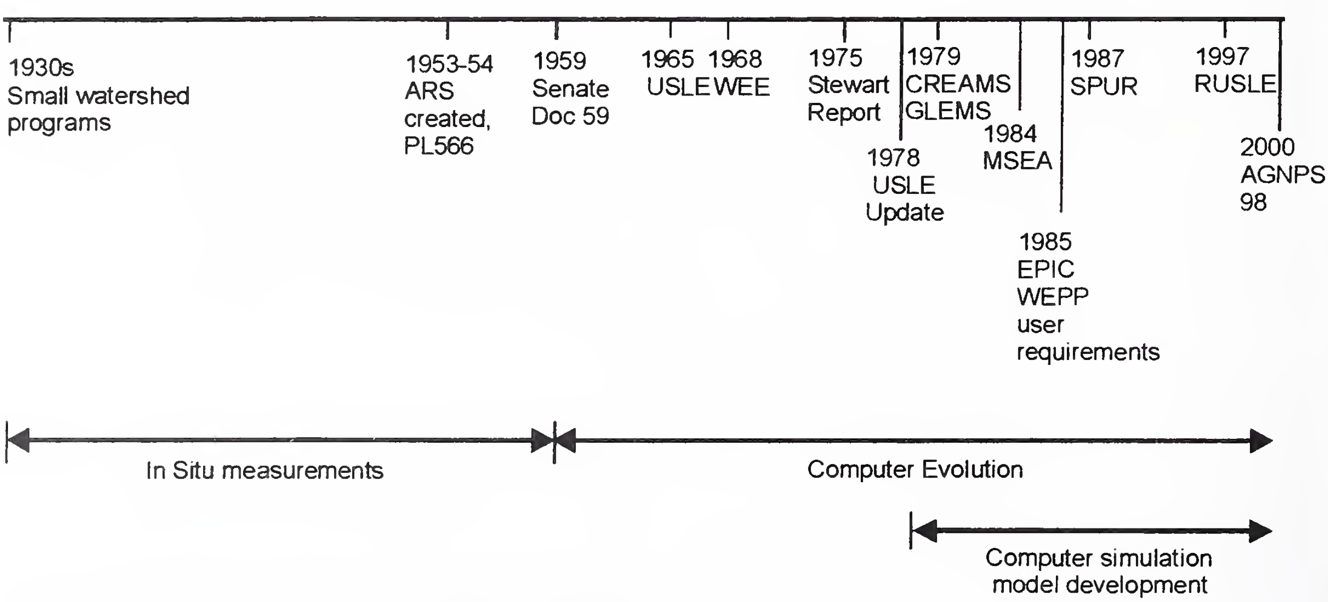

Figure 1. Timeline of simulation models and programs.

and associates generated interest among the international community.

EPIC (Erosion Productivity Impact Calculator)

Environmental concerns evolved in USDA in the early 1980s to predict/manage the effects of soil erosion on long-term production of food and fiber from U.S. land resources. Again in response to a SCS request for assistance, an ARS team was designated to provide analytical assistance for model development. Concurrent with the software model developed, field experiments were designed across the country to assist with quantifying the impacts of erosion on food and fiber production. SCS made many model simulations in the U.S. to quantify erosion impacts across agricultural production areas. The analytical model, called EPIC (Williams and Renard 1985) has been and continues to be used for soil-erosion resource assessment in the U.S.

SPUR

Building on the success of EPIC, ARS assembled a new team in 1986 to develop technology for Simulation of the Production and Utilization of Rangelands, and specifically grazing impacts on the hydrology-erosion-environmental conditions of the western U.S. (Wight 1987). Again some limited additional funds were available to assist with this effort. In contrast to other model efforts, this model was less restrictive area wise. Also important to this effort, was the inclusion of animal and range plant scientists who broadened the model to include animal conditions.

WEPP-WERM-RUSLE

In a 1985 USDA workshop in Indiana, future research needs in water and wind erosion were identified. Workshop participants agreed that an effort was needed to upgrade the science associated with the USLE (Wischmeier and Smith 1978) and the wind erosion prediction equation (Skidmore and Woodruff 1968). Following the workshop, plans evolved to upgrade soil erosion technology to include recent research findings and to produce digital computer technology identified as RUSLE (Renard et al. 1997). The plans also cited the need to produce technology to ultimately replace USLE (Laflen et al. 1981, Lane and Nearing eds1989). This new model which was designated WEPP (Water Erosion Prediction Project) has been delivered to NRCS (they were participants throughout the computer code development). The model receives modest use because of the large data input needs (not always available).

The initial effort to develop RUSLE was completed with the publication of Agriculture Handbook 703 (Renard et al. 1997). The implementation of the computer code in the Windows environment is nearing completion. Because of retirements, assistance for this research continues through the University of Tennessee. NRCS uses this technology in soil conservation programs throughout the United States.

The wind erosion prediction effort, known originally by initials WERM (Wind Erosion Research Model) is now designated WEPS (Wind Erosion Prediction System) (Hagen 1991) and parallels the RUSLE-WEPP activity. Mention of this work here is included because although wind is the driving force, water is an important factor. Furthermore, ARS and NRCS personnel often work with both wind and water.

New Activities and Funding Efforts

Several USDA staff members concerned with water problems have been integrating GIS technology (El-Swaify and Yakowitz 1998). These new technologies have developed such that they can be integral to continuing natural resource analytical programs. Again these new analytical techniques have been developed from existing budgets.

Water quality programs

The need to improve and conserve water resources was recognized with the Presidential Initiative on Water Quality in 1989. The initiative had the objectives of 1) protecting ground water resources from contamination; 2) developing water quality to address contamination; and 3) providing basic information to alter practices and contamination (Bush 1989). USDA responded to the Initiative by establishing interagency research and assessment with the title Management Systems Evaluation Areas (e.g. Onstad et al. 1991).

As part of the initiative, the Soil and Water Assessment Tool (SWAT) model (Arnold et al. 1995) and the Annualized Agricultural Non-Point Source (AnnAgNPS) model (Bingner and Theurer 2001) have become primary tools for planning watershed approaches for agricultural management practices. ARS recently released the second version of the Root Zone Water Quality Model (RZWQM) for determining the interaction between practices, hydrology, crop growth, and chemical fate (Ahuja et al. 1999).

Climate change programs

Climate change research has become a major activity in Federal Agencies. The amount of funding available for ARS water research continues to deteriorate. Some progress has been made to define CO_2 and temperature changes but with only minimal correlation to precipitation or water cycle changes.

Global water cycle

The USGCRP (Global Change Research Program) appointed a Water Cycle Study Group in 1999 (Hornberger et al. 2001). They identified needed water programs at the federal level. USDA bought into this program which as a blueprint for water-cycle activities in the next decade. Necessary ingredients to the initiative include 1) improved observations and measurements; 2) coordinated field, remote sensing, and modeling experiments; and 3) spatially nested regional climate models to link atmospheric, land surface, and subsurface processes. The outlook for the future appears to be strong, with the question about USDA's role. Is it likely that USDA will regain some stature in water programs?

As an outsider, I wonder whether USDA and other Federal Agencies will collaborate with Universities and Professional Societies, and State and Private groups with international connections (e.g. World Bank, UN's FAO).

Technical Considerations

USDA may need to address additional items such as those that follow. These technical issues deserve consideration as we seek to build scientific knowledge from past foundations (Helms et al. 2002).

1930-1950s: Period of field experiments

Field measurements dominated water programs during the period following the dust bowl. During this time, experimental programs were designed that produced data sets being added to, and still used today. Also, USDA had major input to water programs. SCS developed many programs in response to zealous conservationists like H.H. Bennett (a review of Bennett's work was presented at the 2001 SWCS meeting in South Carolina and excerpted in WASWC (World Association of Soil and Water Conservation Newsletter (V17:N4 dated October 2001). A new proponent with USDA ties is needed to return to earlier significance.

1960-1980s: Computer evolution

The advent of modern computers since WWII generated expanded power for data analysis. The computer development changed the way water research is performed. Increased computer speed, personal computer development, and advances in computing technology have major repercussions on water programs. Although advances have been positive, some needs remain.

1990s: Scientific expertise

Computer development continues at high levels today. The options they provide to monitor and manage water resources is significant. A problem involving student training and rewards for professional water scientists/engineers need discussion.

Students from undergraduate and graduate programs have become *computer jockeys* but they often have scant ideas on organizing and planning the collection of *in-situ* knowledge with which to populate analytical models. Often they do not know if an answer is reasonable but rather want to believe computer calculations. It is easy for me to imagine how our *in-situ* measurement technology could be enhanced. Data collection needs to take advantage of current electronic technology. In many instances, water data is being collected using equipment developed in past decades (not considering space-age technology).

The experience of serving on ARS Peer Review panels where measurements were not recongnized in individual accomplishments is disturbing. This attidude was and is restrictive to scientific progress. How can there be reliable modeling without reliable data? Academicians have similar problems. This *archaic* attitude hinders our ability to measure a system response that we now can simulate so easily, leaves major questions unanswered. How long can this continue before there is litigation resulting from unverified model applications?

Opportunities in hydrologic sciences

A blue-ribbon committee of the National Academy of Sciences (Eagleson 1991) produced a document that enumerated needs and opportunities in the hydrologic sciences. The report advocates hydrology as a distinct geoscience interactive on a range of spatial and temporal scales. Although the report has been accepted at the scientific level (in my opinion), funding and problem solving has progressed slowly because of institutional constraints. Water scientists/engineers continue to disagree over turf battles associated with hydrology (engineering, geosciences, or elsewhere)? What federal agency should lead this effort? Is it necessary to have a lead group?

Consortium of Universities for Advances in Hydrological Sciences, Inc. (CUAHSI)

The consortium was recently incorporated in the District of Columbia in 2001 (www.cuahsi.org). CUAHSI is attempting to build consensus within the university hydrologic research community for future research priorities that will be developed through the National Science Foundation (NSF). Based on National Research Council (NRC) reports, CUAHSI is proposing development of Long-Term Hydrologic Observatories (LTHOs) through major infrastructure proposals to NSF. These in parallel with NSF Long-Term Ecological Research (LTER) site concept will form the basis of the university experimental watershed program. Another key infrastructure aspect that CUAHSI is proposing is development of new measurement technology for hydrology. The LTHO's are large experimental watersheds having drainage areas between 1,000 and 100,000 km^2. The intent of CUASHI is not to replicate work of ARS, USFS, or USGS in catchments less than 1,000 km^2, but to address problems with temporal and spatial variability over large areas. It is still unclear how CUAHSI will coordinate or work with Federal hydrologic research agencies to build on existing Federal experimental watersheds or enhance Federal watershed research to include variables not currently observed in the watersheds.

Summary/Conclusions

Future progress will necessitate changes in funding duration (need longer project funding). Current funding (in academia) is generally for a few years and precludes sampling climatic extremes that are known to exist. Most water resource problems are very non linear requiring data sampling for extended periods.

We need to fully utilize computers, and scientific concepts may well be best accumulated and expressed via computer models, but first we must have well-designed field research. If we are at the proper stage, maybe it is better to say we need a good feedback loop between phenomenological research and computer analysis somewhat as astronomists do.

Water-cycle measurements historically have been made by scientists/engineers with little knowledge of modern data acquisition and storage technology. Furthermore, rewards for personnel performing such work was often treated suspiciously. Grants for such water research must include developing instrumentation. Prediction technology cannot improve without improved measurement and understanding of hydrologic processes. Personnel with advanced training in electronics/physics need to be part of the project team from its inception.

A system must be in place to recognize and reward people performing innovative efforts to improve measurement in water programs. Water activities (in USDA and elsewhere) need input from disciplines not heretofore part of the team addressing temporal and spatial variability in the water cycle. Finally, an emphasis is needed in academia to produce students with training in measurement principals and data accuracy.

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Modeling Hydrologic Variables and Terrain Features for Strategically Locating Riparian Buffers

Michael Burkart, Mark D. Tomer, David E. James, Thomas M. Isenhart

Abstract

Vegetated riparian buffers and constructed wetlands can reduce the amount of sediment, nutrients, and pesticides entering streams if they are located to intercept water moving from agricultural land toward streams. Hydrologic factors control processes that affect water quality making it critical to consider these factors in the design and location of riparian buffers if water quality benefits are to be realized. Three areas with differing terrain and hydrology are being used to develop methods to locate riparian areas with the greatest potential to affect water quality. A 30-meter DEM was used to model the distribution of hydrologic and landscape features to identify where runoff, infiltration, and groundwater flow can be influenced by riparian management. In a loess-dominated watershed with natural drainage, steep slopes and significant erosion, riparian buffers located along first order streams would intercept the largest portion of the water flowing to the stream, thus have a greater effect on water quality. In a third-order, clay-till dominated watershed with extensive artificial drainage, areas best suited for buffers occupy generally less than 300 m of stream reach and are widely distributed throughout the basin. In a third basin, several sites meeting criteria for constructed wetlands have been mapped using Hydrologic and terrain features. These findings will be useful to support the application of conservation practices to reduce nutrient contamination of streams over broad areas.

Keywords: terrain analysis, riparian, buffers, wetlands

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Hydrology IV (Remote Sensing and GIS)

Estimating Regional Daytime Net Carbon Dioxide Flux using Remotely Sensed Instantaneous Measurements

Chandra D. Holifield, William E. Emmerich, M. Susan Moran, Ross Bryant, Charmaine L. Verdugo

Abstract

Atmospheric carbon dioxide (CO_2) is steadily increasing as a result of the world's increasing use of fossil fuels and wood biomass. However, the impact of increases in CO_2 on the global carbon cycle is unclear. Semiarid grasslands comprise a large portion of the world's rangeland ecosystem and may play a significant role in the carbon cycle. In a previous study, regional estimates of instantaneous net CO_2 flux were obtained by using a Water Deficit Index (WDI) derived from satellite imagery over a five-year period (1996-2000) covering a grassland site in the Walnut Gulch Experimental Watershed (WGEW). In this study, a linear relationship (R^2 = 0.95) was found to exist between instantaneous and daytime net CO_2 flux estimates, where daytime is the period from 6 a.m. to 6 p.m. This linear relationship was used to convert instantaneous estimates of net CO_2 flux to daytime estimates, and maps depicting spatially distributed daytime net CO_2 flux were generated for WGEW. Remote sensing offers a viable means of obtaining regional estimates of daytime net CO_2 flux in semiarid grasslands.

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Keywords: carbon dioxide (CO_2) flux, semiarid grasslands, remote sensing, Water Deficit Index (WDI)

Introduction

A large portion of the earth's surface contains semiarid grasslands. However, the role these grasslands play in the carbon cycle is unclear. Studies are being conducted around the world in an effort to answer this question. Unfortunately, many of these studies are being conducted on a small scale (a few hundred square meters), when larger, regional scale measurements are most important for examining the global carbon cycle. With its potential for larger scale and global coverage, remote sensing could be a useful tool in determining the role played by semiarid grasslands in the global carbon cycle.

A remote sensing derived measurement of plant transpiration, the Water Deficit Index (WDI), has been used to estimate instantaneous net CO_2 flux on a regional scale (Holifield et al. 2003). Further, studies have shown that daily plant evapotranspiration (ET) can be estimated from an instantaneous ET measurement using the direct relationship that exists between plant ET and incoming solar radiation (R_s) (Jackson 1983, Zhang and Lemeur 1995). Given that CO_2 uptake by plants is tied to transpiration, it follows that an estimation of daytime (6 a.m. to 6 p.m.) net CO_2 flux can be obtained from a measure of instantaneous net CO_2 flux. However, in light of the fact that the aforementioned studies were conducted in fully irrigated environments and semiarid grassland systems are seldom irrigated, a different approach must be employed. Thus, the objectives of this study were to convert instantaneous net CO_2 flux estimates from a semiarid grassland site to daytime estimates; and to generate maps depicting spatially distributed estimates of grassland daytime net CO_2 flux.

Methods

The following subsections include a brief description of the study area and instrumentation, the criteria used in data selection for the conversion of instantaneous net CO_2 flux to daytime estimates, and the procedure used to generate spatially distributed maps of daytime net CO_2 flux.

Study area and instrumentation

The study took place in the Walnut Gulch Experimental Watershed located in southeast Arizona. The study area was comprised of a grassland area of approximately nine square kilometers, dominated by black grama (*Bouteloua eriopoda*), sideoats grama (*B. curtipendula*), blue grama (*B. gracilis*), Lehmann lovegrass (*Eragrostis lehmanniana*), and bush muhly (*Muhlenbergia porteri*). Continuous 20 minute averages of carbon dioxide and water vapor flux measurements were collected by a Bowen ratio energy balance (BREB) system located within the study area (Emmerich 2003).

Data selection criteria

Data covering the monsoon season (1 July to 31 October) over a five-year period (1996-2000) were used for this study. All data were selected based on an incoming solar radiation (R_s) curve to indicate clear days. Days showing clear morning and cloudy afternoon conditions were also selected. Days that had an occurrence of precipitation were eliminated.

For each day, the 11 a.m. measurement was selected for use as the instantaneous CO_2 flux measurement due to its concurrence with Landsat Thematic Mapper (TM) overpasses. The flux measurements from 6 a.m. to 6 p.m. were summed and used as the actual daytime net CO_2 flux for each day.

Calculation of daytime net CO_2 flux

A two-year (1996-1997) data set consisting of 41 days was used to examine the relationship between instantaneous and daytime net CO_2 flux. A three-year data set (1998-2000) composed of 69 days was used for validation of the relation. The resulting equation from that comparison was applied to Landsat imagery to generate spatial maps of daytime net CO_2 flux.

Results and Discussion

Semiarid grassland CO_2 fluxes tended to deviate from the expected relation that exists between plant ET and R_s. Figure 1a illustrates an R_s curve over the course of a cloud-free day. Plant ET would have a very similar curve under cloud-free conditions with unlimited water availability (Jackson 1983). Due to its link with transpiration, CO_2 plant uptake was expected to have the same curve. However, in water limited conditions this was not the case. Figure 1b shows the variability that occurred due to differences in water availability. The trend in CO_2 flux observed in the morning proved to be an indicator of the flux trend for the day.

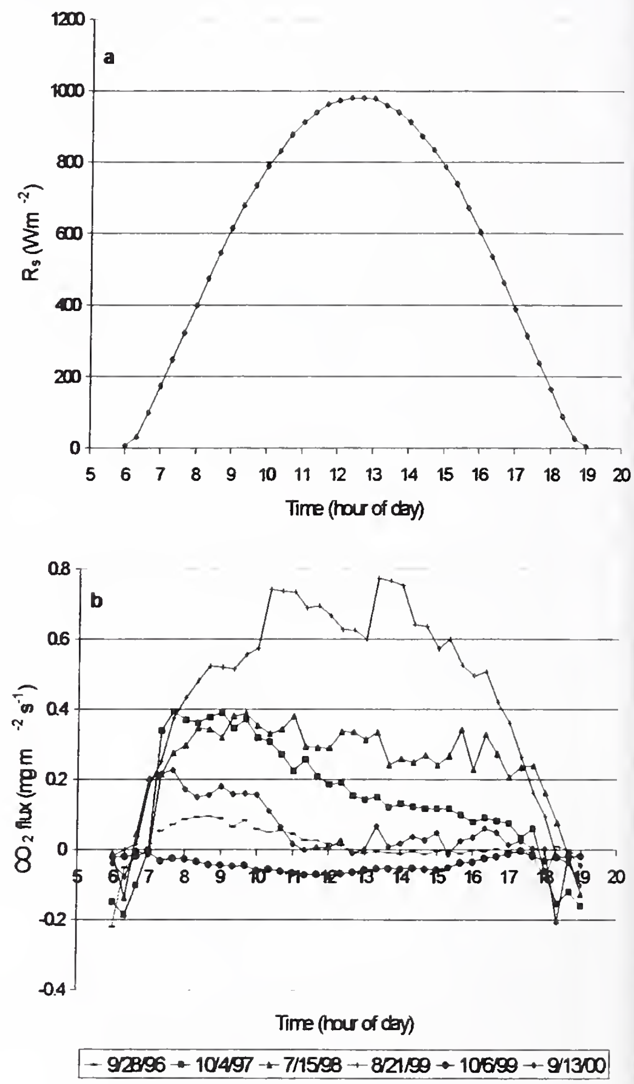

Figure 1. (a) Illustration of incoming solar radiation (R_s) from 6 a.m. to 6 p.m. under cloud-free

conditions. (b) Measurements of net CO_2 flux for cloud-free days with varying plant water availability. The relation between daytime and instantaneous net CO_2 flux (Figure 2) indicated a strong ($R^2 = 0.95$) linear relationship. The resulting equation was validated using data from 1998-2000, where

$$CO_{2d} = 29.338(CO_{2i}) - 0.2461 \quad (1)$$

and CO_{2d} is daytime net CO_2 flux (g m^{-2} (12 hrs)$^{-1}$) and CO_{2i} is instantaneous net CO_2 flux (mg m^{-2} s^{-1}) (Figure 3). The RMSE for negative flux values, defined here as a net loss from the ecosystem, and positive flux values were 0.37 and 1.44 respectively.

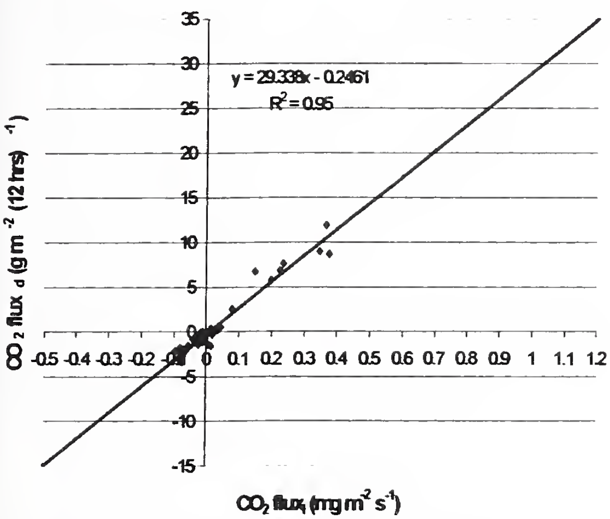

Figure 2. Comparison of daytime and 11 a.m. instantaneous net CO_2 flux for 1996 and 1997 on clear days. Negative flux values indicate net CO_2 loss from the soil. Positive values indicate net CO_2 uptake by plants.

Analysis was done to determine what happens to net CO_2 flux when a day has a clear morning, but clouds appear in the afternoon (Figure 4). This scenario is more typical of atmospheric conditions during the monsoon growing season. The relationship remained linear when the flux was negative. This was apparently due to the fact that soil moisture is the limiting factor when negative net CO_2 flux occurs. As a result, it made no difference whether sky conditions were clear or cloudy because the water limitation dominated. However, when the flux was high and positive, the relationship became slightly curvilinear. When flux is positive, water is generally not a limiting factor. In these instances, R_s becomes the limiting factor. Thus, when cloudy conditions occur, net CO_2 flux begins to decrease due to the fact that plants are forced to limit photosynthesis, decreasing CO_2 plant uptake.

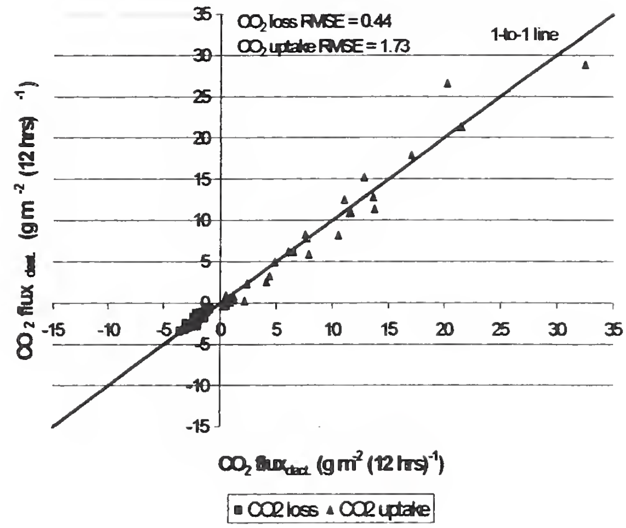

Figure 3. Comparison of estimated and actual daytime net CO_2 flux. Data from clear days in 1998, 1999, and 2000 were used for validation of the relation between daytime and instantaneous flux. Negative and positive flux values indicate net CO_2 loss and uptake, respectively.

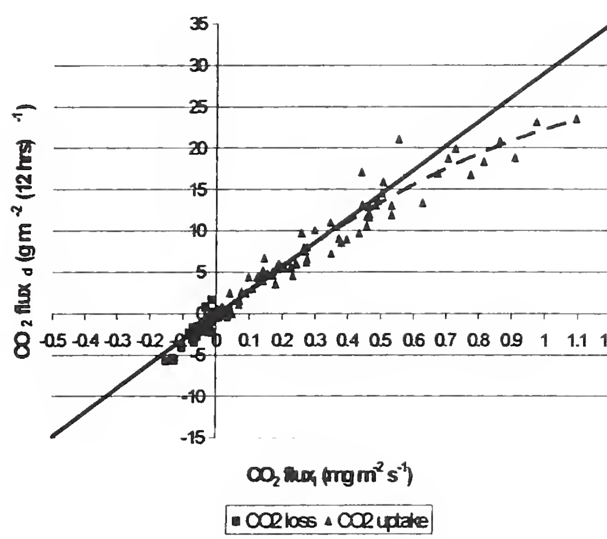

Figure 4. Comparison of daytime and 11 a.m. instantaneous net CO_2 flux for days with clear mornings and cloudy afternoons. Negative and positive flux values indicate net CO_2 loss and uptake, respectively.

Consequently, when Equation 1 was applied using the instantaneous flux values for these days, the result was an overestimation of daytime net CO_2 flux

(Figure 5), with RMSE of 0.70 and 2.00 respectively for negative and positive flux.

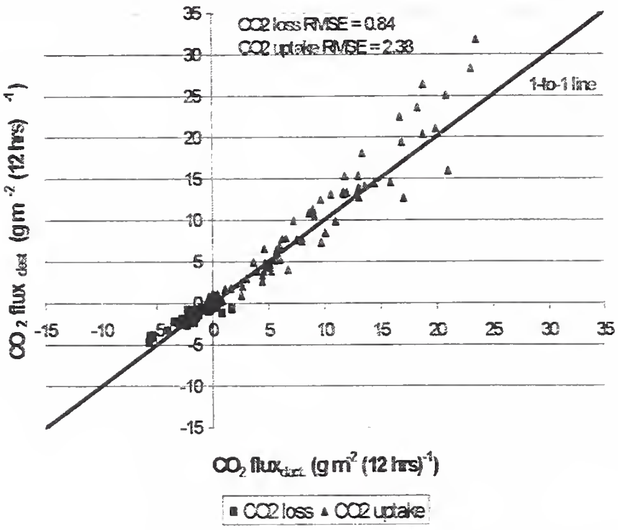

Figure 5. Comparison of estimated and actual daytime net CO_2 flux for days with clear mornings and cloudy afternoons. Negative and positive flux values indicate net CO_2 loss and uptake, respectively.

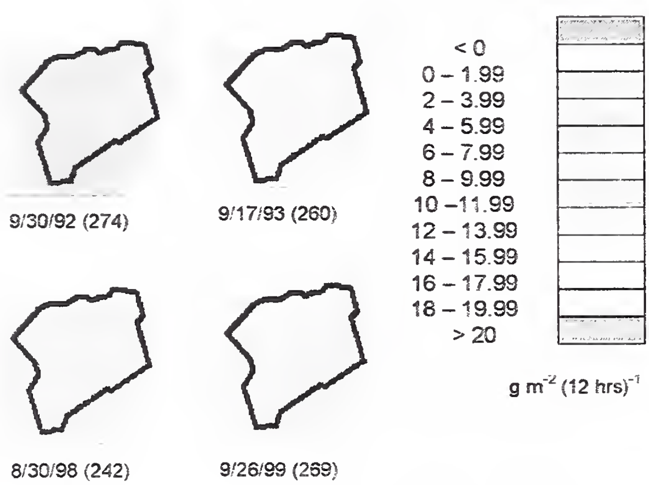

Figure 6. Landsat images of daytime net CO_2 flux (g m^{-2} (12 hrs)$^{-1}$) over a 9 km^2 area for 9/30/92 (DOY 274), 9/17/93 (DOY 260), 8/30/98 (DOY 242), and 9/26/99 (DOY 269). Negative values indicate net CO_2 loss from the soil. Positive values indicate net CO_2 uptake by plants.

Equation 1 was applied to Landsat satellite images of instantaneous net CO_2 flux created using a relation developed by Holifield et al. (2003) between net CO_2 flux and the WDI to generate spatially distributed maps of daytime net CO_2 flux (Figure 6). Each map reflected the impact made by precipitation or lack thereof, and the resulting plant responses. The DOY 274 image was taken during an extremely dry time period and the result was an overall negative flux measurement for the day. The DOY 242 image illustrated the spatial differences that existed due to a somewhat localized storm that had passed through a few days before the image was taken. Higher values of CO_2 uptake by plants were seen along the left portion of the image. This coincided with the storm path. The DOY 269 image reflected the fact that 1999 was a very wet year. Soil moisture was adequate to support high levels of transpiration and thus, high levels of CO_2 plant uptake (10 to 20 g m^{-2} (12 hrs)$^{-1}$).

Conclusions

In this study, instantaneous net CO_2 flux measurements were converted to spatially distributed, large-scale estimates of daytime net CO_2 flux. This conversion was possible because the 11 a.m. instantaneous flux measurement was found to be indicative of daytime flux in environments where water is a limiting factor. Consequently, a linear relationship was found to exist between instantaneous and daytime net CO_2 flux in semiarid grasslands. However, on days when water is not limiting and R_s is limited by cloud cover, overestimation of daytime flux is likely to occur. Nonetheless, this study showed it was possible to obtain reasonable large-scale, spatially distributed estimates of daytime net CO_2 flux for semiarid grasslands and serves to illustrate the promise of this potential tool in determining the role played by semiarid grasslands in the carbon cycle.

Future work could be focused on combining these daytime net CO_2 flux measurements with modeled or measured nighttime net CO_2 flux estimates to determine total daily net CO_2 flux. The ultimate goal would be to determine the seasonal patterns of daily net CO_2 flux and annual net CO_2 flux in semiarid grasslands.

Acknowledgments

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Inferring Root Zone Soil Water Content by Assimilating Remotely Sensed Data Into A Soil Water Model

Patrick J. Starks, Thomas J. Jackson

Abstract

Increased demand for available water supplies necessitates that tools and techniques be developed to quantify soil water reserves over large land areas as an aid in management of water resources and watersheds. Microwave remote sensing can provide measurements of volumetric water content of the soil surface up to about 10 cm deep. The objective of this study was to examine the feasibility of inferring the volumetric water content of the root zone by combining remotely sensed estimates of surface soil water content and modeling techniques. A simple soil water budget model was modified to estimate root zone soil water content from remotely sensed estimates of surface soil water content. Two modeling scenarios were evaluated at four tallgrass prairie sites located in central and south central Oklahoma: 1) model simulation without assimilation of remotely sensed estimates of soil water content, and 2) model simulations with assimilation of soil surface water content estimated from remote sensing. The unmodified model (scenario 1) underestimated measurements with root mean square errors (RMSE) between 0.03 and 0.06 m^3m^{-3} and mean errors (ME) between 0.02 and 0.04 m^3m^{-3}. Simulations from scenario 2 agreed well with measured data at two study sites (0.00 $m^3m^{-3} \geq$ ME $\leq 0.02m^3m^{-3}$, RMSE $\leq$ 0.03 m^3m^{-3}) but underestimated measurements at the remaining sites, in one case by as much as 0.15 m^3m^{-3}. The underestimation was due largely to inaccurate remotely sensed soil surface water content values. These preliminary results suggest that it is feasible to infer root zone soil water content in tallgrass prairies by assimilating remotely sensed estimates of surface soil water into soil water models, provided that the remotely sensed data correctly estimates surface conditions.

Keywords: water budget, microwave, soil profile

Introduction

Soil water accounts for only about 0.0001% of the total water on earth, but its status in the root zone is a key parameter in many aspects of agricultural, hydrological, and meteorological applications. In agriculture, accurate knowledge of soil water content is essential for proper water resource management, irrigation scheduling, crop production, and chemical monitoring. Meteorologically, soil water content plays a significant role in the partitioning of available energy at the earth's surface into heating the air and that used in evapotranspiration. In hydrology, soil water partitions rainfall into infiltration or runoff.

Increased demand for available water supplies coupled with the vagaries and variabilities of climate, necessitate that tools and techniques be developed to quantify soil water resources over large, and often spatially variable, land areas as an aid in management of water resources and watersheds. Point-based, direct measurement methods are either impractical or too expensive for large land area applications. Microwave remote sensing is a technique that offers potential for providing frequent measurements of soil water content over large land areas in a timely and cost-effective manner. However, these measurements only represent the soil surface down to about 10 cm deep, depending upon sensor type and wavelength used (Engman and Chauhan 1995).

In this paper, microwave surface measurements of soil water content are assimilated (input) into a simple soil water budget model to determine the feasibility of estimating soil water content down to about 60 cm. Study sites from the USDA-ARS' Little Washita River Experimental Watershed (LWREW), located in

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southwestern Oklahoma, are used to demonstrate the potential of using remotely sensed data to estimate soil water reserves over large land areas.

Methods

Site description

The LWREW (Fig. 1) is about 610 km^2 (236 mi^2) in size and is climatologically described as subhumid to semi-arid with total annual precipitation of about 75 cm (30 in). The topography is gently rolling and the land use is about 60% pastureland, 20% cropland, and 20% miscellaneous (forests, riparian areas, water and urban areas).

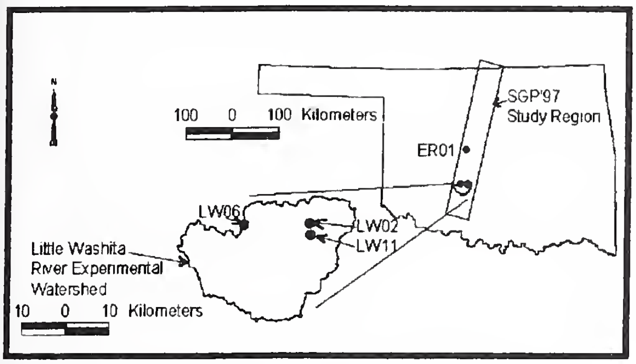

Figure 1. Location of the LWREW and study sites.

There are 64 defined soil series, with fine sand, loamy fine sand, loam and silty loam being the predominant textures of the soil surface (Allen and Naney 1991). The LWREW has a network of 45 meteorological measurement stations, collectively called the Micronet, distributed on a 5 km (3 mi) grid spacing. Each Micronet station measures rainfall, relative humidity, air temperature, incoming solar radiation, and soil temperature at four depths. These data are measured every 5 min and reported every 15 min to a central archiving facility. Co-located at 13 of these sites is a Soil Heat and Water Measurement Station (SHAWMS). Each SHAWMS measures soil water matric potential at 5, 10, 15, 20, 25 and 60 cm, as well as soil temperature at 2.5, 5, 10, 15, 20, 25, 60 and 100 cm, and soil heat flux at 5, 25, and 60 cm. A profiling time domain reflectometer (TDR) waveguide is also located at each SHAWMS.

Three Micronet/SHAWMS sites (LW02, LW06 and LW11) were selected for this study. An additional site (ER01), located on the grounds of the USDA ARS Grazinglands Research Laboratory, El Reno, Oklahoma, was also selected for study to provide vegetation and soil conditions not represented by the other sites (Tables 1 and 2).

Remotely sensed data

Because of its historical data bases and the presence of the Micronet, the LWREW became a primary study site for a large, multi-agency hydrology experiment known as the Southern Great Plains Hydrology Experiment 1997 (SGP97). The SGP97 experiment is described further in Jackson et al. (1998). The experiment was conducted from June 18 to July 17, during which time the electronically scanned thinned array radiometer (ESTAR) was flown to provide microwave-based estimates of surface soil moisture at a spatial resolution of about 1 km (0.6 mi.). Due to weather, instrument, and logistic constraints, the ESTAR was only flown on 10 days out of the 30 day experimental period.

Model

The model chosen for this study was developed by Ragab (1995). This model is a simple two-layer soil water budget that simulates the one-dimensional vertical movement of water in the surface (0-5 cm) and the root zone (in this study, the 0-60 cm) layers. The model operates on a daily time step and the required meteorological data are daily values of rainfall and potential evapotranspiration (ET_P). Rainfall was obtained from the Micronet stations and ET_P was calculated using the Penman-Monteith equation (Rosenberg et al. 1983). Initial water contents needed by the model were based upon measured data and empirical relationships derived between the surface layer and the root zone. Other required soil parameters required by the model are given in Starks and Jackson (2002).

The model was modified to run as originally written until a remotely sensed value of surface soil water content becomes available. At this point, the model replaces the original calculated surface value with the remotely sensed value and then proceeds as normal. Thus, the surface layer is "updated" with remotely sensed data at the frequency of availability, and the root zone calculations are based upon the new surface information.

Table 1. Leaf area index (LAI) and biomass measurements for the study sites. Data taken from Hollinger and Daughtry (1999).

Site	LAI	Green Standing Biomass			Brown Standing Biomass			Surface Residue		
		Wet	Dry	Water Content	Wet	Dry	Water Content	Wet	Dry	Water Content
		---(gm^{-2})---		%	---(gm^{-2})---		%	---(gm^{-2})---		%
ER01	4.7	1403	460	67	133	97	26	967	510	47
LW02	2.2	350	161	53	184	158	19	160	141	14
LW06	0.9	112	41	62	22	18	17	18	12	10
LW11	3.6	940	246	73	67	44	43	494	319	35

Table 2. Soil particle fractions and texture of the profile for each site.

Site	Sand	Silt	Clay	Texture
	--------------%--------------			
ER01	22	60	18	Silt loam
LW02	26	48	26	Loam
LW06	73	17	10	Sandy loam
LW11	54	24	22	Sandy clay loam

The model was run for two scenarios. The first scenario examines the model's ability to simulate the root zone soil water content for the meteorologic, soil, and vegetation conditions at each study site. In the second scenario, ESTAR surface (0-5cm) soil water content estimates are assimilated into the model, at the frequency of availability, to determine if model output is improved over that of the original simulations. Model output is compared to soil water content values acquired from the SHAWMS and/or TDR at the study sites.

Statistical analysis

Wilmott's (1982) d-index, root mean square error (RMSE), the coefficient of determination (r^2), and mean error (ME) are used to evaluate the model simulations. The d-index is a measure of correspondence between model output and measured data. A d = 1 means complete agreement between measured and modeled values, while a d = 0 means complete disagreement.

Results

Scenario 1 - Original model

The range of measured root zone soil water content over the course of the study period was about 0.04 m^3m^{-3} at sites ER01 and LW02, 0.08 m^3m^{-3} at LW06, and 0.14 m^3m^{-3} at LW11. These ranges represent 50, 20, 93, and 61% of the total plant available water (defined as the difference in water content at field capacity and wilting point) at these sites, respectively. Time series simulations from the original model exhibit the general patterns portrayed by the measured data, but the model consistently underestimated measured values at all sites (Figs. 2a-2d). The differences between measured and modeled root zone soil water content generally increase with time at sites ER01 and LW02, while at sites LW06 and LW11 there appears to be a constant offset or bias in the model simulations (Figs. 2a-2d).

The r^2 values indicate that the variation in the modeled values is strongly associated with the variation in the measurements at all sites (Table 3). The d-index (Table 3), however, indicates weak agreement between measured and modeled values at ER01, moderate agreement at sites LW02 and LW06, and stronger agreement at LW11. The ME reveals that the model underestimated measured values from 0.02 m^3m^{-3} at site LW02 to 0.05 m^3m^{-3} at site ER01. Only site LW02 had a RMSE <0.05 m^3m^{-3}.

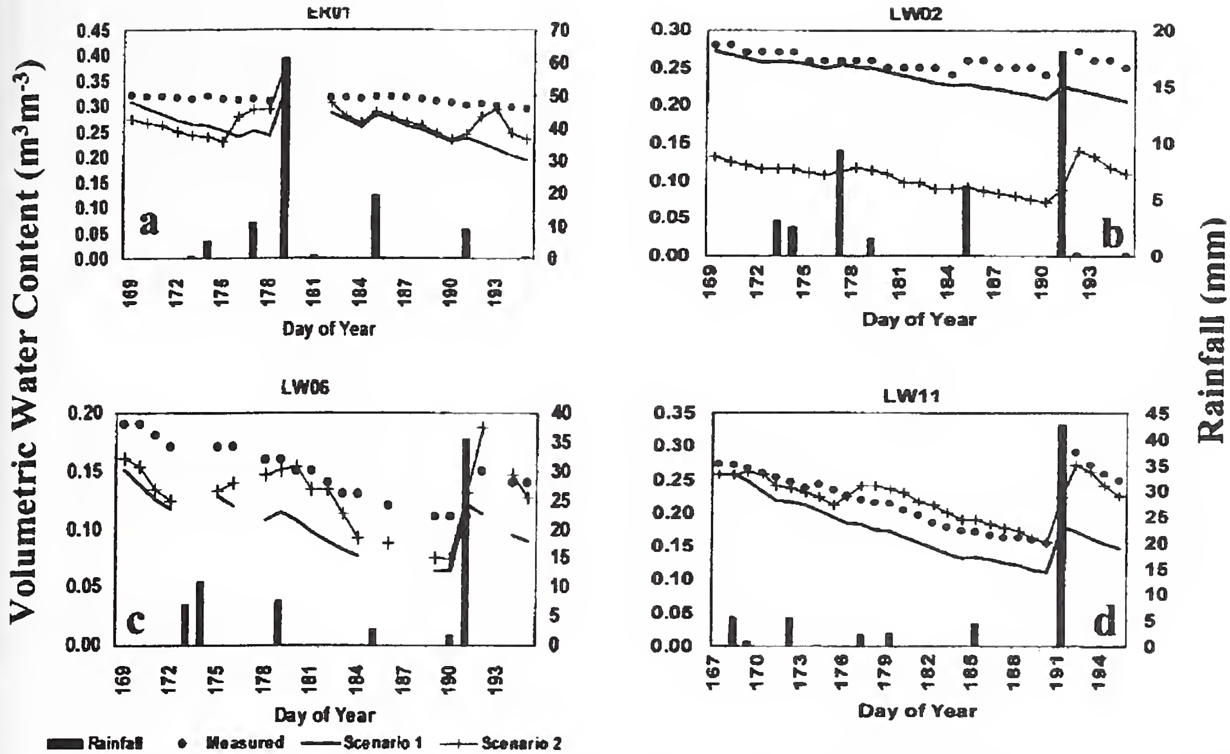

Figure 2a-2d. Time series plots of modeled root zone soil water content for study sites ER01 (a), LW02 (b), LW06 (c), and LW11 (d). Gaps in the time series reflect days when measured values were unavailable.

Scenario 2 – Assimilation of remotely sensed data

Model output at sites ER01 and LW02 did not agree well with measured data (d-index ≤ 0.23) (Figures 2a-d, Table 3), although at site ER01, both the ME and RMSE decreased by 0.01 m^3m^{-3} over that observed in scenario 1. At site LW02, assimilation of remotely sensed surface values into the model produced ME and RMSE values larger than any others encountered in the study. In contrast, the d-index, ME and RMSE values at sites LW06 and LW11 indicated good agreement with measured values.

Table 3. Results from statistical analysis of the comparison of modeled and measured root zone soil water content for the two scenarios.

Site	d	ME	RMSE	r^2
		Scenario 1		
ER01	0.25	0.05	0.06	0.990
LW02	0.52	0.02	0.03	0.996
LW06	0.56	0.04	0.05	0.985
LW11	0.69	0.05	0.06	0.973
		Scenario 2		
ER01	0.23	0.04	0.05	0.989
LW02	0.12	0.15	0.15	0.982
LW06	0.73	0.02	0.03	0.986
LW11	0.91	0.00	0.02	0.991

Conclusions

The objective of this paper was to examine the feasibility of inferring root zone soil water content by combining remotely sensed estimates of surface water content and modeling techniques. The model of Ragab (1995) was selected for this study because of its simplicity and because it does not require detailed soil physical, hydraulic, and vegetation properties to parameterize the model–properties that are not generally available or easily measured over large and/or spatially variable watersheds. This is particularly advantageous for applications where little is known about an area's soil physical properties, since the required model inputs may be estimated from general soil texture information (e.g., Rawls et al. 1982).

The original model was able to reproduce the time series patterns of root zone soil water content, but consistently underestimated measured values from0.02 to 0.05 m^3m^{-3}, on average. When remotely sensed data were assimilated into the model at sites ER01 and LW02, the model output underestimated measurements throughout the study period. At site ER01, the modeling results were similar to those observed in scenario 1, but the simulation at site LW02 underestimated measured values by about 0.15 m^3m^{-3} throughout the study period. Underestimation of the root zone water content at these 2 sites was probably a result of vegetational effects on the ESTAR data. Jackson et al. (1999) noted that tall grasses and heavy litter deposits will cause the ESTAR to underestimate the surface soil water content. Site ER01 was the most densely vegetated of the study sites and possessed the heaviest litter layer. Although site LW02 was classified as a rangeland site, there are a number of trees in the area which have the same effect on the ESTAR surface estimates. Thus, assimilated remotely sensed values from these sites probably led to underestimated root zone values. Jackson et al. (1999) indicated that adjustments to vegetational aspects of the ESTAR soil moisture retrieval algorithm can be made to better account for litter and trees. These adjustments will be necessary if microwave-based remote sensing techniques are to be widely used to assess soil water content. The results from scenario 2 suggest that it is feasible to infer root zone soil water content in tallgrass prairies by combining remotely sensed surface observations into a soil water budget model, provided that the remotely sensed data has not been corrupted by vegetational effects.

A remote sensing/modeling approach such as that described above could be integrated with weather forecasts and/or climate outlooks to project future soil water supplies, as well as assessing the current status of soil water content. Such assessments and predictions could be used by agricultural producers and others to schedule irrigation and predict crop or forage production rates, and by water resources managers to better manage watersheds and surface, soil, and groundwater water resources.

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Evaluation of Hyperspectral, Infrared Temperature and Radar Measurements for Monitoring Surface Soil Moisture

Ross Bryant, David Thoma, Susan Moran, Chandra Holifield, David Goodrich, Tim Keefer, Ginger Paige, David Williams and Susan Skirvin

Abstract

Remote sensing techniques for monitoring soil moisture were tested by comparing hyperspectral reflectance and spectral indexes; surface temperature (T_s) and thermal indexes; and normalized radar backscatter to soil moisture. A laboratory study indicated that hyperspectral reflectance and T_s were sensitive to surface soil moisture (r^2 range from 0.72 to 0.96). However, T_s was the only optical measurement that appeared insensitive to soil type. An index derived from differences between measurements of dry and wet soils (Δ-index) was presented and tested on the optical data as well as on data collected from two radar field studies at the United States Department of Agriculture – Agricultural Research Service (USDA-ARS) Walnut Gulch Experimental Watershed (WGEW). Using the Δ-index, radar backscatter measured by different satellite sensors was merged into a single relationship with surface soil moisture. Furthermore, the radar Δ-index may be physically related to surface soil moisture such that field-based empirical relationships may be unnecessary in sparsely vegetated environments.

Keywords: hyperspectral, thermal infrared, radar, soil moisture

Introduction

Soil moisture conditions at both the surface and deeper layers are primary determinants of cross-country mobility, irrigation scheduling, pest management, biomass production, and watershed modeling. Remote sensing has several advantages over other methods for monitoring surface soil moisture, such as synoptic, timely coverage with repeat passes, and efficiencies of scale that cannot be matched by traditional means. For these reasons, there is much interest in developing remote sensing techniques for monitoring surface soil moisture over large areas.

In this paper we examined two analytical methods, spectral and thermal measurements, and remote sensing radar observations obtained for surface soil moisture assessment. The goals of this work were to determine sensitivity of hyperspectral reflectance, thermal infrared (TIR) temperature and radar backscatter to changes in soil moisture.

Background

Hyperspectral

Soil moisture affects soil reflectance in two basic ways. First, a dry soil will almost always have a higher albedo because light is easily reflected out of soil interstices due to the large difference in the real index of refraction between air and soil mineral

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constituents. Radiation entering soil pores filled with water will have a greater chance of being absorbed or transmitted (Whalley et al. 1991). Second, dry soils have higher reflectance in specific spectral bands (1.450 &m, 1.940 &m, and 2,950 &m) due to lack of water absorption (Bowers and Hanks 1965, Twomey et al. 1986).

Spectral band ratios to ascertain water content in soil have met with qualified success. Musick and Pelletier (1988) found an overall weak correlation ($r^2 = 0.23$) between the ratio of two Landsat TM bands (TM5/TM7) and surface water content for 10 different soils but for any one soil in the group the r^2 ranged from 0.88 to 0.99. Hunt and Rock (1989) developed the Moisture Stress Index (MSI) by ratioing TM bands 4 and 5 (TM5/TM4) and compared it to relative water content in tree leaves and obtained r^2 values ranging from 0.75 to 0.95. A narrow band ratio index was proposed by Whalley et al. (1991) using the 1.45 &m water absorption band. Their index (which we call WISOIL) is the waveband ratio 1.45&m /1.3 &m. They found a curvilinear relationship with gravimetric water content for sandy and sandy loam soils up to 1 cm depth.

Thermal infrared

The temperature of the soil surface (T_s) is primarily dependent on the thermal inertia of the soil solution, which is strongly dependent on soil water content (Price 1982). Consequently, T_s has been related to surface soil moisture content for bare soils (Davidoff and Selim 1988). Because of the strong diurnal variations in T_s due to differences in solar radiation and atmospheric humidity, most applications are based on the difference between air temperature (T_a) and T_s, rather than simply T_s. The T_s-T_a has been the basis for many algorithms linking temperature measurements to soil evaporation and plant transpiration (i.e., as water evaporates, the surface cools).

Radar

The basis for soil moisture measurements using radar is the difference in dielectric constant, έ, for dry soil (έ = 2) and water (έ = 80). As the water content of a dry soil increases, so does the dielectric constant, which directly affects microwave backscatter, σ^o (Henderson and Lewis 1998). Microwave energy penetration of soil is on the order of several centimeters (van Oevelen and Hoekman 1999), but surface roughness and vegetation affect backscatter and their effects must be eliminated to accurately measure soil moisture (Sano et al. 1998). Other researchers have reported that surface roughness and vegetation influence backscatter as much or more than soil moisture (Zribi and Dechambre 2002, van Oevelen and Hoekman 1999). For this reason, the predictive capability of single polarization or single incidence angle radar for soil moisture is generally positive, but weak with r^2 = 0.06 and 0.09 for grass and shrub dominated sites respectively (Sano et al. 1998). Moran et al. (2000) reported better results when they took the difference between a reference (dry) image and changed (wetter) image ($r^2 = 0.93$). In this way, the difference in σ^o was due solely to change in water content when changes in vegetation and surface roughness were minimal.

Methods

The indexes used in this paper include T_s-T_a, WISOIL, MSI and the Δ-index. WISOIL and MSI indexes are defined as

$$\text{WISOIL} = \rho 1.45\ \&m / \rho 1.3\ \&m \qquad (1)$$

where ρ = reflectance in a particular wavelength, and

$$\text{MSI} = (\text{TM5}/\text{TM4}) \qquad (2)$$

where TM5 = Landsat Thematic Mapper band 5, and TM4 = Landsat Thematic Mapper band 4.

Every T_s, ρ and σ^o measurement had a concurrent measurement of the same soil in a dry state allowing normalization of all data to a dry reference condition. We call this the Δ-index, defined as

$$\text{N } \Delta\text{-index} = \text{abs}[(M_{wet}-M_{dry})/ M_{dry})*100] \qquad (3)$$

where M_{dry} = measurement (T_s, ρ or σ^o) of dry soil and M_{wet} = measurement (T_s, ρ or σ^o) of wet soil.

Optical

The optical experiment was conducted on pans of soil with a boom-mounted sensor under natural outdoor light May 7-9, 2003 in Tucson, AZ. The three soils used in the experiment were the Barnes, a dark colored silt/loam Mollisol with approximately 5% organic matter, the Whitehouse (B horizon), a red colored clay and iron rich Aridsol, and the Gila, a light colored, sandy loam Entisol.

The soils were sieved using a 2 mm screen and two samples of each were placed 3 cm deep in pans 21 cm in diameter. One sample of each soil served as a control and was never wetted. The other samples were filled with water to saturation and allowed to drain until all ponded water had soaked into the soil. Measurements were made of all samples with an Analytical Spectral Devices FR radiometer which measures radiation from 0.350 to 2.5 &m at 0.004 &m bandwidths. These measurements were made relative to a pressed halon panel to derive reflectance values. At the same time, soil surface temperature was measured with an Everest Interscience Infrared Thermometer with a 15 degree view angle. Soil moisture measurements were made with a factory calibrated Dynamax ML2X capacitance probe to 3 cm. Three soil moisture measurements were made in each sample and averaged. This procedure was repeated 10 times over a period of three days to document the change in spectral and surface thermal characteristics as the wetted soil samples dried.

Radar

The radar experiment was conducted between 1996 and 2003 on rangelands near Tombstone, AZ using soil moisture data collected from the field concurrent with satellite image acquisition. Radar backscatter, σ°, from ERS-2 and RADARSAT-1 satellite sensors (Table 1) and corresponding soil moisture data were obtained for 12 and 18 locations respectively. The ERS-1 backscatter coefficients were computed as the average of a 7 X 7 pixel window (8,100 m^2), while the backscatter coefficients for the RADARSAT-1 images were computed as the average of a 13 X 11 pixel window (9,100 m^2).

Table 1. Characteristics of radar imagery used in the study.

	ERS-2	RADARSAT-1
pixel resolution	12.5	8
polarization	V V	H H
incidence angle	23°	46°
frequency	C-band (5.3 GHz)	C-band (5.3 GHz)
wavelength	5.6 cm	5.6 cm

Coincident with the ERS-2 scene acquisitions, soil moisture was determined gravimetrically at 49 locations within each pixel cluster. Coincident with the RADARSAT-1 scene acquisitions, soil moisture was determined at 5 cm depth with factory calibrated Vitel probes in one location per pixel cluster. It was necessary to aggregate observations of soil moisture and backscatter for the RADARSAT-1 scenes because it was unlikely that a single isolated sensor could adequately represent soil moisture in large pixel clusters. Thus, the field sites were grouped according to antecedent moisture falling 3 days prior to RADARSAT-1 scene acquisition. This resulted in sample sizes of 9, 5, and 4 for the 3-day cumulative precipitation ranges of 0 to 0.5 cm, 0.5 to 1 cm and > 1 cm, respectively.

Results

Optical

The physical and chemical properties of the soils were apparent in the visible region, where the lighter colored soils were more reflective (Figure 1). Regression of waveband reflectance on water content indicated reflectance and simple indexes explained as much or more of the variation in soil moisture (Table 2) as the Δ-index ratios (Table 3) in most cases. Even though the solar zenith angle varied from 16° to 64° throughout the experiment, the soil surfaces were quite smooth, thus sun angle had less of an effect than it would in a field setting.

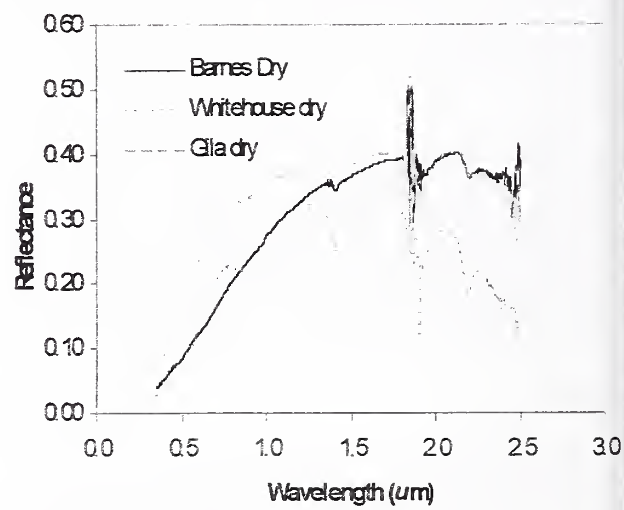

Figure 1. Reflectance signatures for soils used in the optical experiment.

The differences in reflectance of the soils were also apparent in the range of slopes and intercepts reported for any relationship between soil moisture and or reflectance index (Table 2). This finding was expected in light of the variation in soil color properties due to iron and organic matter content. However, the wide range of slopes and intercepts seen in the Δ-indexes for the three soils was unexpected (Table 3). The ratioing technique normalizes some of the spectral properties inherent in the different soils (Figure 2). If the soil surface had been more like a rough field soil, the Δ-index would likely have outperformed the other indexes.

Table 2. T_s and optical indexes versus soil moisture.

Index	Regress. Params.	Barnes	Whitehouse	Gila
		emittance versus soil moisture		
	r^2	0.73	0.85	0.79
T_s	slope	-0.65	-0.75	-0.76
	intercept	39.52	44.53	37.84
		index versus soil moisture		
	r^2	0.85	0.86	0.80
T_s-T_a	slope	-0.28	-0.75	-0.55
	intercept	4.68	18.35	6.56
	r^2	0.96	0.88	0.76
WISOIL	slope	-0.02	-0.02	-0.02
	intercept	1.17	1.28	1.01
	r^2	0.86	0.79	0.79
MSI	slope	-0.02	-0.02	-0.01
	intercept	2.29	1.77	1.15

The Gila soil generally demonstrated the weakest relationship between moisture content and reflectance due to a prominent step function between soil moisture and spectral properties (Figures 2a and 2b). There was a threshold at approximately 10 percent soil moisture where the spectral properties changed dramatically for a small change in the 3 cm integrated soil moisture. This may have been due to breakage of water menisci at the surface as it became dry while the deeper portion remained moist.

Surface temperature, T_s, was the only observation that had relatively uniform slopes and intercepts for all three soils (Tables 2 and 3 and Figure 2c). Therefore, it may provide a measure of surface soil moisture that is relatively independent of soil type.

Table 3. Optical Δ-indexes versus soil moisture.

Index	Regress. Params.	Barnes	Whitehouse	Gila
		---------------Δ/d Index---------------		
	r^2	0.91	0.81	0.88
Δ-T_s	slope	2.31	2.23	2.16
	intercept	-8.20	-17.92	4.17
Δ-narrow	r^2	0.91	0.84	0.72
$\rho 1.45\ \mu m$	slope	2.63	3.82	3.68
	intercept	-18.00	-70.86	-26.26
	r^2	0.84	0.81	0.71
Δ-wide	slope	2.02	2.75	2.86
ρTM 5	intercept	-14.84	-53.99	-23.14
	r^2	0.86	0.80	0.70
Δ-Albedo	slope	1.98	2.59	3.10
	intercept	-8.91	-54.11	-28.28
	r^2	0.66	0.78	0.87
Δ-T_s-T_a	slope	3.79	5.09	6.63
	intercept	19.08	-28.43	-3.10
	r^2	0.95	0.89	0.76
Δ-WISOIL	slope	1.65	2.38	1.86
	intercept	-13.67	-33.92	-2.22
	r^2	0.86	0.74	0.79
Δ-MSI	slope	1.08	1.27	0.59
	intercept	-22.02	-10.33	7.40

Radar

ERS-2 data from Moran et al. (2000) plus three additional data points from RADARSAT-1 were plotted together against soil moisture (Figure 3). In this case, soil moisture and the Δ-index nearly followed a 1:1 line, with scatter due to variability in soil moisture measurements and speckle in the radar backscatter.

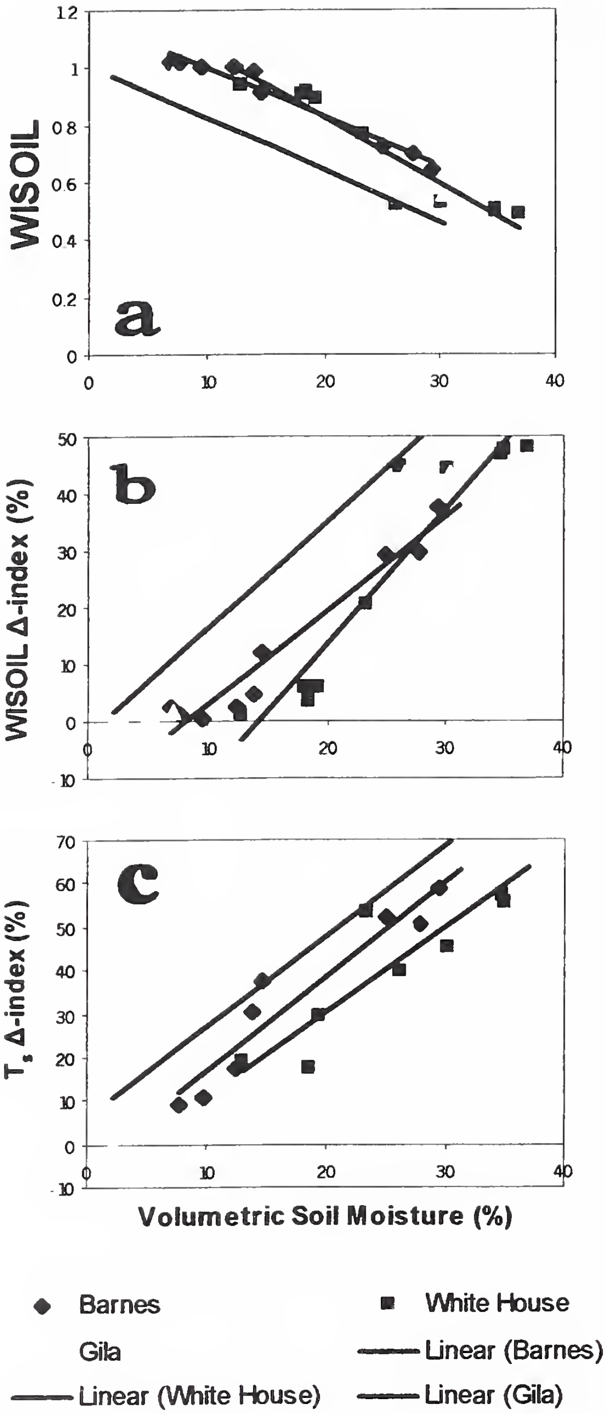

Figure 2. (a) The WISOIL index, (b) the WISOIL index when plotted as a Δ-index, and (c) surface temperature T_s plotted as a Δ-index.

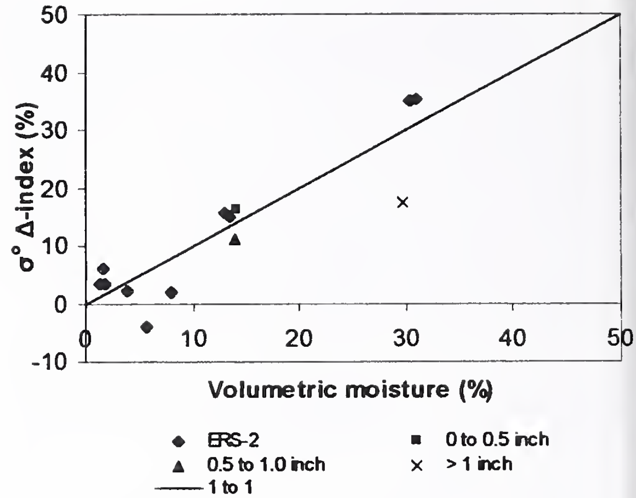

Figure 3. Relationship between delta index and observed soil moisture for ERS-2 and RADARSAT satellites. Diamonds represent ERS-2 observations, and all others RADARSAT-1. Depths indicate cumulative precipitation that fell on soil moisture recording sites 3 days prior to RADARSAT-1 scene acquisition.

Conclusions

Results specific to this study:

1) Reflectance of optical wavelengths were strongly related to soil moisture integrated over a 3 cm depth as indicated by high r^2 values, though a unique equation may be required for each soil type. Additionally, abrupt changes in surface reflectance with small changes in integrated moisture were observed which require further investigation.

2) T_s was the only optical measurement that had relatively similar slopes and intercepts for all three soils. This indicated it might be capable of accurate soil moisture predictions on a wide variety of soil types without the development of empirical relationships for different soils.

3) ERS-2 and RADARSAT-1 data plotted against soil moisture approximated a 1:1 line that suggested data from multiple satellites could be merged to derive surface soil moisture where vegetation is minimal. Additionally, this suggests that empirical relationships between soil moisture and the Δ-index may be unnecessary.

These three results should be evaluated in terms of watershed applications. In that context, several issues need to be addressed.

Data availability is an issue common to all approaches. Although optical remote sensing data are more easily obtained and typically costs less, availability is often limited by poor weather conditions. Thermal remote sensing systems are uncommon and generally provide spatial resolution that is too coarse for watershed scale monitoring. Though radar has good spatial resolution and all weather capability, it is generally more expensive than optical data.

All approaches examined here measure only surface soil moisture (to depths of millimeters in optical bands and centimeters in radar) though management decisions are often based on estimates of root zone soil moisture to depths of 0.5 to 1 m. Methods to extend surface measurements to meaningful depths will make remote sensing of surface soil moisture more useful in watershed management.

Though it may be possible to use ρ, T_s or σ^o to determine surface soil moisture, only σ^o offers the potential for directly measuring soil moisture without the need to derive field-based empirical relationships. For this reason, and because radar data from existing satellite platforms is available, provides a good combination of spatial resolution, and depth integration, it is a powerful tool for watershed soil moisture monitoring.

Acknowledgments

The authors appreciate the support of Kevin Slocum and Joseph Watts at the U.S. Army Engineer Research and Development Center, Topographic Engineering Center, Frank Bårnes at the Department of Hydrology and Water Resources, University of Arizona, and Tom Jackson at the USDA-ARS, Hydrology and Remote Sensing Laboratory.

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Monitoring Climate and Weather Variability in Mississippi

Carlos V. Alonso, John A. Augustine, Tilden Meyers, Garry L. Schaefer, Loren White

Abstract

Climate variability and weather uncertainty pose challenges and opportunities for U.S. scientists assessing the potential consequences of climate change. Recent extreme weather patterns and multibillion-dollar impacts on a wide range of agricultural and commercial businesses are stimulating the need for this assessment. The objective of this poster is to highlight contributions the ARS Goodwin Creek Experimental Watershed is making to national and regional climate and weather monitoring projects. Goodwin Creek houses instrumented stations that are part of the national SURFRAD and ISIS networks developed by NOAA for long-term monitoring and climate research. Their objective is to support climate research with continuous, high quality measurements of the earth's surface radiation budget and energy transfer balance. Four of these NOAA stations were located within the Mississippi River Basin in support of the GEWEX Continental-Scale International Project. The SCAN network was established by NRCS to support natural resources assessments and agricultural conservation activities through integration of information from existing soil-climate data networks, and the establishment of new data collection points through partnership with federal, state, local, and tribal entities. Nine SCAN stations are currently located in Mississippi: two in the Goodwin Creek watershed, two in east central locations, and the rest distributed through out the Delta. Mississippi is home to some of the most active weather in the world; severe thunderstorms and tornadoes are a threat year-round, and winter storms can bring devastating ice accumulations. Development of a Mississippi meteorological mesonet is in the planning stages under the leadership of the Meteorological Program at Jackson State University, Mississippi. As envisioned, all SCAN stations in Mississippi, including those in the Goodwin Creek watershed, will contribute to the planned mesonet.

Keywords: climate, weather, remote sensing

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Ecology I

The Effects of Ecosystem Restoration on Nitrogen Processing in an Urban Mid-Atlantic Piedmont Stream

Paul Mayer, Elise Striz, Robert Shedlock, Edward Doheny, Peter Groffman

Abstract

Elevated nitrate levels in streams and groundwater pose human and ecological threats. The U.S. EPA, USGS, Institute of Ecosystem Studies, and Baltimore County Department of Environmental Protection are collaborating on a multi-year study of the impacts of stream restoration on nitrogen processing in Minebank Run, a Piedmont stream in Baltimore County, Maryland. The study is designed to investigate the nitrate removal capacity of this stream before and after restoration. Restoration techniques such as bank re-shaping, bank reinforcement, and energy dissipation structures will be constructed to reestablish geomorphic stability lost due to impacts from storm water runoff. We will quantify the effects of specific restoration techniques on microbial denitrification, a process that removes nitrate but which requires anaerobic (saturated) conditions and adequate supply of dissolved organic carbon from detritus and organic soils. Restoration may enhance denitrification by increasing groundwater saturation and/or by increasing carbon supply to denitrifers in the subsurface. Therefore, stream geomorphology, surface flow, groundwater flow, and geochemistry are being quantified throughout the stream reach and in a network of 51 wells and piezometers installed at the site corresponding to the restoration techniques of interest. Denitrification activity will be measured throughout the stream and related to limiting factors such as dissolved organic carbon and dissolved oxygen. 3-D hydrologic models of nitrate movement will be developed for the watershed. Our study results will be used to develop stream restoration approaches for reducing nitrate pollution in urban watersheds.

Keywords: urban stream, restoration, assessment, nitrate, denitrification, hydrology

Introduction

Urban streams undergo ecological degradation from various stressors such as channel incision from flashy storm water runoff, chronic atmospheric and terrestrial nutrient inputs, and hydrologic disconnection from the floodplain. However, stream restoration is considered to be a means of alleviating such stressors. Restoration may include numerous individual structural manipulations, each with the potential to alter ecosystem processes. Yet, relatively little effort has been devoted to understanding the basic ecology of urban streams or the response of urban streams to restoration. Here we describe a multi-investigator project currently underway that proposes to identify ecosystem structural and functional response to stream restoration at Minebank Run, a stream located on the eastern Piedmont of Maryland in a highly urbanized region north of Baltimore. Minebank Run is located in a watershed that flows into Chesapeake Bay.

Our research focus is on nitrate nitrogen, a pollutant that threatens ecosystem and human health. Restoring streams in a manner that attenuates and removes nitrate may be a cost-effective means of improving water quality in urban watersheds.

Our research approach was to examine Minebank Run before and after restoration to measure and

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identify controls on microbial denitrification, a natural process occurring in soils and groundwater that removes significant amounts of nitrate in waters by transformation to a biologically inactive gas form. Denitrification is an anaerobic process requiring low or depleted oxygen levels (Tiedje et al. 1982). Microbial denitrifiers require a source of carbon for respiration and often are limited by carbon supply (Korom 1992). Therefore, understanding when and where in the watershed carbon and oxygen levels are optimal for denitrification is critical to identifying approaches for enhancing the conditions necessary to effect nitrogen removal from the stream.

A full understanding of the effects of restoration on nitrate processing in urban streams will require simultaneous study of both biotic and abiotic stream processes. Therefore, our approach depends upon the efforts of several research teams to evaluate the biology, geology, hydrology, and chemistry of an urban stream before and after restoration (Figure 1).

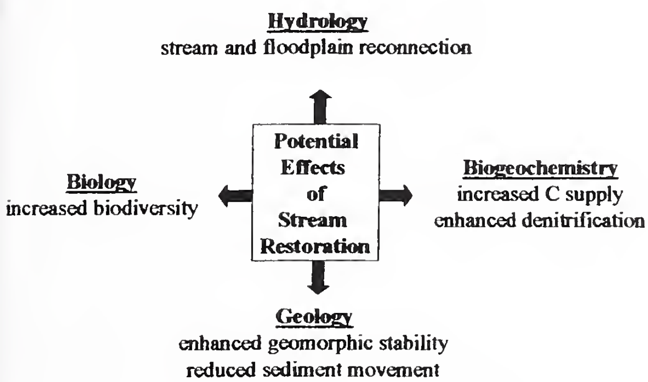

Figure 1. Potential effects of stream restoration.

The upper half of Minebank Run was restored in 1998 and 1999 to improve geomorphic stability and reduce channel incision. The remaining lower reach will be restored in 2004 with using a similar engineered approach employing techniques such as a) installing step-pool structures designed to reduce erosion, b) reshaping stream banks to reconnect stream channel to flood plain, c) armoring stream banks against erosion with large boulders, d) reconstructing stream meander features and riffle zones, and e) re-establishing riparian plant communities.

Restoration may enhance denitrification by reestablishing flood plain hydrology and/or increasing carbon availability. Structures installed in the stream channel to reduce erosion also may trap organic matter long enough to create enriched anoxic zones conducive for denitrification to occur. Restoring and replanting riparian zones may provide the necessary litter inputs for supplying carbon. But, because much of the riparian zone in urban systems cannot be reclaimed or replanted (Paul and Meyer 2001, Pickett et al. 2001), carbon-containing compounds (e.g. glucose, ethanol, yard waste) may need to be added to the stream or subsurface to provide adequate carbon supply to microbes. (Obenhuber and Lowrance 1991, Qian et al. 2001). Therefore, identifying stream features, either natural or constructed, where high denitrification activity occurs may provide important nitrate reduction tools and direct future restoration efforts (Burt, et al. 1999, Groffman et al. 2002, Groffman and Crawford 2003).

Our broader objectives were to: 1) assess ecosystem benefits of restoration, 2) identify stream restoration methods that enhance nitrate control, 3) develop predictive models of stream hydrology, and 4) develop ecologically-based guidelines for stream restoration. Lab and field-based research has and will be conducted collaboratively by scientists from U.S. Environmental Protection Agency, U.S. Geological Survey, Baltimore County Department of Environmental Protection, and Institute of Ecosystem Studies. The data presented here characterizes the unrestored condition of Minebank Run. Our results are preliminary in the sense that we cannot yet compare the conditions of the restored stream with our baseline information. However, the information collected to date provides critical insight into the level of ecosystem function in this degraded urban stream.

Methods

Data presented here relate to the potential controls on dentrification, including overall dissolved organic carbon, dissolved oxygen, and concentrations of bio-reactive nitrogen (nitrate plus nitrite). We collected surface water samples from the Minebank Run and groundwater samples from piezometers nests installed in the stream channel. Piezometers were installed in three locations along the stream corresponding to planned restoration measures including bank reshaping, deposition of rip rap, and stream channel relocation. Piezometers consisting of 2.5 cm diameter stainless steel pipes with 0.25-mm wire-wrapped screen at the bottom 15-cm of the standpipe were positioned in nests of 3 at 61, 122, and 183 cm below the surface of the stream bed.

Additional triplicate nests of piezometers were installed on each stream bank perpendicularly to the channel approximately 4m from the stream piezometers at depths that matched the mean sea level of the corresponding stream bed depths. A total of 33 piezometers and 4 surface water stations were sampled. Water was sampled seasonally in December 2001, March, May, July, and October 2002. Water was collected with a peristaltic pump through a flow cell and Hydrolab instrument to measure dissolved oxygen. Samples were stored on ice and filtered in the lab with 0.45 micron filters for dissolved organic carbon (DOC). DOC was measured on a Dohrman or Tekmar instrument that measures organic carbon directly via UV-persulfate digestion method. Nitrate and nitrite concentrations were measured on unfiltered samples using Lachat Flow Injection Analyzer.

We measured microbial biomass carbon and denitrification enzyme activity (DEA) in hyporheic sediments sampled at the shallow depth of the near and in-stream piezometers described above and in saturated, deep floodplain sediments that were taken when wells were established for groundwater flowpath analysis. Microbial biomass carbon was measured using the chloroform fumigation-incubation method (Jenkinson and Powlson 1976). Soils were fumigated to kill and lyse microbial cells in the sample. The fumigated sample was inoculated with fresh soil, and microorganisms from the fresh soil grew vigorously using the killed cells as substrate. The flush of carbon dioxide (CO_2) released by the actively growing cells during a 10-day incubation at field moisture content were assumed to be directly proportional to the amount of carbon and nitrogen in the microbial biomass of the original sample. A proportionality constant (0.45) was used to calculate biomass carbon from the CO_2 flush. Carbon dioxide was measured by thermal conductivity gas chromatography.

DEA was measured using the short-term anaerobic assay developed by Smith and Tiedje (1979) as described by Groffman et al. (1999). Sieved soils were amended with nitrate, dextrose, chloramphenicol and acetylene, and were incubated under anaerobic conditions for 90 minutes. Gas samples were taken at 30 and 90 minutes, stored in evacuated glass tubes and analyzed for N_2O by electron capture gas chromatography.

Results

The seasonal pattern of DOC concentration in surface water did not necessarily match that in groundwater (Figure 2). March groundwater DOC levels were high on average when compared to surface water, though concentrations in the piezometers were highly variable. This result may reflect the seepage of high carbon waters from the riparian zone into the banks and/or hyporheic zone. The reverse is true in July when surface water DOC levels were much higher than in groundwater. High surface water DOC levels may reflect runoff events discharging allochthonous material from the riparian zone into the stream and/or increased production of autochthonous material in stream from epiphytic algae.

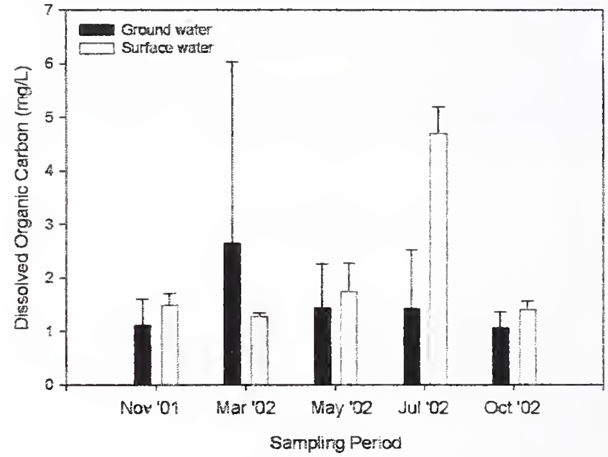

Figure 2. Mean surface water and groundwater DOC concentration over time at Minebank Run, Baltimore County, MD, USA (Error bars = 1 SD).

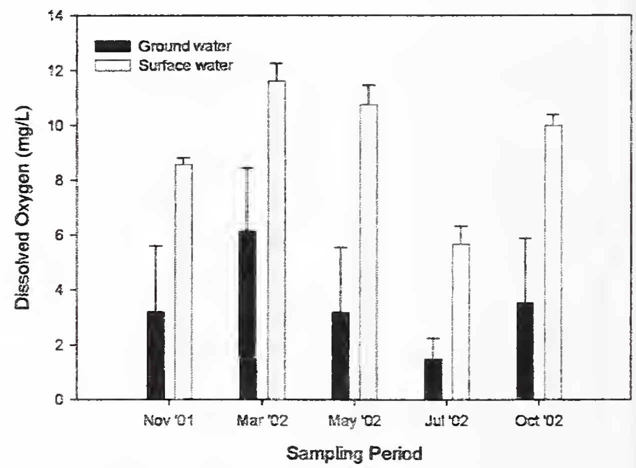

Figure 3. Mean surface water and groundwater dissolved oxygen concentration over time at Minebank Run, Baltimore County, MD, USA (Error bars = 1 SD).

Seasonal patterns of dissolved oxygen (Figure 3) and of bioreactive nitrogen (Figure 4) in groundwater and surface water appear much more synchronized. That is, groundwater oxygen levels match that of surface water because hyporheic flow appears to strongly dictate the amount of oxygen entering the groundwater. Oxygen levels declined dramatically in July when a mid-summer drought reduced base flow of the stream creating some stagnant pools. On average, nitrogen responds similarly but with greater variability in groundwater concentrations during a sampling period (Figure 4).

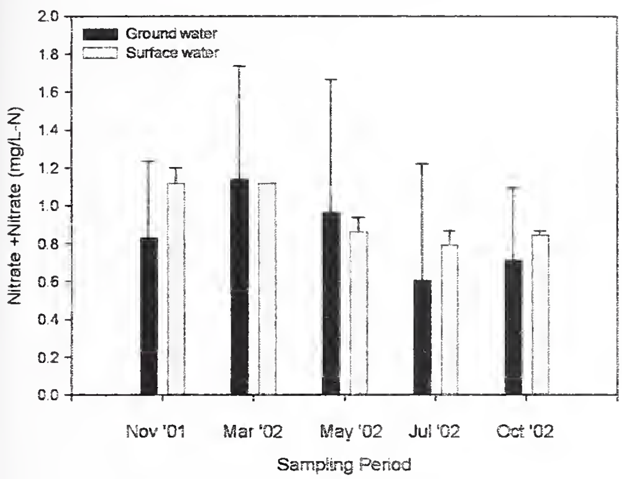

Figure 4. Mean surface water and groundwater bioreactive nitrogen concentration at Minebank Run, Baltimore County, MD, USA (Error bars = 1 SD).

Patterns of DOC by depth and location (Figure 5) indicate that mean DOC concentration is higher in the stream and hyporheic zone than in the banks, suggesting that autochthonous inputs of carbon (e.g. algae and aquatic vegetation) are significant. Furthermore, mean DOC increased (but with high variability) at the 183-cm depth suggesting a deposition of carbon perhaps because sediment layers were less porous at this depth.

Dissolved oxygen concentrations declined with depth in both the stream channel and the banks (Figure 6). Oxygen in the banks remained relatively high even at the 183 cm depth and corresponded closely to oxygen concentration in the stream hyporheic zone. Depending on the height of the bank, piezometer depth in the banks was over 3 meters below the surface of the soil. Therefore, the close correspondence in oxygen between stream channel and banks suggests that oxygen flows from the hyporheic zone into the banks.

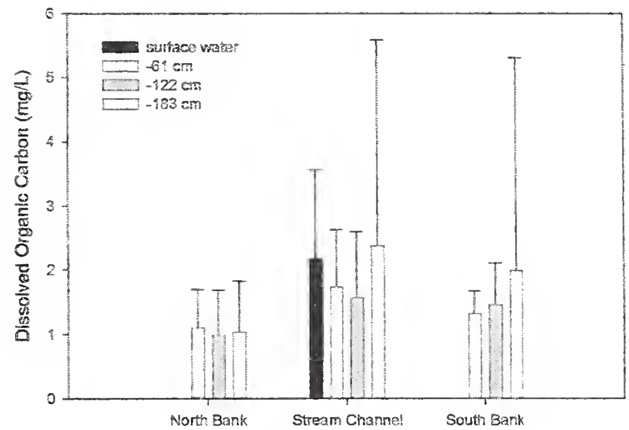

Figure 5. Mean DOC concentration in stream channel and groundwater at Minebank Run, Baltimore County, MD, USA (Error bars = 1 SD).

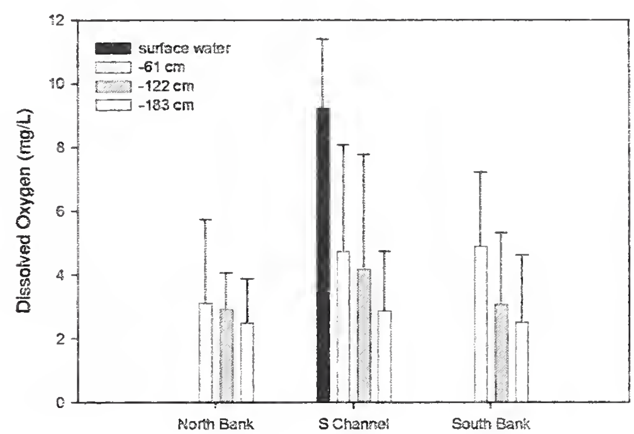

Figure 6. Mean dissolved oxygen concentration in stream channel and groundwater at Minebank Run, Baltimore County, MD, USA (Error bars = 1 SD).

In general, mean bioreactive nitrogen concentrations decreased with depth but, most notably, were higher on the north bank of Minebank Run (Figure 7), reflecting, perhaps, the influence of runoff (e.g. fertilizer) from
a suburban residential development on the north side of the stream. The area along the south bank of Minebank is several hundred meters from the nearest concentrated urban development.

Microbial biomass carbon and DEA (Figures 8 and 9) were both higher in hyporheic sediments (in and near stream piezometers) than in deep floodplain sediments. These results support the idea that the hyporheic zone is responding to and processing carbon and nitrate from upstream and/or riparian sources. These results also suggest that the restoration technique(s) that increase carbon flow to these sediments could increase denitrification capacity of the stream ecosystem.

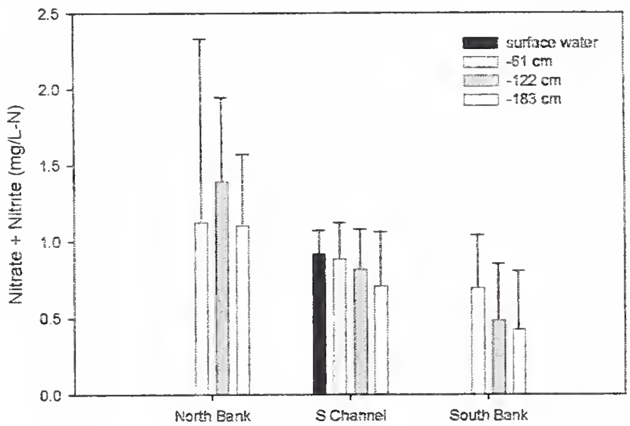

Figure 7. Mean bioreactive nitrogen concentration in stream channel and groundwater at Minebank Run, Baltimore County, MD, USA (Error bars = 1 SD).

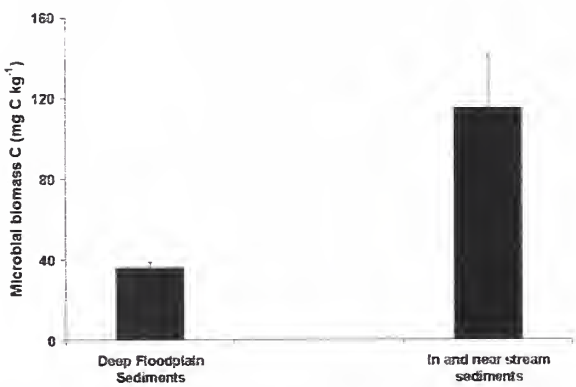

Figure 8. Microbial biomass carbon (C) in hyporheic (in or near stream) and deep floodplain sediments from
Minebank Run, sampled in November 2001.

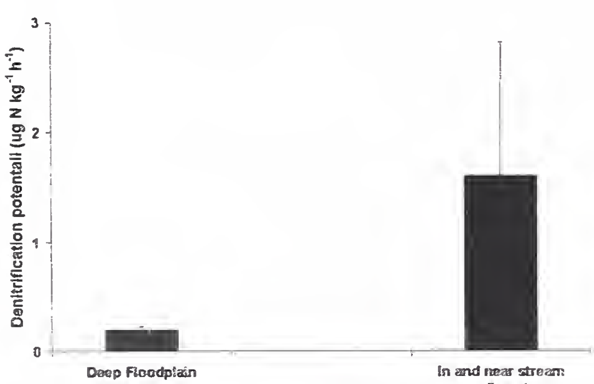

Figure 9. Denitrification enzyme activity in hyporheic (in or near stream) and deep floodplain sediments from Minebank Run, sampled in November 2001.

Conclusions

DOC levels at Minebank Run were relatively low throughout the study but seasonal patterns are evident in the surface water. Stream sediments appear to be supplied with carbon from upstream sources. Thus, restoring riparian vegetation where possible may serve to increase carbon input to the stream. However, artificial additions of carbon (yard waste, ethanol, etc.) also may be an important management approach to increasing denitrification at Minebank Run and other urban streams.

Dissolved oxygen levels were lowest in July at Minebank Run probably due to a combination of drought conditions, low base flow, and high water temperatures. Corresponding nitrate and nitrite levels also were low in July suggesting that nitrogen removal from urban streams may vary seasonally or, at least, in relation to conditions conducive to producing low dissolved oxygen levels. However, nitrogen uptake by vegetation also may be important. Therefore, it may be easier to effect a reduction in stream nitrogen during spring or summer seasons whereas nitrate may be transported with little biological transformation during colder months and/or high base flow. Regardless of season or temperature, precipitation events can potentially flush nitrogen and organic matter rapidly out of the system and into receiving water bodies (e.g. Chesapeake Bay), bypassing any processing by denitrifying microbes. Channel restoration techniques effective at reducing the flashy flows at Minebank also may reduce the potential for nitrogen flush and/or increase the retention of carbon in the channel. Other techniques involving channel and bank reshaping may reconnect stream and floodplain hydrology in a way that increases the duration or region of saturation in carbon-rich soils of the riparian zone. Such changes may sustain a high potential for denitrification.

Management of nitrogen in urban streams via restoration will require a combination of technologies and approaches specific to the chemistry, geology, hydrology, and biology of the site. With the project still in the pre-restoration stage, we cannot assess the effects of stream restoration on denitrification or on overall water quality. However, we expect to demonstrate that nitrogen processing in urban streams is strongly controlled by geomorphology, hydrology, and carbon supply. If true, such results would point to a number of ecologically based guidelines for stream restoration that would optimize the nitrogen attenuation capacity of urban streams in a manner that would benefit overall water quality and ultimately, ecosystem health.

Acknowledgments

The views expressed in this manuscript are those of the authors and do not necessarily reflect the views and policies of the U.S. Environmental Protection Agency.

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Vegetation Community Impacts on Soil Carbon, Nitrogen and Trace Gas Fluxes

Dean A. Martens, Jean E. T. McLain

Abstract

Changes in C and N cycling were pronounced with mesquite (*Prosopis* spp.) N inputs into a previously N limited grassland or riparian area. Expansion of mesquite into semi-arid grasslands (*Sporobolus* spp.) increased soil C content and decreased C isotope values compared with soils of non-mesquite grasslands. Cool season litter (2.9% N) collection (October 2001 to March 2002) recovered 66 g mesquite C and 4.5 g mesquite N m^{-2}. Measurement of surface litter present in the different riparian plant communities ranged from 750 g litter m^{-2} (mesquite-sacaton), 598 g litter m^{-2} (mesquite) to160 g litter m^{-2} (annuals and forbs). An estimated 3 to 8 years of high quality plant litter remained on the soil surface under mesquite due to yearly inputs. Soil cores removed from the mesquite understory and incubated at constant moisture potentials determined CO_2–C was linearly respired for up to 80 d indicating the litter remaining on the mesquite soil was highly labile, but the disconnect between high litter fall and low moisture conditions present in the ecosystem allowed the mesquite litter to accumulate. Changes in the chemistry of the mesquite soil, especially the N content, resulted in greater fluxes of the greenhouse gas nitrous oxide compared to grass or sacaton soils, but no differences were found in carbon dioxide evolved from the three sites. Of interest was the finding that the semi-arid soils were a strong sink for atmospheric methane during the majority of the year.

Keywords: riparian, nitrous oxide, carbon dioxide, methane

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Introduction

The abundance of woody plant species in semi-arid environments has increased substantially during the last 50 to 300 yrs with a concomitant observed loss of grassland productivity (Humphrey 1958). Such changes in plant species over a large area have potential regional and global impacts, including increased desertification and changes in watershed hydrology, nutrient cycling, soil erosion and energy fluxes. Shifting dominance among herbaceous and woody vegetation also alters primary production, plant allocation, rooting depth and soil faunal communities that in turn affect nutrient cycling and carbon storage (Jackson et al. 2000).

The quantity and composition of the plant litter falling to the soil surface has a tremendous impact on the composition of the soil carbon (C) and nitrogen (N) pools (Barth and Klemmedson 1978). When water is available during summer rainy periods, semi-arid ecosystem productivity is tightly linked to soil nitrogen availability (Ettershank et al. 1978). Mesquite is a N-fixing legume that produces high quality litter compared with the low N, high fiber litter from native grass vegetation, and thus, the quality and quantity of soil C and N under and in between woody species increases with mesquite abundance (Klemmedson and Barth 1975). Tiedemann and Klemmedson (1973) reported that N availability was up to 15 times higher under mesquite than in adjacent open sites. The increased availability of N under mesquite coupled with the lack of leaching from semi-arid environment can result in very high spatial variability of N and a strong localized impact on N cycling.

In this study, we investigated the distribution of soil C and N content between a mesquite, a mesquite-sacaton, a sacaton (without mesquite), and a brush with forb and annual grass community. The C and N content of the vegetation and the litter fall were also monitored. The greenhouse gases carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4) were also measured to determine the impact of soil C and N distribution with different vegetation on trace gas fluxes.

Methods

Site and soil sampling

The study site is located along the San Pedro River just south of Fairbank, Arizona on river terraces that were historically composed of grasses and forbs. The study site encompassed three distinct vegetation types, the first dominated by velvet mesquite (*Prosopis velutina*), a leguminous tree (mesquite site). The second soil was dominated by sacaton (*Sporobolus wrightii*), a perennial bunchgrass (sacaton site), and the third was populated by annual herbaceous dicots, including peppergrass (*Lepidium thurberi*), Fremont's goosefoot (*Chenopodium fremontii*), and toothleaf goldeneye (*Viguiera dentata*) (open/outside site). Additional soil samples were taken from a mesquite site with a sacaton under story (mesquite-sacaton). Triplicate soil and litter (O-horizon) samples were collected to a depth of 60 cm from the different vegetation zones, weighed for bulk density determination, processed to remove rocks and passed through a 1 mm sieve.

Litter and soil analyses

Samples were analyzed for pH (2.5 g soil:10 mL 0.05 M CaCl) and C and N by dry combustion with isotope analysis (Europa mass spectrometer). Carbohydrate content (Martens and Loeffelmann 2002) and amino acid content (Martens and Loeffelmann 2003) were extracted by acid digestion and determined by ion chromatograph and pulsed amperometry detection. Mineralization experiments were conducted with cores removed from the site (0-5 cm with litter and 5-10 cm) and incubated at 20°C in 1 liter sealed jars fitted with gas sampling valves. The amount of CO_2-C evolved from the incubations was determined with an infrared gas analyzer (Qubit Systems, Inc., Kingston, Ontario) for up to 78 d.

Trace gas measurement

Nitrous oxide and methane fluxes were measured by the static chamber method (Hutchinson and Mosier 1981) using 22-cm diameter PVC collars permanently installed at the soil surface. Lids were firmly affixed to the collar and 10 ml sub-samples of the chamber atmosphere were removed through a sampling port every 15 min for 1 hour. Gas sub-samples were transported to the laboratory and analyzed within 24 hours using a Shimadzu GC14-A Gas Chromatograph (Shimadzu Corp., Columbia, MD), fitted with dual detectors (Flame Ionization Detector and Electron Capture Detector), and an 80/100 HayeSep-Q column (Supelco, Inc., Bellefonte, PA). Net gas fluxes were then calculated from the exponential regression of the time series of gas concentrations within the chamber headspace (Koschorreck and Conrad 1993).

CO_2 fluxes were also measured using the static chamber with sealed lid. After 1 hour, the headspace air was pumped from the chamber into an infra-red gas analyzer (Qubit Systems, Inc.), and the CO_2 concentration in the chamber headspace was used to quantify net efflux from the soil surface.

Soil moisture potentials were measured at the 0-5 and 5-10 cm depths using permanently installed gypsum blocks and the Delmhorst KS-D1 Soil Moisture Tester system (Delmhorst Instrument Company, Towaco, NJ). Soil temperatures were also monitored on sampling days using a long stem handheld thermometer (Fisher Scientific, Pittsburgh, PA).

Results

Detailed chemical analysis of litter at the soil surface shows vast differences between the plant communities included in this study, indicating that the mesquite has the potential to alter the C and N cycling in this semi-arid riparian area (Table 1). The high N content (up to 3.5% N) of the mesquite litter enhances the under story sacaton grass productivity compared to sacaton sites without the input of mesquite-N. The yearly pulse of mesquite N increased sacaton productivity in the mesquite grove to 550 g (n = 3 plants) of above ground biomass compared with 148 g (n = 3) in the sacaton grassland without mesquite. The increased productivity of the mesquite-sacaton community has resulted in accumulation of large amounts of C and N in the litter and soil horizons (Table 1). The incorporation of mesquite C into the soil C pool can be determined by investigating the soil □$^{13}C/^{12}C$ composition (Table 1). The mesquite tree is a C_3 plant that produces litter with a reduced □$^{13}C/^{12}C$ enrichment of –27‰ compared with the –14 to –15‰ found for the C_4 sacaton grass. The lower ^{13}C content of soils sampled from the mesquite community is an indication of the importance of mesquite inputs to the soil C. Leaf litter fall results from the fall 2001 through spring 2002 found an average yearly input of 66 g litter C and 4.5 g litter N per m^{-2} from the mesquite-dominated communities. This large litter input from the mesquite and mesquite-sacaton sites results in pronounced litter layers not found in the non-mesquite "outside" sites (Figure 1). In mesquite sites, an organic matter build-up representing 3 to 8 yrs of litter fall is present under the trees, due to two possible reasons. First, the litter may increase yearly because of an existing disconnect between litter input and the seasonality of rainfall. The time of maximum litter input (fall and early winter) does not historically coincide with sufficient rainfall necessary to support maximum decomposition. Second, the chemistry of the mesquite litter may render it resistant to decomposition. In addition, seasonal cold winter temperatures may influence the accumulation of litter fall C and N as winter moisture has little impact on litter mineralization due to very cool temperatures that limit microbial activity. Thus the lack of residue mineralization may be due to temperature, moisture, residue biochemistry, or interactions among all three.

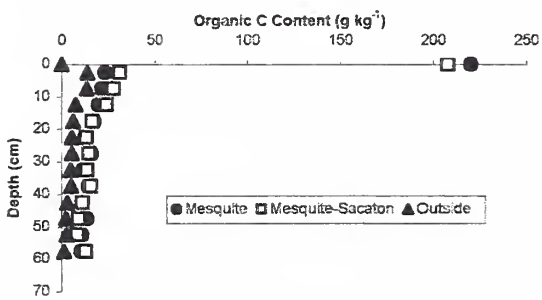

Figure 1. Organic C content of the litter and soil under different vegetation communities to 60 cm.

Laboratory studies found that a substantial portion of the mesquite litter is composed of carbohydrates and amino acids, substrates which do not accumulate in the litter layer or soil in more temperate areas, where plants and microbes favor them as highly labile forms of C and N. When soil and litter samples from the mesquite-sacaton site were incubated at optimum moisture conditions, the mineralization of plant litter and soil C was linear for the 78 d incubation (Figure 2). The results confirm that the mesquite litter will mineralize at a very fast rate if optimum moisture and temperature conditions are present and confirms that litter accumulates in the mesquite sites due to the interaction of the seasonality of precipitation events and soil temperatures.

The high soil N content (Table 1) and the rapid rate of mineralization (Figure 2) result in enhanced sacaton growth as sacaton growing under the mesquite canopy is larger and has a higher leaf N content than sacaton growing in open areas with no mesquite N additions. Enhanced plant productivity resulting from the mesquite-N also translate into increased soil C and N in the mesquite grove. Soil C to a 60 cm depth is twice as high under the mesquite and the mesquite-sacaton communities than in sacaton outside of the mesquite grove (Figure 3).

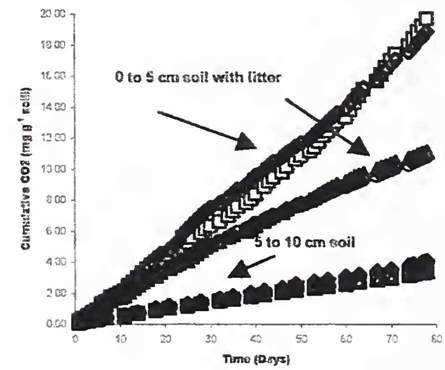

Figure 2. Cumulative carbon dioxide-C evolved from mesquite soil cores incubated at 23 °C for up to 78 d.

The importance of N additions for the increase in soil C has just begun to be understood (Martens and Loeffelmann 2003). It has been shown that N addition from atmospheric or plant fixation is important for stabilization of the N-organic matter mineral complexes (Neff et al. 2002). The close correlation found between N content and C content of the soils in this San Pedro riparian zone is shown in Figure 4.

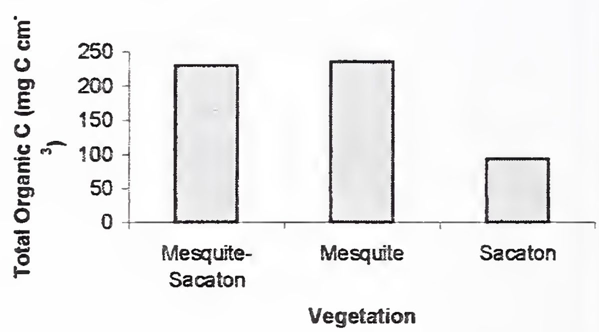

Figure 3. Organic C content to a 60 cm depth accumulating under different vegetation communities.

Because of the wide range in soil C and N under the different vegetation communities (Table 1) and the importance of C and N cycling to the production of greenhouse gases, the fluxes of CO_2, N_2O and CH_4 were monitored to confirm the climate forcing or mitigation potential of this semi-arid riparian zone.

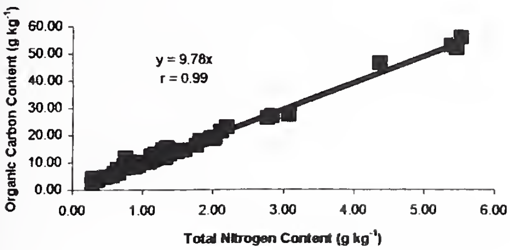

Figure 4. Relationship between C and N in the San Pedro riparian soils.

Nitrous oxide

Field monitoring prior to the 2002 monsoon rains showed that N_2O fluxes were nearly zero in the mesquite, sacaton, and grass/forb "open" vegetation systems (Figure 5), but emissions were evident soon after monsoon moisture inputs began in July. From July through the end of the monsoon in mid-September, N_2O emissions from the mesquite soils averaged 21.02 ± 13.42 μg N_2O m^{-2} h^{-1}, a number ~6 times higher than the sacaton site (3.66 ± 4.70 μg N_2O m^{-2} h^{-1}) and more than 10 times higher than the open site (1.91 ± 3.99 μg N_2O m^{-2} h^{-1}). As soils dried and seasonal temperatures fell after monsoon rainfalls ended in the fall 2002, N_2O emissions in all 3 vegetation sites fell to nearly zero (Figure 5), but increased temporarily in response to sporadic rainfall and warmer temperatures in March and April 2003. Nitrous oxide production was highest following heavy rainfall, leading to a weak, but significant relationship between soil moisture potential and N_2O efflux across all three sites ($r^2 = 0.33$, $p = 0.01$). When data from each vegetation type was considered separately, however, the correlation between N_2O efflux and soil moisture was only significant in the mesquite ($r^2 = 0.47$, $p = 0.03$) and not in the sacaton ($r^2 = 0.26$, $p = 0.24$) or the open ($r^2 = 0.02$, $p = 0.93$) areas.

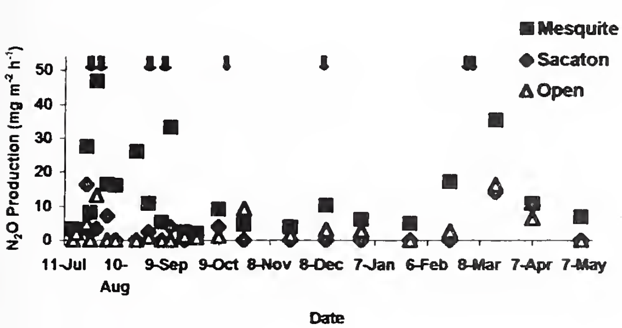

Figure 5. Flux of N_2O from the San Pedro soils in 2002 and 2003. The large arrows indicate moisture events >20 cm and small arrows indicate non-monsoon moisture events >6 cm.

Methane

Methane consumption at the soil surface was minimal prior to the start of the monsoon rains in July (Figure 6). Precipitation induced the development of a sizeable CH_4 sink, which was highest during the monsoon season in the open area (-36.35 ± 6.54 μg CH_4 m^{-2} h^{-1}), closely followed by the mesquite (-25.63 ± 7.61 μg CH_4 m^{-2} h^{-1}) and the sacaton (-16.39 ± 5.37 μg CH_4 m^{-2} h^{-1}) sites. Because methanotrophs (methane-consuming microbes) are not particularly xerotolerant (Schnell and King 1996), it was hypothesized that the CH_4 sink would weaken as soils dried in the fall and winter, but surprisingly, the CH_4 sink strength did not disappear with the autumn drying of surface soils (Figure 6). Only when extremely warm and dry conditions prevailed in late spring of 2003 did the CH_4 sink diminish to pre-monsoon levels (Figure 6).

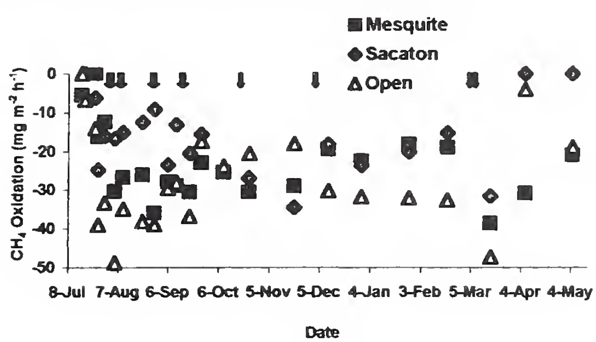

Figure 6. Flux of CH_4 from the San Pedro soils in 2002 and 2003. The large arrows indicate moisture events >20 cm and small arrows indicate non-monsoon moisture events >6 cm.

Statistical analyses showed that neither moisture ($r = 0.0$, $p = 0.99$) nor temperature ($r = 0.02$, $p = 0.32$) was significantly correlated with CH_4 consumption in this semi-arid ecosystem. This result was surprising, given the strong moisture and temperature controls of CH_4 consumption in other ecosystems (Torn and Harte 1996). Laboratory incubations of soils collected in July 2002 from the open system indicate that methanotrophs were most active at the 10-15 cm soil depth (data not shown), agreeing with the close-to-surface maximum in

CH_4 consumption in temperate soils (Koschorreck and Conrad 1993). It is possible, however, that the depth of maximum CH_4 oxidation moves downward in the soil profile as the soil dries after the monsoon season.

Carbon dioxide

CO_2 efflux from the soils under the 3 vegetation types was also low prior to the onset of monsoon rains (27.00 ± 16.93 mg CO_2 m^{-2} h^{-1}). Monsoon moisture input stimulated CO_2 efflux in all 3 vegetation zones, which averaged 241.99 ± 64.30 mg CO_2 m^{-2} h^{-1} in the mesquite site, 240.62 ± 60.74 mg CO_2 m^{-2} h^{-1} in the open site, and 224.90 ± 62.94 mg CO_2 m^{-2} h^{-1} in the sacaton site from July through September. As rainfall decreased and temperatures cooled in the winter and spring, CO_2 production diminished in all three sites, and averaged 49.90 ± 31.87 mg CO_2 m^{-2} h^{-1} in the mesquite site, 37.02 ± 27.83 mg CO_2 m^{-2} h^{-1} in the open site, and 43.63 ± 29.78 mg CO_2 m^{-2} h^{-1} in the sacaton site.

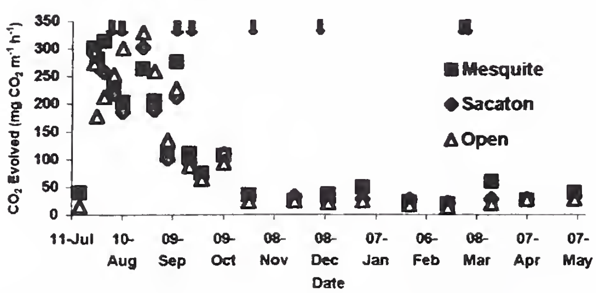

Figure 7. Flux of CO_2 from the San Pedro soils in 2002 and 2003. The large arrows indicate moisture events >20 cm and small arrows indicate non-monsoon moisture events >6 cm.

Monitoring of CO_2, N_2O, and CH_4 fluxes is continuing in the 3 vegetation sites to quantify annual source and sink strengths and to calculate the global mitigation potential of the San Pedro Riparian ecosystem. Preliminary data from the first year of C and N balancing in the San Pedro suggests the existence of a sink of atmospheric C and N in the riparian ecosystem.

Conclusions

The San Pedro riparian zone is a unique oasis in a semi-arid environment. Availability of water in the riparian zone is the driving force that creates this diverse environment. The C and N accumulation in the mesquite under story is mainly due to seasonal moisture limitations that occur in the semi-arid site. Mesquite encroachment in semi-arid systems could significantly impact climate change by creating a sizeable sink for atmospheric C and N. Changes in water availability in the riparian zone, resulting either from climate change or from increased water use in neighboring communities could have tremendously dire impacts on the C and N resources in these plant communities and could thus spell the end of the vibrant San Pedro riparian zone, as has happened in other semi-arid riparian areas in the southwestern United States.

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Table 1. Carbon and nitrogen content and isotope composition of a mesquite, a mesquite-sacaton, a sacaton and a open forb/annual grass communities.

Sample Site (cm)	$\delta^{15}N/^{14}N$	$\delta^{13}C/^{12}C$	Total N	Organic C	C/N Ratio
		-- ‰ --	------- g kg^{-1}	-------	
Mesquite 0-5	7.09	-25.42	4.79	46.53	9.67
Mesquite 5-10	8.25	-22.50	1.63	15.19	9.32
Mesquite >10	9.17	-19.14	0.92	10.55	10.23
Mesquite-sacaton 0-5	8.07	-21.27	3.08	29.90	9.58
Mesquite-sacaton 5-10	8.95	-18.08	1.27	11.34	8.94
Mesquite-sacaton >10	7.26	-16.74	0.77	8.18	9.02
Sacaton 0-5	7.07	-15.96	1.81	17.61	9.68
Sacaton 5-10	6.95	-15.27	1.21	11.61	9.61
Sacaton >10	6.65	-14.49	1.23	13.48	10.93
Open 0-5	9.38	-18.90	0.63	5.83	9.19
Open 5-10	9.34	-20.01	0.58	6.04	10.16
Open >10	8.86	-17.97	0.37	3.65	9.03

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67.

Influences of Alluvial Fans on Upland Watersheds in Central Nevada

John L. Korfmacher, Jeanne C. Chambers

Abstract

Geomorphic, hydrologic, and vegetation processes of upland watersheds in central Nevada are influenced by side-valley alluvial fans. Discontinuous longitudinal stream profiles and spatial variation in stream channel incision are often associated with the alluvial fans. In many cases, groundwater flow is restricted immediately upstream of side-valley fans resulting in elevated water tables and the occurrence of springs and seeps. Riparian vegetation patterns are influenced both longitudinally and laterally with respect to the stream channel, and large meadow complexes often occur upstream of side-valley fans. Restoration and appropriate management of these watersheds requires knowledge of the history and influence of side-valley fans.

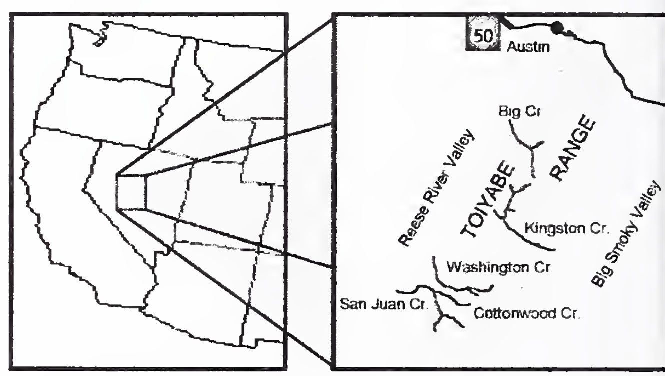
Keywords: alluvial fans, riparian vegetation, stream morphology, groundwater, Great Basin

Introduction

Side-valley alluvial fans ("tributary junction fans" in Meyer et al. 2001) influence stream morphology, groundwater hydrology, and riparian vegetation composition. Here we examine the influence of side-valley fans on these watershed characteristics in central Nevada, drawing primarily upon research conducted within the Great Basin Ecosystem Management Project (GBEMP) over the past 10 years. We define a side-valley fan as one that occurs at the mouth of a tributary drainage (usually ephemeral) and lies on or, in some cases, completely across the valley floor of an axial stream (usually perennial). Considerable literature exists regarding the characteristics and formative processes of alluvial fans (e.g., Bull 1977, Blair and MacPherson 1994), but much of this effort has focused on terminal or "mountain edge" alluvial fans. Less is known about the valley segment or riparian corridor scale effects of alluvial fans occurring within montane watersheds. We present here a summary of fan-related research conducted by the GBEMP and its cooperators in upland watersheds of central Nevada.

Study Area

Most of the research discussed in this paper was conducted in Kingston, Big, Washington, Cottonwood, and San Juan Creeks, on the



Humboldt-Toiyabe National Forest, in central Nevada's Toiyabe Range (Figure 1).

Figure 1. Study area, central Nevada, on the Humboldt-Toiyabe National Forest.

This mountain range, in the basin and range physiographic province of western North America, exceeds 3300 m in height and is bounded on the east and west by intermontane valleys of 1800-1900 m elevation. The climate is semiarid with annual moisture varying with altitude from 25-60 cm (USDA Forest Service unpublished data). The streams are high-gradient, low-flow (0.015 to 0.063 m^3/s base flow) systems with low sediment loads. They flow through valley fill derived primarily from volcanic rocks, although some drainages have a

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significant amount of sedimentary lithology (Kleinhampl and Ziony 1985).
Upland vegetation consists of sagebrush-grass, pinyon-juniper, and mountain brush communities. Riparian vegetation includes woody species (e.g., *Salix* spp., *Betula occidentalis, Populus tremuloides, Rosa woodsii, Artemisia tridentata* ssp. *tridentata,* and other shrubs), sedges and rushes (e.g., *Carex nebrascensis, C. rostrata, C. douglasii,* and *Juncus balticus*), grasses (e.g., *Deschampsia cespitosa, Poa pratensis,* and *P. secunda*), and a number of forbs.

Geomorphic Effects

Benda (1990) and Wohl and Pearthree (1991) investigated side-valley fans formed during storm events in humid and semi-arid regions, respectively. They found that debris fans could temporarily block the axial (trunk) stream, causing ponding and accumulation of a lens of sediment immediately upstream of the fan. Upstream deposits were subject to incision when, days or weeks later, the axial stream breached and subsequently removed all or part of the debris fan. Leeder and Mack (2001) documented this phenomenon ("toe cutting"), which is governed by axial stream power.

Geomorphic effects of side-valley alluvial fans in the upland watersheds of the Great Basin were investigated by Miller et al. (2001). They determined that hillslope erosion and consequent valley bottom and side-valley fan aggradation occurred 2500-1900 years ago as a result of a climate shift from a cool-wet period (the "Neoglacial") to a dry-warm regime. Fans prograding completely across the valley floor temporarily blocked downstream transport of the axial streams' sediment load, resulting in sediment accumulation and local moderation of stream slope. As a consequence of the hillslope erosion and a depletion of available sediments, the stream systems are currently sediment limited and exhibit a natural tendency to incise. In contrast to the above studies, the degree to which the fan deposits have incised depends largely on basin sensitivity to natural and anthropogenic disturbance as governed by factors such as basin geology and morphometry, hydrologic attributes, and sediment storage characteristics (Germanoski and Miller 2003). Watersheds in which the fans are largely incised are underlain primarily by Tertiary volcanic rocks, have large, high relief basins and relatively high stream power. The channels have longitudinal profiles that are relatively smooth, and a high number of semi-continuous terraces. Watersheds in which few of the fans are incised are underlain primarily by sedimentary and meta-sedimentary rocks, are generally less rugged, have lower relief and exhibit lower relative stream power than watersheds in which the fans are largely incised. The fans in these watersheds result in unstable longitudinal profiles that are relatively flat upstream of the fans and significantly steeper where the channels traverse the fans (Figure 2). Relatively coarse alluvium associated with the fans has slowed or prevented incision through many of the fans to date, but the oversteepened segments are inherently unstable.

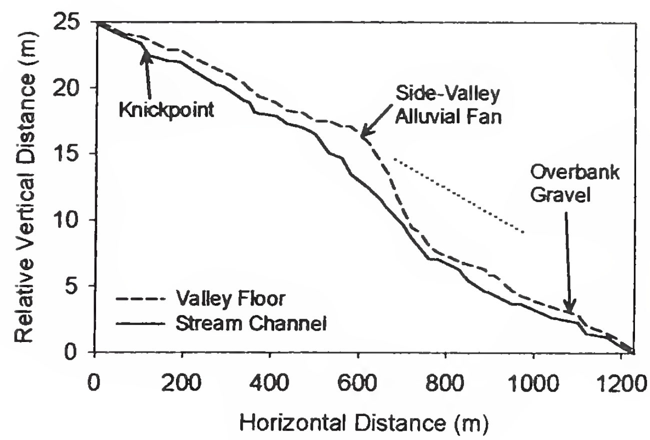

Figure 2. Typical longitudinal profile of an axial stream influenced by side-valley alluvial fans (modified from Miller et al. 2001).

In those basins with relatively unincised fans, Miller et al. (2001) indicated that fans act as local base-level controls, contributing coarse-textured materials to the stream channel in reaches immediately below fans. Channel incision varies as a function of downstream distance from a side-valley fan, with greatest incision occurring where the stream traverses the fan toe, and overbank gravel deposition occasionally taking place immediately above fans. Davis (2000) studied this characteristic in detail, and found that reaches above coarse-textured fans exhibited relatively finer channel particles, little channel incision, and significantly less steep stream slopes than reaches adjacent to alluvial fans. These trends generally were not present for reaches coincident with fans composed of finer-textured materials.

Davis (2000) and Korfmacher (2001) both found that valley floor morphology can be affected by side-valley alluvial fans. Generally, above-fan locations have broad, flat (in the direction perpendicular to the stream) valley floors; below-fan locations had narrower valley floors; in at-fan locations, where recognizable valley floors existed, they were both narrower and more steeply sloped valley floors (Table 1).

Table 1. Valley morphological characteristics by fan position for 53 locations in Big, Kingston, Washington, Cottonwood, and San Juan Creeks, central Nevada. Values are mean +/- standard errors (modified from Korfmacher 2001).

Fan Position	Floor Slope (m/m)	Width (m)
Above	0.0353 ± 0.0035	67.7 ± 5.0
At	0.1193 ± 0.0157	21.0 ± 4.0
Below	0.0365 ± 0.0085	54.3 ± 4.0

Hydrologic Characteristics

Alluvial fans influence hydrogeologic characteristics both within individual fans and at a broader, stream segment scale. Hydrologic characteristics of alluvial fans are important determinants of riparian vegetation community composition (Miller et al. 2001, Chambers et al. 2003) and comprise the link between geomorphic phenomena within a stream system and riparian plant ecological processes and dynamics. Alluvial fans in central Nevada are typically composed of weakly bedded material of varying textures, deposited in units corresponding to climate changes (Ritter et al. 2000, Miller et al. 2001). Hydrologic characteristics of the fans are therefore spatially and seasonally variable, as are the fans' effects on the adjacent riparian zones. Jewett et al. (2003) studied the hydrologic characteristics of a fan-influenced site at Big Creek in central Nevada's Toiyabe Range (Figure 3), using a series of piezometers and observations wells to assess horizontal hydraulic gradients, and nested piezometers to assess vertical gradients. Side-valley fans play several roles in the groundwater characteristics at the study site. They provide a subsurface conduit for groundwater input, influence groundwater depths and hence groundwater availability, and govern groundwater input to the stream.

In the uppermost area of the study site ("A", Figure 3), a small side-valley fan grades out from a tributary watershed. Groundwater recharge from the stream is the dominant feature of the subsurface hydrology in this part of the study site. During the wet season, lateral groundwater flow from the stream occurs and the stream is characterized as a losing reach. However, this characteristic is seasonal. In the summer (dry season) lateral groundwater flow is minimal, although vertical (downward) and down-valley groundwater flow continues throughout the year. The water table on the fan itself is deep (generally 2-3 m). Just downstream of this fan ("B", Figure 3), the water table is shallower, varying seasonally between surface flow and 30-40 cm depth. Further downstream at the lower end of the study site ("C", Figure 3), a larger side-valley fan constricts the stream. This fan influences groundwater flow patterns by restricting the area through which the groundwater may pass. This phenomenon, in concert with the effect of a local bedrock high (Jewett et al. 2003), forces groundwater closer to the surface and produces above-surface flow just upstream of this fan. The riparian zone contains a number of small springs and seeps in this area that remain active throughout the year. Depth to groundwater is shallow with above-surface flow present. The stream is characterized as a gaining reach throughout the year. The vertical hydraulic gradient in this area is predominantly upward, and was continuously upward in a wet year (1998).

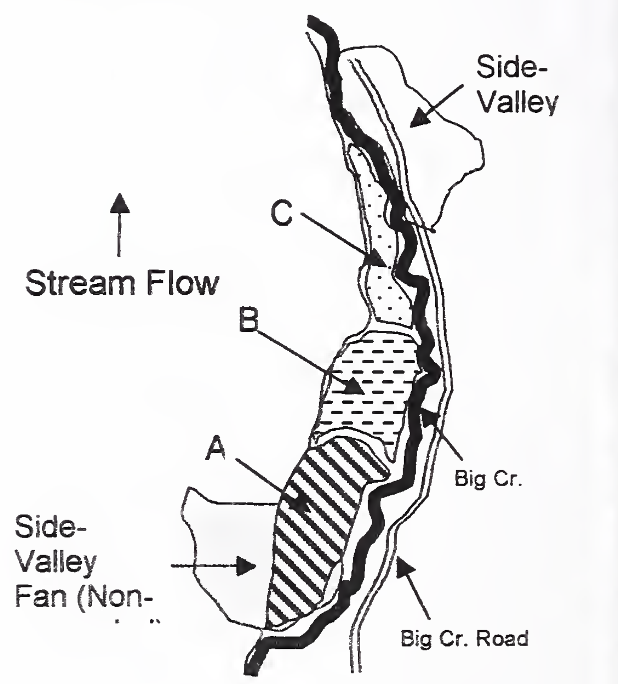

Figure 3. Big Creek hydrologic/hydrogeologic study site. After Jewett et al. (2003).

Vegetation Effects

The composition and abundance of riparian vegetation are influenced by geomorphic position and its influence on flood characteristics (frequency, magnitude, and duration) (Hughes 1997, Tabacchi et al. 1998), and water tables. For near stream vegetation types in central Nevada, median channel particle size (D_{50}), terrace height, width/depth ratio, channel slope, bank particle size (% < 2 mm), incised channel depth, number of terraces and bankfull depth all are related to vegetation type (Chambers et al. 2003). Terrace height, a surrogate for water table depth, is highly correlated with vegetation type. Similarly, in riparian meadow ecosystems, water table depth is often the major factor determining species occurrence (Castelli et al. 2001, Martin and Chambers 2001, Chambers et al. 2003).

Geomorphic factors that are related to vegetation type vary with respect to fan position. Riparian vegetation also exhibits distinct longitudinal patterns in watersheds with non-incised side-valley alluvial fans. Korfmacher (2001) characterized the riparian vegetation associated with side-valley fans in five drainages in central Nevada using high-resolution aerial imagery and a GIS. Figure 4 illustrates vegetation patterns by fan position and lateral distance from the stream. Above-fan locations exhibit a broader riparian zone with a significantly greater percentage of meadow vegetation types (wet and mesic) dominated by woody species, but with a significant presence of upland species. Characteristics of below-fan sites are variable but are similar to at-fan sites.

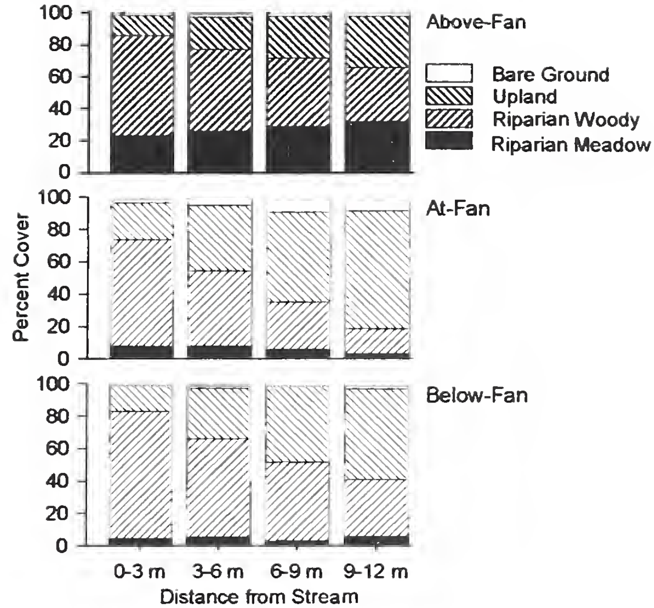

Figure 4. Mean percent aerial cover of riparian zone vegetation stratified by fan position and distance from stream at 53 locations in Big, Kingston, Washington, Cottonwood and San Juan Creeks, central Nevada (modified from Korfmacher 2001).

In watersheds dominated by alluvial fans, headcutting and incision can result in lowering the groundwater table associated with the stream and, consequently, loss of the shallow water tables that are necessary for meadow development and persistence. The fans and their associated riparian ecosystems can be described as metastable. The fans have increased watershed response times to natural and anthropogenic disturbances and preserved the meadow ecosystems, but multiple episodes of incision slowly result in progressive entrenchment as the streams work to smooth their longitudinal profiles.

Synthesis

In fan-dominated watersheds of central Nevada, side-valley alluvial fans have distinct effects on geomorphic processes, groundwater hydrology and riparian vegetation. Many of the riparian meadow complexes in the area are associated with side-valley alluvial fans. Because the stream systems in these upland watersheds are currently sediment-limited, they have an underlying tendency to incise that is being exacerbated by anthropogenic disturbances such as roads in the valley bottoms. This is resulting in progressive incision of the alluvial fans and the loss of valuable meadow ecosystems. Effective management of these watersheds requires an understanding of fan-related geomorphic processes, groundwater hydrology, and riparian vegetation dynamics.

To prevent the continued degradation of riparian meadow complexes associated with side-valley alluvial fans, proactive management/restoration will be required. Geomorphic analyses indicate that the fans tend to be coarser grained and more poorly sorted than upstream or downstream reaches (Davis 2000, Miller et al. 2001), and that the coarse grained nature of the fans has prevented or reduced the rates of incision through the fans (Miller et al. 2001). This suggests that techniques designed to directly increase the stability of the fans without blocking the movement of resident fish or aquatic organisms, such as grade control structures and armoring, may be highly effective. Current research efforts are focused on increasing our understanding the linkages between basin characteristics (morphometry, water yield), alluvial fan characteristics, groundwater systems, and riparian vegetation. This information in combination with knowledge of fan incision processes will be used to prioritize restoration/management activities.

Acknowledgments

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Carbon Dioxide Fluxes on Walnut Gulch Experimental Watershed

William Emmerich

Abstract

Carbon dioxide is increasing in the atmosphere, presumably from human activities. Many soils on Walnut Gulch Experimental Watershed (WGEW) in southeastern Arizona contain carbonates that have accumulated over long periods of time. The hypothesis is that these soils are maintaining this carbon pool under present climatic conditions and are a sink for some of the increasing atmospheric carbon. Bowen ratio systems were used to measure CO_2 fluxes from a brush and a grass community with different soil types on WGEW. Contradictory to the hypothesis, the two sites were found to be losing carbon annually. The brush site with higher inorganic carbon in the soil, had an average annual loss of 144 g C m^{-2} and the grass site 127 g C m^{-2}. Based on measured aboveground biomass data and estimates of belowground biomass, the brush site took up 80 g C m^{-2} and the grass site 135 g C m^{-2} of organic carbon during the growing season. Inorganic soil carbon analysis showed a significant seasonal difference with more in the fall season. The average fall season soil inorganic carbon was 2.24% and the spring season was 1.96% to a depth of 30 cm. This significant seasonal difference indicated some of the measured CO_2 fluxes were into and out of the inorganic carbon pool. The source of carbon for the measured annual losses from these sites was concluded to be from the large inorganic carbon pool with carbon cycling through both the organic and inorganic pools at the sites.

Keywords: carbon dioxide, inorganic carbon, organic carbon, rangeland

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Introduction

Arid and semiarid soils contain large amounts of inorganic carbon (750 to 950 Pg C) that has accumulated through long periods of time in the form of carbonates (Schlesinger 1985, Eswaran et al. 2000). Only the oceanic (38,000 Pg C) and soil organic (1,550 Pg C) carbon pools are larger (Schlesinger 1997). The uptake of carbon through the formation of carbonates requires an arid environment for the precipitation of the carbonate and a source of Ca/Mg from a non-carbonate source. The increasing atmospheric concentration of CO_2 may aid in the formation of carbonates and serve as a sink for some of the carbon being released from human activities.

Arid and semiarid zone soil accumulations and losses of carbonates are controlled by the carbonate-bicarbonate equilibria:

$$CO_2 + H_2O \rightleftharpoons HCO_{3^-} + H^+ \qquad (1)$$

$$CaCO_3 + H^+ \rightleftharpoons Ca^{+2} + HCO_{3^-} \qquad (2)$$

Equation 1 and 2 equilibriums are constantly shifting to the right and left controlling uptake and loss of inorganic carbon to the soil. Under the present climatic conditions, uptake and loss of carbon for the large inorganic carbon pool are mostly unquantified.

Intertwined with the inorganic carbon fluxes are the organic carbon fluxes from plant uptake and decomposition. Separation of inorganic and organic carbon fluxes is problematic, at best. In a tallgrass prairie, it has been estimated that several years of flux data are needed to begin an accurate quantification of grasslands as a sink/source for carbon (Suyker and Verma 2001).

The study hypothesis is that semiarid rangeland soils on Walnut Gulch Experimental Watershed (WGEW) typical of millions of hectare containing a large inorganic carbon pool are maintaining and /or taking up carbon on an annual basis under present climatic conditions. The objectives of this study were to: (1) characterize carbon fluxes in two different semiarid plant and soil communities on WGEW for four years to determine the magnitude of a carbon sink/source; and (2) use seasonal changes in soil carbon and aboveground biomass to relate the measured CO_2 flux to its possible organic and inorganic pool.

Methods

Experimental site descriptions

The two sites for this study are located on the Walnut Gulch Experimental Watershed in southeastern Arizona. Mean annual precipitation is 356 mm and mean annual temperature is 17°C. A brush community site was selected in 1996 in an area known as Lucky Hills (-110° 3' 5" W. 31° 44' 37" N.; elevation; 1372 meters). The dominant shrubs at this site are whitethorn Acacia (*Acacia constricta*), tarbush (*Flourensia Cernua*), creosotebush (*Larrea divaricata*), and desert zinnia (*Zinnia pumila*). The soil at this site is Luckyhills series (Coarse-loamy, mixed, thermic Ustochreptic Calciorthids) with 3 to 8 % slopes. The eluvial parent material for this soil contains many rock fragments of limestone. Surface A horizon (0-6 cm) contained 650 g kg^{-1} sand, 290 g kg^{-1} silt, and 60 g kg^{-1} clay with 290 g kg^{-1} coarse fragments >2mm, 8 g kg^{-1} organic carbon, and 21 g kg^{-1} inorganic carbon.

A grass site was selected in1996 in an area identified as Kendall (-109° 56' 28" W. 31° 44' 10" N. elevation; 1526 meters). Vegetation at the site is predominantly sideoats grama (*Bouteloua curtipendula*), black grama (*Bouteloua eriopoda*), harry grama (*Bouteloua hirsuta*), and lehmann lovegrass (*Eragrostis lehmanniana*), with a few existing shrubs of fairy duster (*Calliandra eriophylla*), and burroweed (*Haplopappus tenuisectus*). The soils at the site are a complex of Stronghold (Coarse-loamy, mixed, thermic Ustollic Calciorthids), Elgin (Fine, mixed, thermic, Ustollic Paleargids), and McAllister (Fine-loamy, mixed, thermic, Ustollic Haplargids) soils, with Stronghold the dominant soil. The eluvial parent material for these soils contains some limestone rock fragments. Slopes range from 4 to 9 %. The Stronghold surface A horizon (0-3 cm) contains 670 g kg^{-1} sand, 160 g kg^{-1} silt, and 170 g kg^{-1} clay with 790 g kg^{-1} coarse fragments >2mm, 11 g kg^{-1} organic carbon, and 7 g kg^{-1} inorganic carbon.

Micrometeorological measurements

Continuous, 20-minute average carbon and water vapor flux measurements were made at both sites using a Bowen ratio energy balance system (BREB) (Model 023/CO_2 Campbell Scientific Inc., Logan, UT, USA). The theory and procedures used to calculate the fluxes has been presented in detail by Dugas (1993) and Dugas et al. (1999). Briefly, atmospheric gradients of air temperature, moisture, and CO_2 were measured every 2 seconds and averaged every 20 minutes. The 20-minute averages were stored in a datalogger (model 21X, Campbell Scientific Inc.). Carbon dioxide, water vapor, and energy fluxes were calculated from the 20-minute average data. Temperature and water vapor gradients were used to calculate Bowen ratios. Bowen ratio, net radiation, soil heat flux, and soil temperature were used to calculate sensible heat flux. Eddy diffusivity was calculated from sensible heat fluxes and temperature gradients and assumed to be equal for heat, water vapor, and CO_2. Fluxes were calculated as the product of the eddy diffusivity and CO_2 and moisture gradients.

Biomass and soil measurements

Centered at each BREB system, radial transects were laid out at a compass direction of every 30° with sample locations at 80, 90, 100, and 110 m. Biomass and soil samples were taken at one sample location on each radial in the spring and fall. Biomass was clipped at the soil surface from 2 m by 2 m plots at the Lucky Hills site and 1 m by 1 m plots at the Kendall site and the material separated into shrub, grass, forb, and litter. The biomass was dried at 65° C, weighed, and amounts were calculated on a kg ha^{-1} basis. Subsamples of each biomass type were ground and analyzed for total carbon by total combustion (CN-2000 Leco Corp. St. Joseph, MI, USA). Total aboveground carbon was calculated from the amount of biomass and carbon concentration data.

Soil samples were collected with a hand spade on the 12 radials adjacent to the biomass plots at depths of 0-15 and 15-30 cm. The samples were air dried, ground to a powder, and analyzed for total carbon by total combustion (CN-2000). Subsamples were analyzed for carbonate-carbon by the pressure-calcimeter method (Nelson, 1982). Organic carbon was calculated as the difference between total and carbonate carbon. A SAS procedure for mixed models analysis of variance (Littell et al. 1996) was used to analyze the soils data for site, year, season, and depth effects and mean separation was determined by the student-t test ($P < 0.05$).

Results and Discussion

Precipitation effects on CO_2 flux

Precipitation was a major influence on CO_2 fluxes at both sites (Figures 1 and 2). Data obtained during precipitation showed numerous events associated with a sudden loss of CO_2, down to the 20-minute time step of the data collection. These releases of CO_2 were due, at least in part, to the dissolution of $CaCO_3$ by low pH (i.e. as low as 3.5 to 7.5) rainfall events known to occur in the area (author's unpublished data) (Equations 1 and 2).

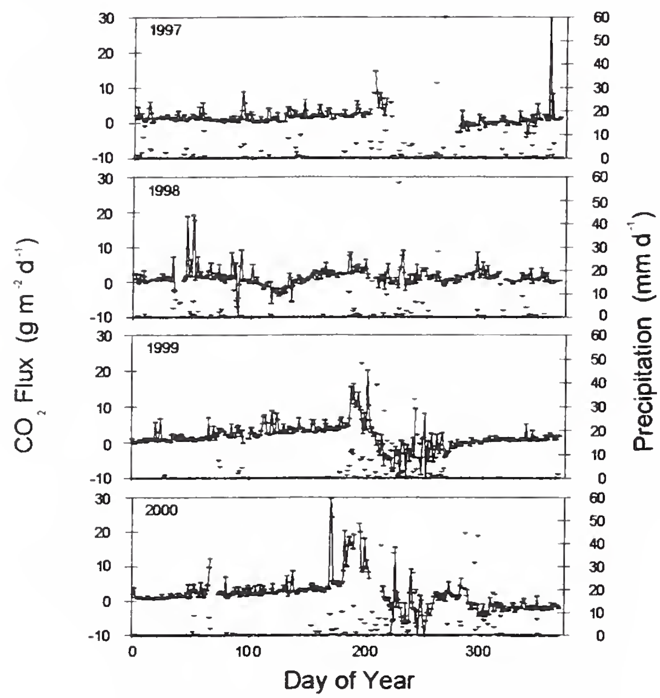

Figure 1. Lucky Hills yearly carbon dioxide flux and precipitation (negative values, indicate carbon uptake).

Acid precipitation would drive the equilibrium for Equation (2) to the right and Equation (1) to the left, thus releasing CO_2 to the atmosphere. Around day 200 for most years at the start of the summer precipitation and growing season, precipitation events large enough to wet the soil profile caused increased CO_2 losses starting 6-24 hours after the rainfall (see Lucky Hills 1999 day 185+ Figures 1 and 3). These types of CO_2 losses have been attributed to increases in microbial activity (Kessavalou et al. 1998). The losses of CO_2 continued until the vegetation started its summer growth period, reversing the CO_2 loss with uptake of carbon into plant biomass. Once the summer growing season started, the carbon uptake rates were generally greater for the Kendall grassland site than for the Lucky Hills brush site (Figures 1 and 2).

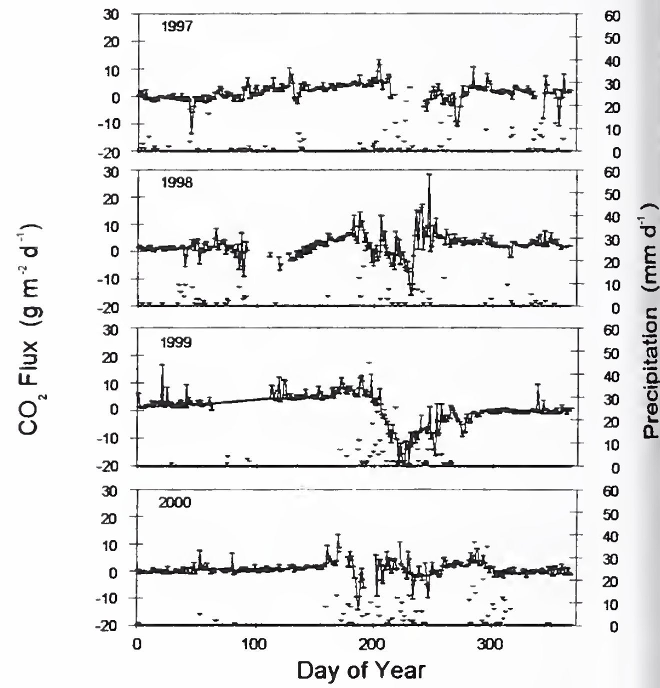

Figure 2. Kendall yearly carbon dioxide flux and precipitation (negative values, indicate carbon uptake; values estimated 1999 day 62-115 and 226-271).

The absence of precipitation throughout the year was another large influence on CO_2 flux. In 1999 and 2000, winter and spring at both sites were unusually dry (Figures 1 and 2). As the soil profile dried and became warmer, the CO_2 loss rate gradually increased until the summer rainy season started. Equilibrium for Equation (1) was shifting to the left by the removal of water from the soil solution causing the release of CO_2, and the equilibrium for Equation (2) was shifting to the left with the precipitation of $CaCO_3$. Soil moisture was exceedingly low at less than 0.02 kg kg^{-1}, hence soil microbiological activity as a large source of the CO_2 during this time was unlikely. With out microbial activity as the source of CO_2, the loss was concluded to be from the large inorganic soil carbon pool.

Precipitation pattern was more typical at the sites in fall 1997 and winter/spring 1998 as contrasted with the dry winter/spring in 1999 (Figures 1 and 2). The soil profiles in 1998 then contained moisture that could be utilized by the vegetation as the temperatures warmed up for plant growth. The vegetation utilized this soil moisture as evidenced by the uptake of carbon starting around day 100 and continuing until soil moisture was depleted. At the Kendall site, sporadic equipment failure prevented continuous measurement of carbon fluxes, but it is clear from the data collected that carbon was taken into plant biomass during this time (Figure 2).

Potential CO_2 flux sources

Carbon dioxide fluxes in and out of the two sites were from the organic and inorganic pools. Separation of the carbon fluxes from the two pools is difficult without the use of carbon isotopes, but the data did provide important insights into the origin of the measured fluxes. The dry winter and spring in 1999 and 2000 provided an opportunity to look further at CO_2 fluxes that likely would originate from the inorganic source as the CO_2 from microbial activity and plant growth would be moisture limited. The 1999 daily fluxes were separated into day and nighttime fluxes (Figures 3 and 4).

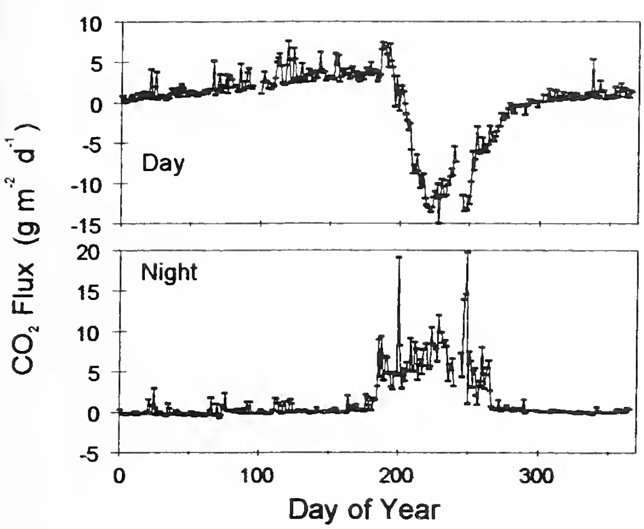

Figure 3. Lucky Hills, 1999 year, day and night carbon dioxide fluxes (negative values carbon uptake).

Nighttime fluxes are respiration fluxes and they remained relatively constant and small until the advent of summer rains, when night fluxes greatly increased. The constant nighttime fluxes during the dormant period were a further indication that there was no detectable increase in soil microbial activity as the soil and air temperatures increased. The increase in nighttime fluxes corresponding to the beginning of the summer rains represented soil microbial activity and plant respiration, hence an organic origin of the CO_2 fluxes (Linn and Doran 1984). The increasing daytime fluxes from winter to spring (i.e. dormant period) would then be from the inorganic carbon pool, controlled by evaporation shifting equilibrium for Equations 1 and 2 to release CO_2.

The growth and decomposition of the plant biomass at the two sites would be sources of organic carbon fluxes. Measured spring and fall aboveground biomass showed there was more carbon in the fall (Table 1). Averaged over 4 years, the annual increase in aboveground biomass carbon in the fall was 28 g C m^{-2} at the Lucky Hills site and 19 g C m^{-2} at Kendall. This represents a part of the organic carbon taken up during the summer growing season (Figures 1 and 2). The total organic carbon taken up was likely greater because some would have gone into below ground biomass. Cox et al. (1986) measured biomass distributions in these same plant communities and determined 65% of the biomass was below ground at the Lucky Hills site and 86% at Kendall. If these same percentages hold for carbon taken up into the aboveground and below ground biomass, the average annual total organic carbon taken up for the Lucky Hills site would be 80 g C m^{-2} and 135 g C m^{-2} for Kendall.

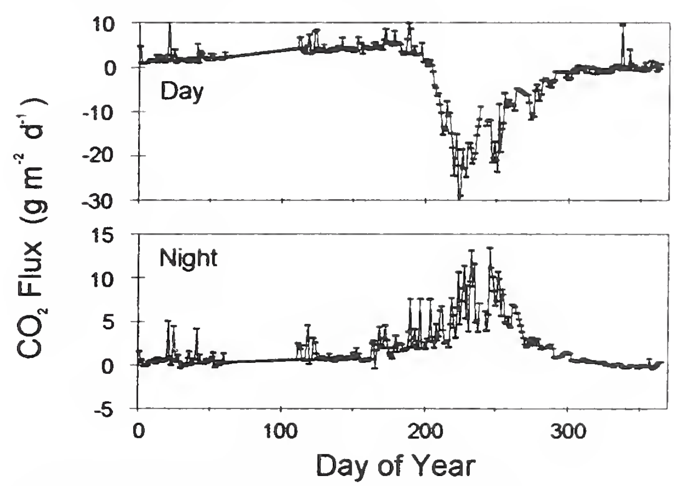

Figure 4. Kendall site 1999 year, day and night carbon dioxide fluxes (negative values, indicate carbon uptake, values estimated day 62-115 and 266-271).

The results of the soil analysis gave additional information on the potential sources of the CO_2 fluxes at the sites. SAS (Littell et al. 1996) analysis of variance results indicated there were significant site, depth and seasonal differences in the soil

Table 1. Yearly spring and fall aboveground biomass for Lucky Hills and Kendall sites.

Year	Spring	Fall
	------------ g C m^{-2} -----------	
	Lucky Hills	
1997	190	170
1998	150	240
1999	210	230
2000	210	230
	Kendall	
1997	90	120
1998	60	80
1999	100	150
2000	90	60

Table 2. Total annual and annual day and night time carbon fluxes at Lucky Hills and Kendall sites for years 1997 through 2000 (positive = source, negative = sink).

Year	Total	Day	Night
	------------------- g C m^{-2} --------------		
	Lucky Hills		
1997	130	70	60
1998	140	10	130
1999	155	5	150
2000	150	10	140
	Kendall		
1997	130	-30	160
1998	210	-140	350
1999	110	-80	190
2000	60	-100	160

inorganic carbon at the two sites. The significant seasonal effect had a mean 22.5 g kg^{-1} inorganic carbon in the top 30 cm of soil in the fall season and 19.4 g kg^{-1} in the spring season averaged over both sites. The seasonal differences imply that some of the measured carbon loss from fall through spring at the sites was from the inorganic pool and that some of the carbon taken up during the summer growing season went into the inorganic carbon pool (Figures 1 and 2).

Annual CO_2 fluxes

Annual CO_2 flux totals showed both sites were losing carbon (Table 2). The four-year average annual loss was 144 g C m^{-2} for the Lucky Hills site and 127 g C m^{-2} for Kendall. The Kendall site during the daytime took up three times the carbon lost from the Lucky Hills brush site on an annual basis. This was due to the higher carbon uptake at Kendall compared to Lucky Hills (Figures 1 and 2). The higher fluxes agree with the estimated higher total biomass accumulations during the growing season at the Kendall site. The average annual nighttime CO_2 flux losses at Kendall were twice those of Lucky Hills. The higher estimated biomass accumulation and higher nighttime CO_2 fluxes suggest there was more carbon cycling through Kendall than through Lucky Hills annually.

The 127 -144 g C m^{-2} average annual loss of carbon from these sites indicates they are a source of carbon to the atmosphere under the present climatic conditions (Table 2). For each of the four years, there was a carbon loss, even for 2000, a year with 30% above annual precipitation. The additional precipitation did not indicate the potential for organic carbon uptake annually. Recent studies indicate that some prairie sites are near equilibrium for organic carbon annually (Frank and Dugas 2001, Suyker and Verma 2001). If the Arizona sites are near annual equilibrium for organic carbon, much of the carbon loss could be from the inorganic carbon pool, which is substantial due to the limestone parent material. An estimation of the inorganic carbon in the soil profile was not made, but soil pits at the sites indicate increasing carbonate with depth (SCS, 1974). This large inorganic carbon pool could easily supply the observed annual carbon flux losses.

Conclusions

The hypothesis was that Walnut Gulch Experimental Watershed semiarid rangeland soils already containing carbonates are still taking up carbon into the inorganic and/or organic pools on an annual basis under the present climatic conditions. Actually, the two rangeland sites with different vegetation and soil types were found to be a source of CO_2 to the atmosphere annually. The carbon source appears to be from the large inorganic carbon pool in these soils. The Lucky Hills brush site with more inorganic carbon in the soil had an average annual loss of 144 g C m^{-2}, while the Kendall grass site, with less inorganic carbon in the soil, had an average annual carbon loss of 127 g C m^{-2}. The significantly greater inorganic soil carbon in the surface 30 cm on the soil in the fall than in the spring, implied that some of the observed CO_2 fluxes during the winter dormant and summer growing seasons cycled through the inorganic carbon pool. Carbon dioxide flux losses during very dry periods suggest that the source was primarily from the inorganic carbon pool. The next area of research is to determine the amount of carbon being lost from the inorganic pool through isotope analysis of the CO_2 fluxes.

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Watershed Networks and Data Management II

Hydrologic Data Processing Technology at the Southwest Watershed Research Center - Past, Present and Future

Carl Unkrich

Abstract

The evolution of automated data processing at the Southwest Watershed Research Center has paralleled the revolution in electronics, computer hardware, operating systems, and software design. The technology has progressed from mainframe computers, keypunch machines and electro-mechanical analog-to-digital converters, through minicomputers and ASCII terminals, to personal computers, digitizing tablets and graphical user interfaces. The data sources, however, have until recently remained the same: mechanical recorders driven by mechanical clocks. The equipment, software, and data reduction practices employed at each stage of this technological progression is reviewed. The impact of technology on the degree of human labor required, on accuracy and precision in the analog-to-digital conversion process, on strategies to address data storage limitations, and on the potential for human error will be examined. Finally, there is a look ahead to some of the challenges presented by new types of data streams; those generated by electronic sensors connected to programmable dataloggers.

Keywords: hydrologic data, data processing, analog to digital conversion

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Walnut Gulch Experimental Watershed (WGEW) Database Support

H. Dale Fox, Scott N. Miller

Abstract

The United States Department of Agriculture's (USDA) Agricultural Research Service (ARS) has developed an active and diverse rangeland research program on a number of different watersheds across the southwestern United States. One of the most significant of these, established in 1954, is the Walnut Gulch Experimental Watershed (WGEW) located near Tombstone, AZ. The Southwest Watershed Research Center (SWRC) in nearby Tucson, AZ was established in 1961 to support the research and data collection at WGEW. Since that time, SWRC scientists have been seeking new and improved methods to more efficiently and effectively utilize rangelands while improving their long-term sustainability. An initial effort required to achieve this objective involved collecting data and developing databases from the watershed that could be used to help understand problems from WGEW and related areas with little or no research data. Early efforts to collect data for database development focused on basic resources such as geology, soils, vegetation, rangeland ecological site, instrumentation, climate, precipitation, watershed boundaries, drainage, and roads. This poster is used to demonstrate how some of the data from these basic databases is currently being used to describe WGEW for education, research, technology development and transfer.

Keywords: rangeland, watershed, hydrology, database, geographic information system (GIS)

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Introduction

The need for hydrologic information has been widely recognized by scientists, legislators, and conservationists for decades. Major data collection and analysis programs have been promoted in many developed countries to facilitate this need.

This paper describes the effort to summarize this data in a spatial Geographic Information System (GIS). The ARS has been conducting research on the hydrologic responses of rangeland watersheds since the agency was established in 1953. To assist the development of a new scientific understanding of hydrologic processes, and to solve water resources problems in the United States, the Administration and Congress invested several million dollars in the expansion of the watershed experimental program. During the 1950s and 1960s some intensively instrumented watersheds were established in different physiographic regions of the United States.

The Walnut Gulch Experimental Watershed WGEW was selected as a research facility by the United States Department of Agriculture (USDA) in the mid-1950s. The Southwest Watershed Research Center (SWRC) in Tucson, Arizona was established in 1961 to administer and conduct research on the WGEW. Research objectives specifically identified for this area include soil, water, and air concerns within a semi-arid rangeland environment (Renard et al. 1993).

The WGEW encompasses approximately 150 square kilometers in southeastern Arizona. The watershed is representative of approximately 60 million hectares of brush and grass covered rangeland found throughout the semiarid southwest. It is considered a transition zone between the Chihuahuan and Sonoran Deserts. Elevations within the watershed range from 1250 m to 1585 m MSL. Livestock grazing is the primary land use with mining, some urbanization and recreation uses also occurring.

SWRC scientists use a variety of basic data collected from within the area to study the hydrology of rangeland watersheds and the effects of changing land uses and practices on the hydrologic cycle.

Methods

Data collected within and about the WGEW can be used and displayed in various ways to describe the watershed for better understanding by scientists for research problem identification and analysis, and for technology transfer of research results to land managers and other users. One way of displaying this information is through the use of GIS technology (Wagenet 1988, Whittaker 1993).

A variety of national, regional, and local data can be used to generate maps and other visual tools using GIS technology including:

- Four Corners Region
- Arizona
- San Pedro River Basin
- Walnut Gulch Experimental Watershed (boundary, contours, flumes, geology, geomorphology, rain gauges, metflux stations, soil moisture plots, streams, stream order, township and range, land ownership, stock ponds, rangeland ecological sites, roads, soils (NRCS - STATSGO, SSURGO), shop location, sub-watersheds, thiessen map, vegetation, utilities, dems, flow accumulation, flow direction, slope, streams, landcover, etc.)

Results

GIS technology is being used to model physical, structural, and functional characteristics of arid and semiarid ecosystems (Coffin and Lauenroth 1989, Burke et al. 1990, Price et al. 1992). A brief description of the results using GIS to display soils and rangeland ecological site data follows.

A soil survey was conducted on the WGEW in 1965-1966 and updated in 1993-1994 by the Soil Conservation Service (SCS), with soil names and descriptions being approved in 1967. The survey was conducted through a request made by the Hereford and Whitewater Draw Soil Conservation District for the Agricultural Research Service. During this survey, soil boundaries were outlined on a map and associated with a specific mapping symbol (USDA SCS 1993). The fundamental unit of the soil survey is the soil series, the basic soil taxonomic unit from which the rangeland ecological site is built. The rangeland ecological site represents a combination of different soil series with similar characteristics. The soil series that comprise a rangeland ecological site are often associated by location, texture, etc. and other soil properties.

The U.S. Department of Agriculture Natural Resources Conservation Service (NRCS) has established the National Soil Information System (NASIS) that includes three geographic databases representing different intensities of mapping (Soil Survey Staff-NRCS 1997). The State Geographic database (STATSGO) has been developed at a scale of 1:250,000 and archived in one by two degree topographic quadrangle units (Figure 1). Soil Survey Geographic (SSURGO) maps are made at scales ranging from 1:12,000 to 1:63,360 and digitized so that they duplicate the original county soil survey maps (Figure 2). These data are archived in 7.5-minute USGS topographic quadrangle units and are patched together to create county versions of SSURGO.

The National Soil Geographic (NATSGO) database is the most general geographic database. It contains digital data developed nation-wide on a scale of 1:7,500,000. The database consists of spatial data, such as the digital MLRA map and attribute data, including data on map unit components and composition that are derived from the STATSGO file.

GIS data layers were created for each of the soil survey mapping symbols (Figure 3) and used to generate a watershed map that displays the soil boundaries within the WGEW.

A rangeland ecological site survey was conducted by the SCS in 1965-1966 in conjunction with the soil survey. Soil mapping units, vegetation, and site attributes were used to determine boundaries of the sites (USDA SCS 1997). The rangeland ecological site is the basic unit of a hierarchical natural resource classification system. A rangeland ecological site is a distinctive kind of land that differs from others in its ability to produce a characteristic natural plant community. Rangeland ecological sites are taxonomic units, not mapping units. However, when a rangeland ecological site occurs over a relatively large area, the site can be designated as a mapping unit, even on relatively small-scale imagery.

Figure 1. STATSGO provides minimum definition of soils within WGEW due to large scale.

Figure 2. SSURGO is widely used due to detail shown within watersheds.

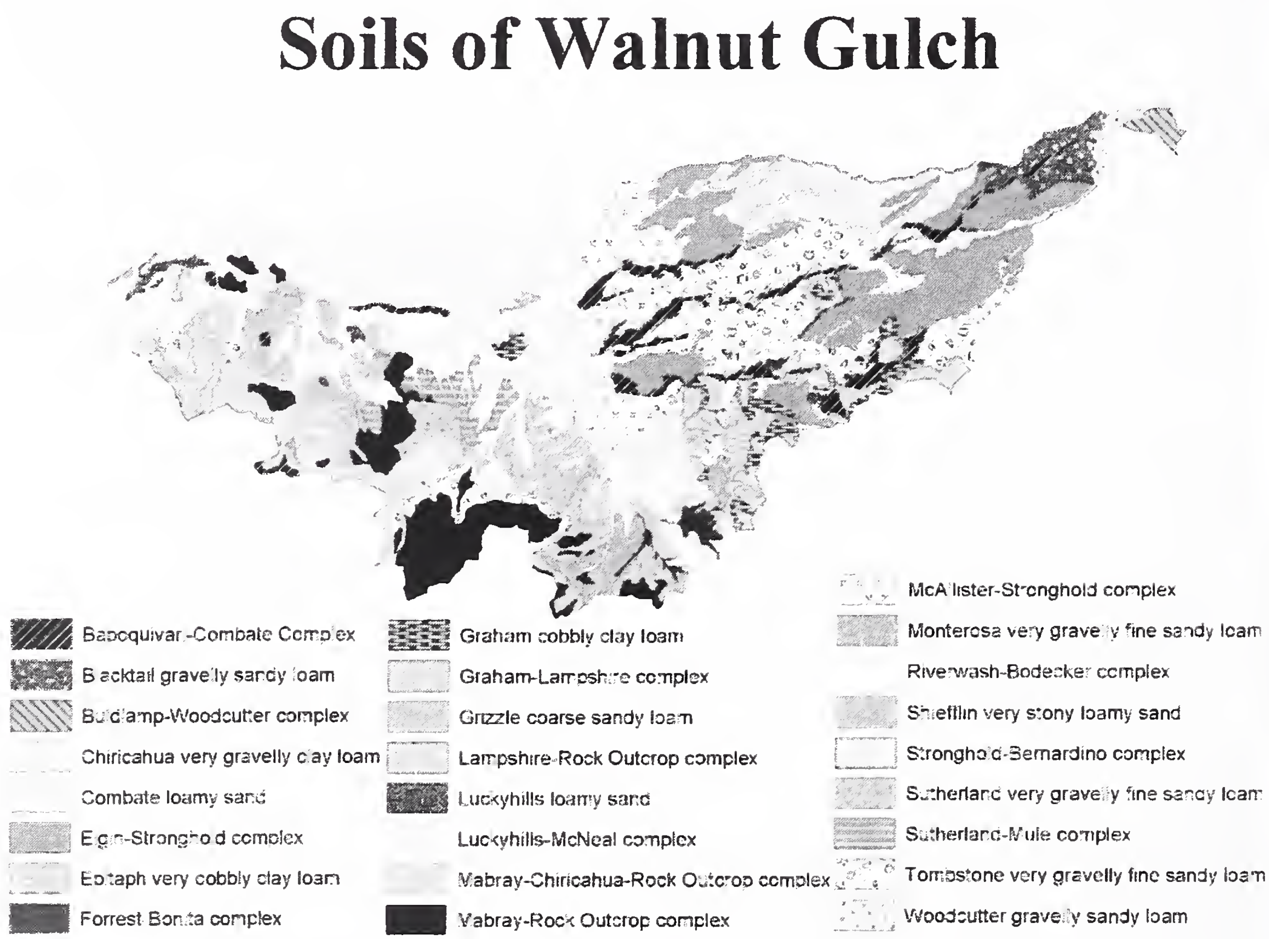

Figure 3. WGEW with soil mapping unit boundaries delineated.

A land classification system developed by USDA and described in Agricultural Handbook No. 296 (USDA, 1981) divides the United States into several hierarchical categories based on natural resource characteristics and land uses. Land resource categories, as described in this handbook, used at national levels are Land Resource Units (LRU), Major Land Resource Areas (MLRA), and Land Resource Regions (LRR). The information provided in this handbook affords resource managers a basis for making decision about national and regional agricultural concerns, identifies needs for research and resource inventories, provides a broad base for extrapolating the results of research within national boundaries, and serves as a framework for organizing and operating resource conservation programs.

GIS data layers were created for each of the rangeland ecological sites (Figure 4) and used to generate a watershed map displaying the site boundaries within the WGEW.

As an illustration, characteristics of MLRA 41 (Southeastern Arizona Basin and Range), which covers parts of Arizona and New Mexico, are given in the following example: each LRR is comprised of a number of MLRAs, which in turn are comprised of a number of LRUs, which are made up of multiple rangeland ecological sites. The following example shows one of the three Resource Units within LRR d, MLRA 41, and several rangeland ecological sites within the Resource Unit as used in Arizona.

-LRR D-Western Range and Irrigated Region
- -MLRA 41 Southeastern Arizona Basin and Range
  - -Land Resource Unit 41-3 Southeastern Arizona Semi-Desert Grassland
    - -Ecological (Rangeland) Sites
      - -Loamy Upland 30-40 cm annual precipitation

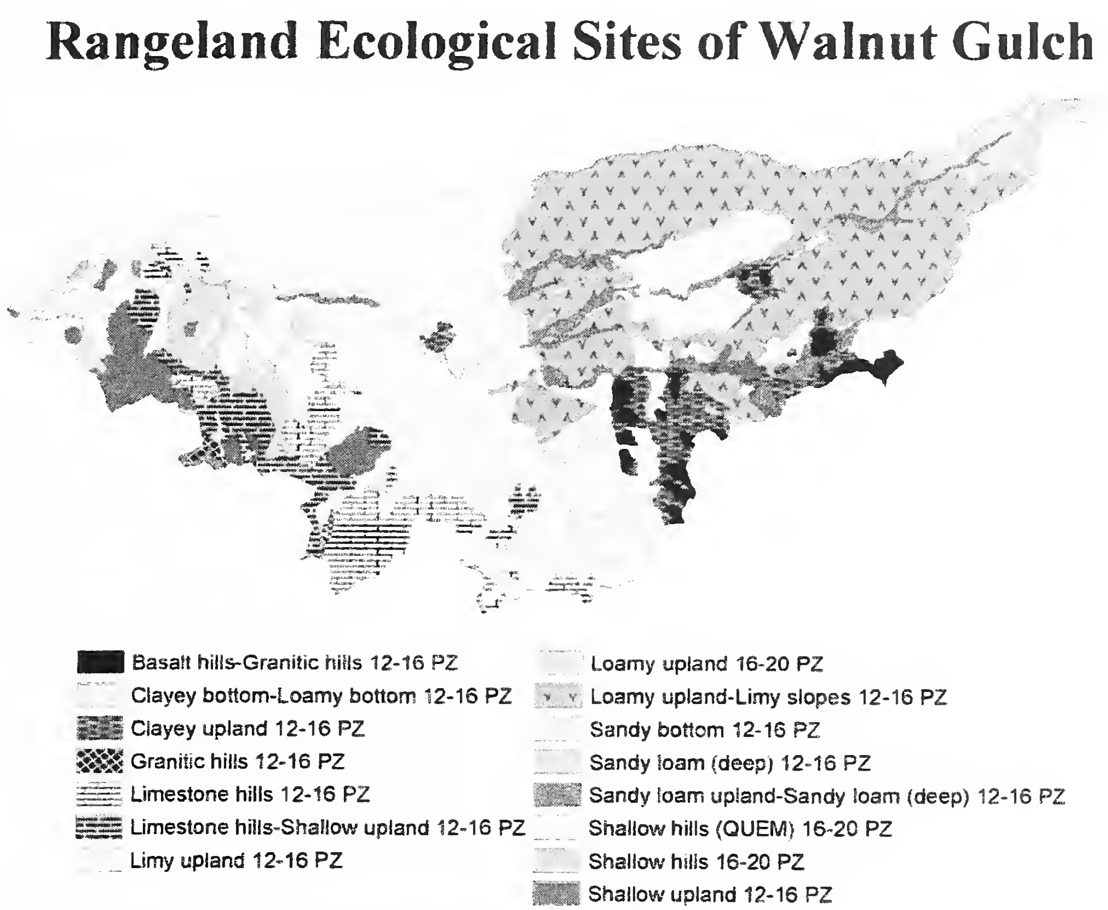

Figure 4. WGEW with rangeland ecological site boundaries delineated.

Computer and information management technologies have the potential to link together the incremental improvements from a large number of new technologies and knowledge (Holt and Rawlins 1990). Due to the volume and variety of pertinent data, efficiency of information systems is a major factor determining success in planning and implementation of an agency's management strategies (Loh and Power 1993). The data required in all natural resource management fields is naturally voluminous and heterogeneous. Creation of regional and national databases is imperative for model development, testing, and application. For example GIS can be particularly valuable in assessing temporal and spatial responses to management.

Conclusions

Due to the data requirements needed to advance system understanding, and the expense of data collection activities, more development and use of computer technology must be used. GIS can be a substantial help in bridging this gap between the data collection effort and the users of this data for technology development and transfer.

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Turbidity Threshold Sampling in Watershed Research

Rand Eads, Jack Lewis

Abstract

When monitoring suspended sediment for watershed research, reliable and accurate results may be a higher priority than in other settings. Timing and frequency of data collection are the most important factors influencing the accuracy of suspended sediment load estimates, and, in most watersheds, suspended sediment transport is dominated by a few, large rainstorm events. Automated data collection is essential to effectively capture such infrequent events. Turbidity Threshold Sampling, a method that distributes sample collection over the range of rising and falling turbidity values during each significant turbidity peak, has been used since 1996 at the Caspar Creek Experimental Watershed in northern California.

Keywords: suspended sediment, turbidity, automatic sampling, water quality

Introduction

The Caspar Creek Experimental Watershed is located on the Jackson State Forest, in northwest California, approximately 15 km southeast of the city of Fort Bragg. The 897 ha study area, located about 7 km from the Pacific Ocean, encompasses the North and South Forks of Caspar Creek. The topographic development consists of uplifted marine terraces that are deeply incised by coastal streams (Henry 1998). The Mediterranean climate is typical of low-elevation watersheds on the Pacific coast where the fall and winter seasons are moist with low-intensity rainfall and persistent cloud cover. Snow rarely occurs because of the moderating effect of the nearby Pacific Ocean. The mean annual rainfall from 1962 through 1997 was 1,190 mm. Prior to the 1860s the forest was composed primarily of old-growth coastal redwood. Current second- and third-growth forest stands are primarily coastal redwood and Douglas-fir, with a small component of hardwoods.

A primary focus of research on the Caspar Creek Experimental Watershed is to evaluate management activities and forest practice regulations on the production of sediment from watersheds. It is difficult to identify the causes of erosion within a watershed because factors such as increased subsurface flow and loss of root strength are difficult to observe (Lewis 1998). In addition, when naturally occurring landslides and erosion from historic land use combine with recent management activities the controlling factors become complex and intertwined. The erosion research in Caspar Creek has relied on the paired-watershed design. The watersheds were chosen because they are physically close together, have similar soil types, and rainfall tends to be spatially uniform in volume and intensity. The studies on suspended sediment in Caspar Creek have relied on comparing data from treated and untreated watersheds that are measured before, during, and after treatment. Gaging structures, either weirs or flumes, are typically installed at the base of the selected watersheds to measure discharge, and suspended sediment samples are collected primarily during storm events.

Suspended sediment data collection is challenging in many environments because most of the annual suspended sediment load is transported during a few large storm events when it may be difficult or impractical to collect an adequate number of samples (Lewis and Eads 2001). Water discharge is often a poor predictor of suspended sediment concentration (SSC) when sediment inputs to the channel are highly episodic. An efficient form of automated sampling is desirable that distributes sample collection based on a suitable real-time surrogate for suspended sediment. The relation between turbidity and SSC is quite good for most

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streams, particularly when changes in particle size during storm events are minimal. Although turbidity cannot replace SSC, it can be a valuable aid in deciding when to collect physical samples. Recent advances in turbidity sensors and mounting configurations now permit reliable and continuous in-stream monitoring of turbidity.

Methods

An approach called Turbidity Threshold Sampling (TTS) utilizes a programmable data logger, turbidity sensor, and automatic pumping sampler to collect SSC samples at user-specified turbidity thresholds (Lewis 1996). The resulting set of samples can be used to accurately determine suspended sediment loads by establishing a relationship between sediment concentration and turbidity for any sampled period and applying it to the nearly continuous turbidity data. A distribution of turbidity thresholds that provides adequate sampling for load estimates during small events, but does not over-sample during large events, can be constructed by evenly spacing the square roots of the thresholds to cover the range of the turbidity sensor. Because a larger portion of the sediment discharge occurs during the prolonged recession, more thresholds are required during the falling than the rising portion of the turbidigraph. Although turbidity is a better surrogate for SSC than discharge, errors in turbidity records are more common and can arise from fouling or inadequate flow depth. A set of rules in the TTS algorithm attempts to prevent excessive sample collection by accounting for short-term spikes in turbidity from passing debris, while recognizing valid ephemeral spikes from sediment inputs (Lewis 2002). Spikes in the turbidity record that are a result of fouling, and that are accompanied by a physical sample, can be recognized when the SSC in question is not elevated but in general agreement with surrounding SSC values. Fouling of the turbidity sensor's optics by sediment or organisms can be identified on the turbidity plots as a gradual but unexpected increase in turbidity. In very large storm events, turbidity values may exceed the sensor's range, and in this case, the TTS algorithm will attempt to collect fixed-time samples until the turbidity returns within the sensor's range. Since this condition could exhaust the pumping sampler's available bottles, it is important that a field crew visits the station and exchanges the pumping sampler bottles. On average, we expect to collect about eight samples per storm for each station once the equipment installation is satisfactory and the sampling parameters are correctly set.

The TTS program resides in a programmable data logger, either a Campbell CR10X or CR510, and interrogates a turbidity sensor and pressure transducer at 10-minute intervals (mention of trade names in not an endorsement by the USDA Forest Service). When the sampling rules are satisfied, a signal is sent to the automatic pumping sampler to collect a sample.

Stage and discharge

Two of the TTS gage sites in Caspar Creek are fitted with a compound 120° V-notch weir, and 19 sites have either a Parshall or Montana flume to measure discharge. The flumes are reasonably efficient at maintaining sediment suspension through the throat, with the exception of coarse bed material that can settle on the flume floor during falling stages, requiring subsequent manual removal. The unimpeded passage of sediment is important when tracking sediment routing in nested watersheds. Ultimately, all of the bed load, and nearly 40% of the suspended load, is deposited in the weir debris basin at the bottom of the watershed. The sediment accumulation is measured annually and the basin sediment is excavated every 5 to 7 years. A flume or weir eliminates the requirement to develop and maintain a discharge rating for each station, and when properly sited, they provide a stable discharge record over time. Flume sizes in Caspar Creek were chosen to accommodate 100-year peak flows. None of the gage site structures, with the exception of the weirs, is grouted to bedrock. Although all flumes have cutoff walls extending into the alluvial bed, an unknown amount of the flow is not captured. Minor discharge errors are acceptable for sediment transport research because most sediment is transported during storm flows, when leakage is proportionally negligible.

Although pressure transducers are deployed at all the Caspar Creek gage sites to measure stage, with minor changes the TTS program could accept other stage measurement technologies. With the exception of the QUE station, all pressure transducers are mounted in stilling wells that are connected to the flume or weir pond. Because the TTS program computes the mean of 150 stage readings in three seconds, the use of a stilling well is not always required to achieve the desired accuracy because fluctuations in water surface elevations are electronically dampened. For example, at station QUE, the pressure transducer is mounted in conduit with the opening at right angles to the direction of flow. The TTS algorithm uses stage to determine when the flow is deep enough to adequately submerge both the pumping sampler intake and the turbidity sensor. Below this minimum stage, no sample collection is attempted.

Turbidity sensor

Collecting reliable turbidity information depends on the placement of the turbidity sensor in the stream, its mounting configuration, optical sample volume, and its ability to remove fouling on the optical surface by a mechanical wiper or other automated means. In our experience, mounting the turbidity sensor near the thalweg and approximately mid-depth in the water column, provides the most reliable measurement location. Mounting the sensor near the bed can increase noise in the record when bed material becomes mobilized at higher flows. Mounting the sensor close to the water surface can produce unacceptable records because of air entrainment, floating debris, sunlight, or emergence of the sensor from the water (Eads and Lewis 2002). An articulated sampling boom, mounted on the bank, bridge, or cableway (Figure 1), can limit the amount of debris that is trapped near the turbidity sensor.

Figure 1. Montana flume with turbidity sensor housing suspended downstream on cable-mounted boom at station Porter in the South Fork of Caspar Creek.

The articulated boom is self-cleaning when large debris accumulates on the leading edge causing it to rise towards the water surface and release the debris. When the sensor is mounted on an articulated sampling boom, field personnel can raise the boom and access the sensor for inspection or cleaning at any flow. In most streams, the turbidity record is improved when the sensor automatically cleans the optical surface to remove fine sediment or macroinvertebrates before each measurement.

We have experimented with a number of turbidity sensor housing designs and have found that mounting the sensor on the downstream edge of the boom, aligned with the direction of flow, produces the most reliable records. The sensor's optics are aimed across the flow and clear of any obstructions. Maintaining a housing length of 30 cm, or more, from the boom reduces the likelihood that debris on the boom will be viewed by the sensor.

Automatic pumping sampler

For watersheds that have a rapid hydrologic response, automatic pumping samplers provide a way to collect unattended samples during important events. Their application is limited to streams and rivers that transport mostly fine sediments, where the height from the water surface to the sampler is less than about 6 m, and where the intake line can be routed so that there are no dips or horizontal runs. Reducing the length of the intake line improves sampling efficiency and decreases power consumption. At Caspar Creek, the sampler intake is mounted in the downward sloping floor of the flume throat and projects horizontally into the flow. At other Caspar Creek installations, the intake is fixed at 9.14 cm above the bed, or mounted on the weir wall at the base of the V-notch, or on the sampling boom. Since samples collected in this manner are point samples they may not be representative of the instantaneous cross-sectional average SSC in cases where the sediment is not adequately mixed. We collect a suite of simultaneous depth-integrated and point samples to correct for bias in SSC data.

Example

The utility of TTS is illustrated by an example (Figure 2) from station Ogilvie on a South Fork tributary draining about 19 ha. The peak flow of $0.133\ m^3s^{-1}$ has a recurrence interval of between once and twice per year. The turbidity spiked sharply several times during the 9-hour rising limb of the event, triggering seven pumped samples. In the 39 hours following the peak, there were a few small turbidity spikes, including a jump of 100 NTU just before 11:40 pm on Feb. 20. Recession limb spikes such as this are often indicative of bank failures, and sometimes can be tracked at downstream stations.

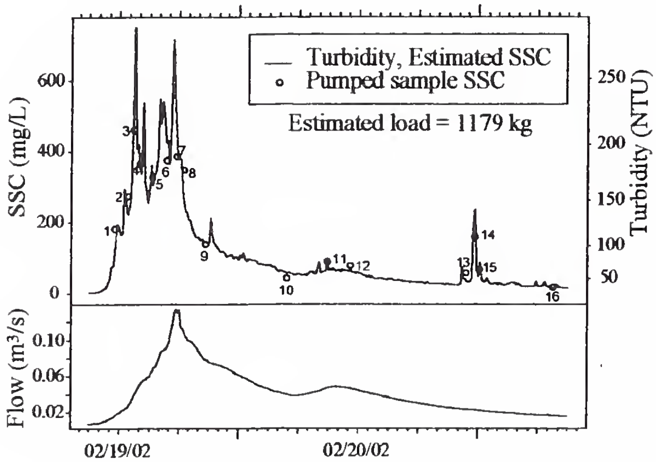

Figure 2. Turbidity, pumped sample SSC, and flow for a storm event at station Ogilvie in the South Fork of Caspar Creek.

In a log-log plot of SSC versus turbidity (Figure 3), samples 2-4 confirm the first spike, samples 13-15 confirm the last spike, and the overall low scatter (r^2=0.94) suggests that the sensor was functioning normally throughout the event.

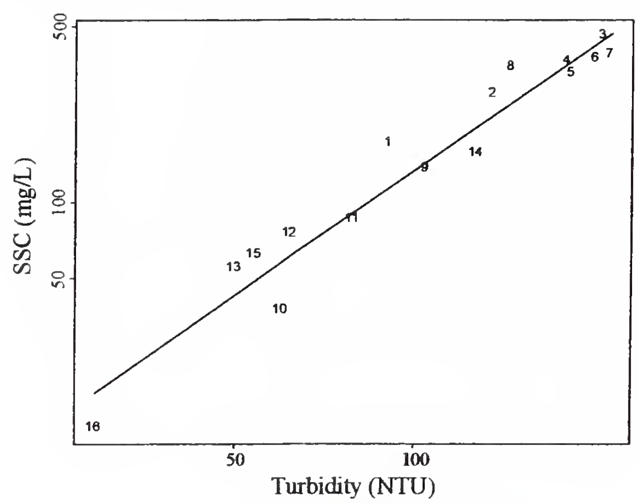

Figure 3. SSC versus turbidity for pumped samples collected at station Ogilvie during event of Figure 2.

Without the physical samples to verify the turbidity fluctuations, it would have been impossible to ascertain that the spiking was not caused by fouling of the sensor. By comparison, a log-log regression of SSC versus discharge (Figure 4) is very poor (r^2 = 0.31). The 95% confidence limits for sediment discharge are 690 and 3078 kg based on the relation in Figure 4, compared to 1010 and 1349 kg based on the relation in Figure 3. A sampling program based on discharge or fixed-time intervals would likely have missed most of the sediment spikes in this event. In any case, without the continuous turbidity record, little or nothing would have been revealed about the duration or shape of the sediment pulses.

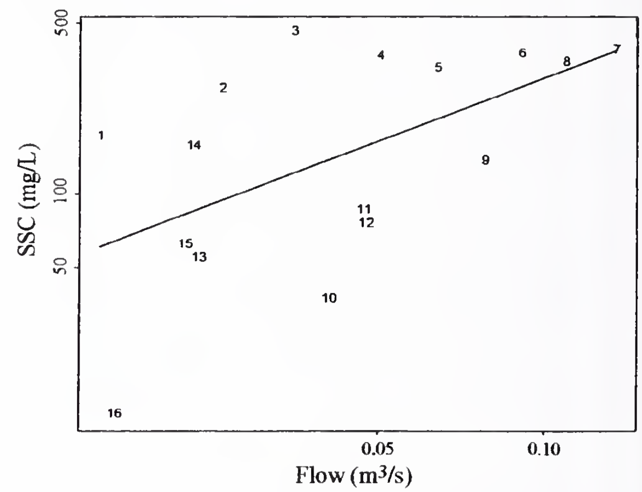

Figure 4. SSC versus flow for pumped samples collected at station Ogilvie during event of Figure 2.

Conclusions

Turbidity Threshold Sampling is an efficient and proven method for accurately measuring suspended sediment loads in rivers that transport mostly fine sediment. The quality of the continuous turbidity record is dependent on the mounting of the sensor and depth of flow at the measurement location. Very small drainages present a considerable challenge because shallow flow depths, during the start and end of the hydrograph, may be inadequate to submerge both the turbidity sensor and the pumping sampler intake. Small watersheds in the Pacific Northwest have steep channels that produce turbulent flow that can result in unacceptable noise in the turbidity record from air bubbles if suitable measurement locations are not available. As the drainage size increases, the deployment of the instrumentation becomes simpler and the quality of the turbidity data improves. However, very large channels present another set of problems, related to the use of pumping samplers. The sampler must be no more than about 6 m above the water surface. In addition, point samples in large rivers may not be easily correctable to a cross-sectional mean SSC.

Using an articulated sampling boom and an appropriate sensor housing can reduce fouling of the turbidity sensor by organic debris. However, fouling may become chronic if the material on the leading edge of the boom is long enough to wrap around the boom and interfere with sensor's optics, or if the force of the flow is not sufficient to allow the boom to rise to the water surface and allow the debris to pass underneath the boom. Most turbidity records are improved when a turbidity sensor with a self-cleaning mechanism, such a mechanical wiper, is used that reduces fouling of the optics from algae, fine sediment, and macroinvertebrates.

Acknowledgments

We wish to thank the California Department of Forestry and Fire Protection for its cooperation and support of the Caspar Creek Experimental Watershed since 1962. The Pacific Southwest Research Station and California Department of Forestry and Fire Protection have entered into a memorandum of understanding to continue watershed research in Caspar Creek for an additional 100 years.

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Use of the ARS Watershed Network for Developing and Validating Models

Jeffrey G. Arnold, R. Daren Harmel, Clarence W. Richardson

Abstract

The ARS experimental watersheds are a unique source of data for developing and validating field and watershed scale models. Long term weather, flow and sediment data have been collected at several ARS watersheds for nearly 70 years. Other advantages of the ARS watershed network are: 1) availability of input data for modeling applications including detailed soils, land use and management, 2) dense weather gages, 3) nested subwatersheds, 4) continuous subdaily flow and sediment, and 5) watersheds scattered across the U.S. in varying hydrological and ecological regions. In addition to continuous flow and sediment data, water quality data are also collected at several watersheds. The water quality data include carbon, nitrogen, phosphorus and pesticides. In this paper, several examples are given showing how the Soil and Water Assessment Tool (SWAT) model has been refined and validated using ARS watershed data. The examples include: 1) phosphorus runoff from poultry litter application fields at Riesel, Texas, 2) flood control structures at Chickasha, Oklahoma, 3) riparian zones in Tifton, Georgia, and 4) surface cracking of clay soils in Riesel, Texas. SWAT is a watershed model that is being used for developing best management practices for pollution control and in U.S. EPA's Total Maximum Daily Load (TMDL) program.

Keywords: watershed models, validation, best management practices

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Distribution and Application of Research Watershed Data

Mark S. Seyfried

Abstract

Research watersheds administered by the Agricultural Research Service (ARS) have collected valuable and unique hydrological data for more than 35 years. This paper describes the effort of one watershed to distribute that data to the public. A few examples of potential applications related to characterization of stream flow, sediment discharge and stream temperature are provided.

Keywords: hydrology, database, sediment, stream temperature

Introduction

The USDA-ARS currently maintains 13 research watersheds across the United States. ARS watersheds are distinct from other research watersheds in that land management on the watersheds is representative of regional agriculture and the hydrology therefore reflects management impacts. Although each watershed location is unique and conducts research specific to the local environment, they have in common intensive monitoring of hydrologic variables over many (in most cases >35 years) years.

The research watersheds may be thought of as outdoor laboratories for conducting hydrologic research. One product of these watersheds that has potential scientific and management applications is the database that results from long term, intensive monitoring. In many cases ARS watersheds are the only source of small order watershed data in the region.

One of the challenges confronting the watersheds is the dispersal of "clean", comprehensive data sets. By clean, we mean data that has been scrutinized for errors and, in some cases, made temporally complete, usually using correlation procedures. This step is tedious, time consuming and requires considerable expertise in both the instrumentation and local hydrology. Given the volume of data collected each hour and shifting personnel, data can easily be acquired much faster than it is processed. The result has been that, while ad hoc clean data sets are available upon request, each dataset requires a special effort and is rarely comprehensive. In addition, the true scope of potential data is rarely appreciated outside the research unit.

Currently there is an effort in the ARS to address this problem. In this paper we describe the approach taken by the Northwest Watershed Research Center, located at Boise, Idaho, which manages the Reynolds Creek Experimental Watershed (RCEW). We published a series of data reports describing the watershed database and made that data available electronically. In this paper we briefly describe those data reports. More information is available in the cited references. We then describe some examples of how experimental watershed data may be of value in addressing management concerns.

Reynolds Creek Experimental Watershed

The Northwest Watershed Research Center and RCEW were established in 1960 to develop fundamental information on runoff and erosion in a region where much of the available streamflow is derived from snowmelt, rain on snow and rain on frozen soil in rangelands. The 234 km^2 watershed is in the Owyhee Mountains 80 km southwest of Boise, Idaho, and is representative of snow-dominated high-relief, mixed-vegetation rangeland watersheds of the interior Pacific Northwest and northern Great Basin. Elevation ranges from 1098 m to 2254 m msl. Annual precipitation ranges from less than 250 mm at lower elevations to over 1000 mm at higher elevations, where 75% of annual precipitation is snow. Rangeland plant communities are dominated by sagebrush, bitterbrush, encroaching juniper, and perennial and annual grasses and forbs. Douglas fir and aspen stands are found in high-elevation snow accumulation areas. Land

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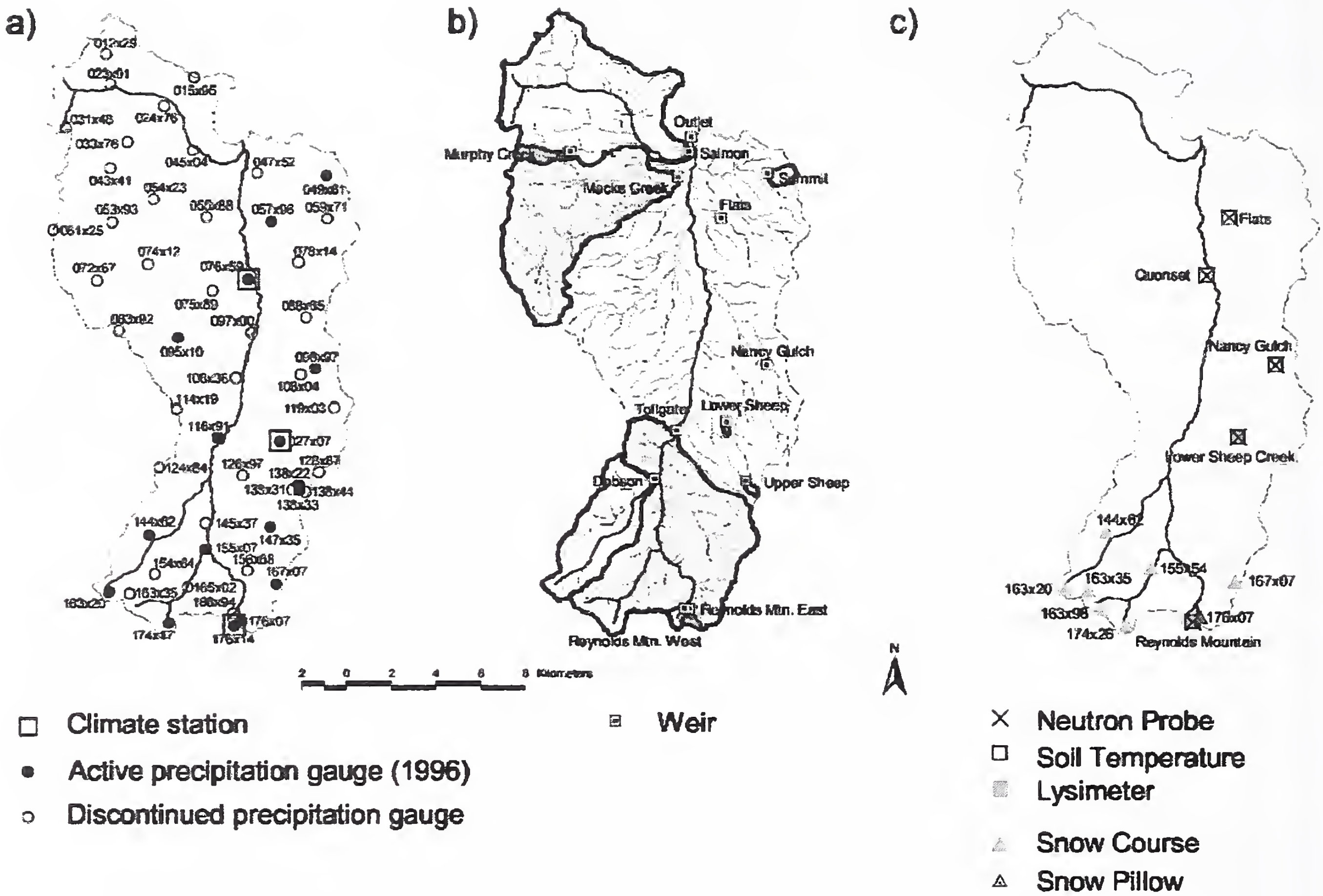

Figure 1. Location of primary instrumentation presented in the data reports.

ownership is 23% private and 77% public. All public lands are under grazing allotment. Hay production and grazing are primary uses of private lands. Wildlife (including a wild horse herd) and year-around recreation are increasingly important resource uses. All research is conducted within the context of a landscape managed under practices common to western resource management agencies and private producers.

Data reports

Instrumentation at the watershed is constantly changing. The published data reports represent the network and data collected between 1961 and 1996 (Slaughter et al. 2001). As of 1996, the RCEW network included 18 dual-gauge precipitation sites, three comprehensive climate stations, seven snow courses and nine stream gauging stations (Figure 1). Data are linked to a relational data base and archive in Boise via real-time radio telemetry with automated data quality checking protocols.

The methodology for gathering data has gone though a vast change since the establishment of the RCEW. Data recording has evolved from spring-driven chart recorders, which were reduced and digitized by hand, to the design and construction of custom data loggers and telemetry systems, to the current use of standard commercial data loggers, instrumentation and telemetry. This has significantly reduced the time and cost of maintaining the database and allowed us to take full advantage of the technological advances in electronics during the last decade.

There are eight published data reports that describe the data collected. Data are presented, along with tables showing the period of record, number of measurement sites, methods used, and examples of how each data set has been used in publications over the years. These are summarized, excepting the geographic database, which is in a different format, in Table 1.

Table 1. Summary of data presented in the data reports.

Data Report	Parameter Measured	Number of stations Maximum	1996	Years of Record	Sampling Interval
Precipitation	shielded precipitation	53	17	1962-1996	breakpoint
	unshielded precipitation				
	calculated precipitation				
Snow	snow course SWE	8	8	1961-1996	biweekly
	snow pillow	1	1		15 min
Daily Climate	T_{max} and T_{min}	3	3	1964-1996	daily
	pan evaporation	3	3	1974-1996	15 min
Continuous climate	humidity				
	solar radiation				
	wind speed, direction				
	barometric pressure				
Soil Lysimeter	lysimeter water	4	0	1976-1991	hourly
Neutron Probe	soil water content	18	14	1970-1996	biweekly
Soil Temperature	soil temperature	5	5	1981-1996	15 min
Discharge, Sediment	suspended sediment	3	3	1965-1996	event based
	stream discharge	13	8	1963-1996	breakpoint, 15 min

In the geographic data report (Seyfried et al. 2001a) both spatially continuous and spatially discrete site location data are presented for all parameters described in the reports. The spatially continuous data are georeferenced to a 30 m digital elevation model (DEM) UTM projection derived from U.S. Geological Survey contours. In addition to basic topographic data from the DEM, this report includes watershed boundaries, stream channel locations, geology, soils, vegetation, land ownership, and road network data. These are available as 14 separate layers.

The precipitation report (Hanson 2001) contains data from 53 sites (Figure 1a). The period of record varies from as little as 6-8 years for a few sites to 35 years for 15 of the sites. Included with the precipitation data are measurements of shielded and unshielded precipitation and the computed wind-corrected value.

In the snow data report (Marks et al. 2001), 35 years of biweekly data from eight snow courses and 14 years of hourly snow water equivalent data from a snow pillow operated in the upper portion of the watershed are presented. Locations of the snow measurement sites are shown in Figure 1c. The snow course data include average snow water equivalent and snow depth taken along each snow course.

Climate data (other than precipitation) from three sites (Figure 1c) is presented by Hanson et al. (2001). It consists of 35 years of climate data from three sites, including daily values of maximum and minimum air temperature and pan evaporation. Since 1981, a number of climatic variables have been measured hourly (Table 1).

Basic soil water content and temperature data are presented in the lysimeter (Seyfried et al. 2001b), neutron probe (Seyfried et al. 2001d) and soil temperature (Seyfried et al. 2001c) data reports. Neutron probe data at several sites (Figure 1c) has been collected for over 25 years and consists of multiple depths. The lysimeters functioned for about 12 years and were used to check the neutron probe calibration. The soil temperature data, to a depth of 180 cm, has been functional since 1981 at 5 sites (Figure 1c).

Pierson et al. (2001) present hourly and breakpoint streamflow data from 13 weirs (Figure 1b). Nine weirs are currently in operation, with periods of record ranging from 23 to 35 years. Suspended sediment data were collected by manual sampling and several different automated samplers on an event basis.

Data availability

Data presented in each of these reports are available from the anonymous ftp site: ftp.nwrc.ars.usda.gov in the directory "databases/rcew" maintained by the USDA Agricultural Research Service, Northwest Watershed Research Center in Boise, Idaho. Each type of data presented is stored in an appropriately named subdirectory as ASCII files that have been compressed using a standard "zip" compression utility. Each file has an ASCII header providing brief information on file contents, location (Easting and Northing, UTM zone 11), both the GPS elevation and the DEM elevation, time format, period of record, column contents and units, missing data key, contact, citation and disclaimer information. In addition, a subdirectory contains a viewable version of a more detailed description of each data set, including additional photographs, data analysis, and graphical data presentation, that is stored in PDF format as intro.pdf geog.pdf, precip.pdf, climate.pdf, snow.pdf, soil_micro.pdf, and flow_sed.pdf. An ASCII README file in each directory gives a detailed description of the file formats and contents.

Example Applications

The existence of the basic data and scientific infrastructure facilitates research into many topics. Some examples of current research at the RCEW include, effects of fire on hydrologic response and cattle grazing: subgrid variability of remote sensing pixels, and effects of wind redistribution and canopy variation on snow melt and stream flow dynamics. The data are also informative for a number of management related parameters.

Characterization of stream flow

Basic stream flow data for smaller order streams is often lacking but management of watersheds may require some characterization of the stream flow regime. Questions arise such as: Were samples collected during a "typical" year? What are high or low stream discharge levels? How does flow vary downstream? Data collected from a research watershed are site-specific but can provide considerable insight into these and other questions within the region, especially considering the general lack of such data and the fact that many years of data collection are required to establish a valid statistical relationship.

In Figure 2, average monthly stream discharge from three sites along Reynolds Creek (Figure 1) are shown. Although the overall seasonality of flow is similar for all three locations, the relative distribution changes considerably downstream as the source areas become more variable. Discharge increases downstream, although the change from Tollgate to the Outlet is small considering that the Tollgate drainage area is only about a quarter that of the Outlet.

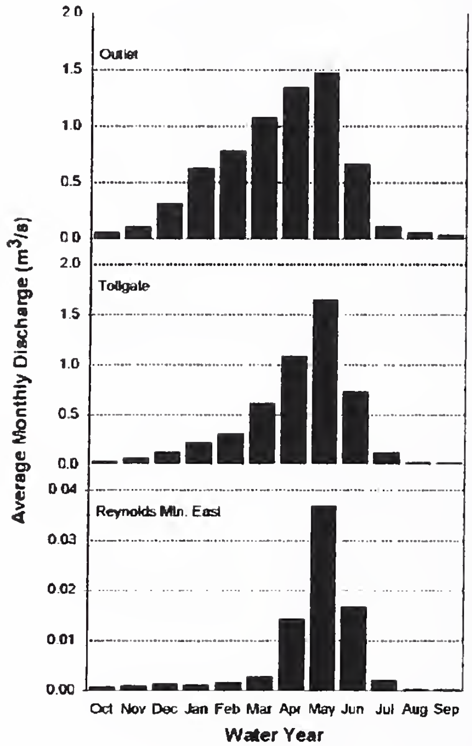

Figure 2. Average monthly stream discharge from three locations on Reynolds Creek.

Sediment discharge

Sediment discharge is of great concern in many watersheds, but numerous questions surround the issue of sampling. Although sediment concentration is generally related to stream discharge level, it is also strongly dependent on whether stream discharge is increasing or decreasing, the nature of the event causing discharge and the nature of the source area for discharge. Figure 3 shows suspended sediment concentrations at three locations along Reynolds Creek (Figure 1). Note that concentrations were much higher for a given level of discharge when stream flow was increasing than when it was decreasing. There was very little suspended sediment at Reynolds Mountain East when flow was generated by snowmelt, but increased dramatically during a rainfall event. Some of the highest sediment

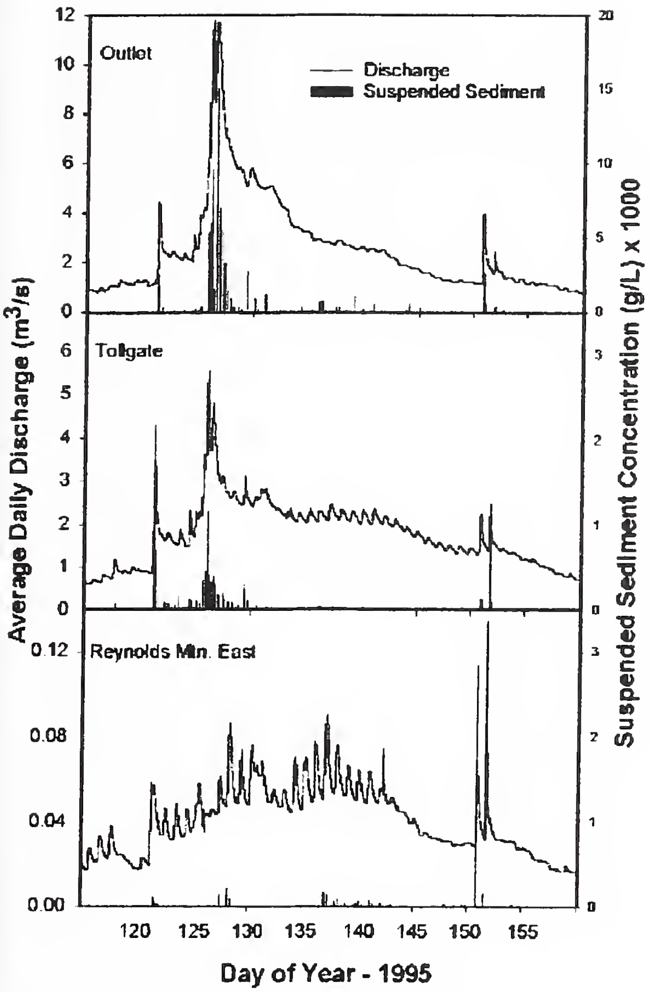

Figure 3. Average daily discharge rate and associated suspended sediment concentration for Outlet, Tollgate, and the Reynolds Mountain East watershed during spring flow in 1995.

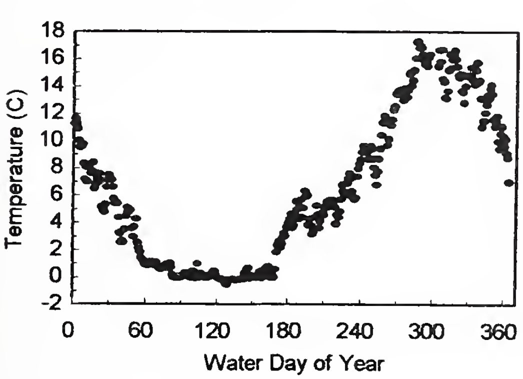

Figure 4. Noon-time well-mixed stream temperature at the Dobson Creek weir for one year.

concentrations we observe result from conditions in which the lower elevation, sparsely vegetated areas of the watershed contribute to stream flow.

Stream temperature

As needs change, so does the nature of data collected at the watersheds. Stream temperature was not collected at the RCEW prior to 1997 and therefore not included in the data reports. However, as stream temperature has become an issue of increasing interest, a stream temperature monitoring program has begun. The data presented here were collected at the weir throat and represent a mixed water condition. Although this may not be the most critical temperature for fish survival, it does represent an index of how temperatures vary with time and space within a watershed.

As with sediment concentration, there are several issues related to sampling of stream temperature. Again, stream temperature is related to discharge level, but also exhibits pronounced seasonal and diurnal variations. These factors should be taken into consideration when characterizing stream temperature.

In Figure 4 we show the variation in stream temperature at noon each day for one year taken at the Dobson Creek weir. During the winter, temperatures are close to freezing as the weir is often covered with snow and/or ice. Temperatures are most variable during the spring, when discharge may be derived from either snowmelt or rainfall. Another period of high temperature variably is in late summer, when temperatures are near maximum.

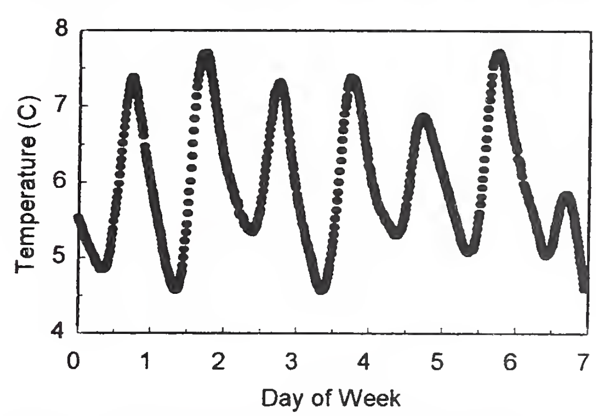

Figure 5. Stream temperature measured at the Dobson Creek were at 15 minute intervals from May 1-7, 2002.

Another concern is the amount of diurnal temperature fluctuation. Stream temperature, measured on 15 minute intervals from 1 May to 7 May, 2002 at the same site are shown in Figure 5. Stream temperatures at the weir ranged 2 to 3 °C during the day. Although the signal is considerably damped relative to air temperature, it is clear that air temperature variations are reflected in the stream temperature. Maximum temperatures during this week occurred between 5 and 6 PM. Temperatures at noon were much closer to the minimum than the maximum.

Acknowledgments

Steve Van Vactor assisted in compiling data. Gerald Flerchinger, Danny Marks, Clayton Hanson and Fred Pierson provided essential data. Reviews by Gerald Flerchinger and Danny Marks are appreciated.

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Merging Hydrology and Water Quality – The Mahantango Creek Watershed

W.J. Gburek, A.N. Sharpley, M.S. Srinivasan, P.J. Kleinman, T.L. Veith, P.A. Vadas, R.B. Bryant, W.L. Stout, C.J. Dell

Abstract

The Pasture Systems and Watershed Management Research Unit, USDA-ARS, University Park, PA, has as part of its mission, "conduct research leading to development of land, water, and plant management systems which insure the profitability and sustainability of northeastern agricultural enterprises while maintaining the quality of ground and surface waters." Watershed-related research in support of this mission is conducted within WE-38, a 7.3 sq-km subwatershed of Mahantango Creek within the Susquehanna River Basin about 40 km north of Harrisburg, PA. ARS began collecting data within WE-38 in 1967; since then, streamflow at the watershed outlet with supporting rainfall and climate data have been continuously recorded. In 1972, twelve ground water observation wells were installed within WE-38; water levels have been continuously recorded since then. Several specialized data sets have also been compiled, including: tri-weekly water quality sampling at the outlet of WE-38; geologic and hydraulic properties of the aquifer underlying WE-38 via rock cores, borehole testing, and geophysical sampling; annual land use surveys since 1990, including nutrient and pesticide application rates, land use and management, and crop yields; and ground water recharge over a seven-year period via monolith lysimeters and observation wells. Finally, specialized field sites and subwatersheds with specific research objectives have been established within WE-38. Results from a number of our studies illustrate how the wealth of data make WE-38 an ideal "outdoor laboratory" for exploration of surface and subsurface flow systems in the humid-climate upland watershed environment and the related transport of agricultural contaminants.

Keywords: watersheds, water quality, surface water, ground water

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Remote Sensing in Watershed Scale Hydrology

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Abstract

Remote sensing provides a means of observing hydrological state variables over large areas. The ones which we will consider is this paper are: land surface temperature from thermal infrared data, surface soil moisture from passive microwave data, snow cover and water equivalent using both visible and microwave data, water quality using visible and near infrared data, landscape cover using both visible and near infrared data and landscape surface roughness using lidar.

Keywords: remote sensing, watershed scale, soil moisture, snow, surface temperature, landscape features.

Introduction

Remote sensing is the process of inferring surface parameters from measurements of the upwelling electromagnetic radiation from the land surface. This radiation is both reflected and emitted by the land. The former is usually the reflected solar while the latter is in both the thermal infrared and microwave portions of the spectrum. There is also reflected microwave radiation as in imaging radars. The reflected solar is used in hydrology for snow mapping vegetation/land cover and water quality studies. The thermal emission in the infrared is used for surface temperature and in the microwave for soil moisture and snow studies. Remotely sensed observations can contribute to our knowledge of these quantities and, especially, their spatial variation. With remote sensing we only observe the surface but can obtain the spatial variability and if observations are made repeatedly the temporal variability. A major focus of remote sensing research in hydrology has been to develop approaches for estimating hydrology/ hydrometeorological states. In this paper we will concentrate on those applications of remote sensing which we believe are the most promising in hydrology which are land surface temperature, near-surface soil moisture, snow cover/water equivalent, water quality, landscape features such as roughness and land cover. We will not discuss the use of radars for precipitation estimation.

Land Surface Temperature

Land surface temperature is the result of the equilibrium thermodynamic state dictated by the energy balance between the atmosphere, surface and subsurface soil and the efficiency by which the surface transmits radiant energy into the atmosphere (surface emissivity). The latter depends on the composition, surface roughness, and physical parameters of the surface, e.g. moisture content. In addition, the emissivity generally will vary with wavelength for natural surfaces.

Land surface temperatures. derived using ASTER satellite imagery covering an area around the USDA-ARS Grazinglands Research Facility in El Reno, Oklahoma is displayed in Figure 1.

The spatial distribution of land surface temperature, T_{SURF}, reflects some significant differences in land cover conditions this time of year (September) with large areas of bare soil and wheat stubble from

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harvested winter wheat fields and grasslands used for cattle grazing, with a small areas of irrigated crop lands and water bodies. This type of spatially-distributed information is very useful for evaluating spatial patterns of evapotranspiration over large areas (Schmugge et al. 2002).

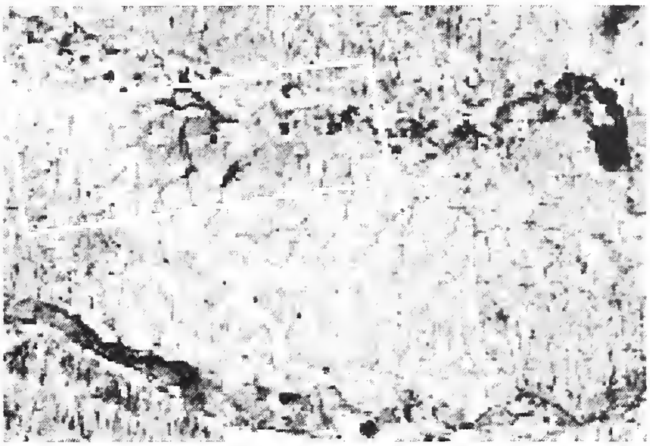

Figure 1. ASTER TIR imagery for a region in central Oklahoma on September 4, 2000 at 17:34 GMT for band13 of ASTER. The temperatures range from 36 degrees C (black) to 57 degrees C (white). The spatial resolution is 90 m.

Near Surface Soil Moisture

Passive microwave remote sensing techniques are used to measure near surface soil moisture because at microwave frequencies the most striking feature of the emission from the earth's surface is large contrast between water and land. Microwave remote sensing offers four unique advantages over other spectral remote sensing techniques:

1) The atmosphere is effectively transparent providing all-weather coverage in the decimeter range of wavelengths.
2) Vegetation is semi-tranparent allowing the observation of underlying surfaces in the decimeter range of wavelengths.
3) The microwave measurement is strongly dependent on the dielectric properties of the target which for soil is a function of the amount of water present.
4) Measurement is independent of solar
5) illumination which allows day or night observation.

Remote sensing cannot replace ground based methods for providing high quality profile data at a point. Its advantage is in mapping conditions at regional, continental and even global scales and possibly on a repetitive basis (Schmugge et al. 2002). Recently it has been shown that repetitive measurements of microwave brightness temperatures can yield subsurface soil hydraulic properties (Burke et al. 1998).

Passive microwave observations were made on eight days. The ESTAR data were processed to produce brightness temperature maps of a 740 km^2 area on each of the eight days. Using the algorithm developed by Jackson et al. (1995), these data were converted to soil water content images. Gray scale images for each day are shown in Figure 2.

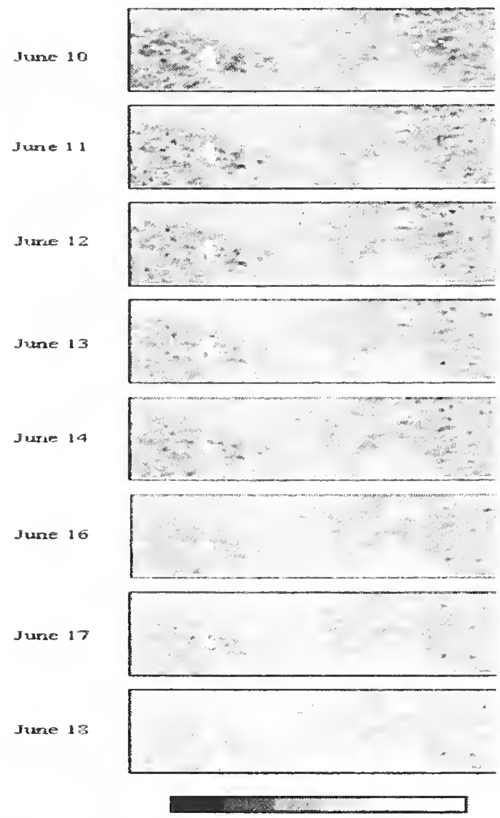

Figure 2. Near Surface (~ 0-5 cm) soil water content maps for the USDA-ARS Little Washita Experimental Watershed derived from passive microwave data collected onna series of days in June 1992. The spatial resolution is 200 m.

For most satellite systems the revisit time can be a critical problem in studies involving rapidly changing conditions such as surface soil water content. With very wide swaths it is possible to obtain twice daily coverage with a polar orbiting satellite. For most satellites, especially if constant viewing angle is important, the revisit time can be much longer. Optimizing the time and frequency of coverage is a critical problem for soil water content studies. Currently, all passive microwave sensors on satellite platforms operate at high frequencies (> 7 GHz). A more recent option is the multiple frequency Advanced Microwave Scanning Radiometer (AMSR) satellite systems that will include a 6.9 GHz channel. AMSR holds great promise for estimating soil water content in regions of low levels of vegetation. AMSR is not the optimal solution to mapping soil water content but it is the best possibility in the near term. Based on the published results and supporting theory (Wang 1985, Choudhury and Golus 1988, Owe et al. 1992, Ahmed 1995, Njoku and Li 1999), this instrument should be able to provide surface moisture information in regions of low vegetation cover, less than 1 kg m^{-2} vegetation water content. To pursue the use of space observations further research programs are underway to develop a space based system with a 1.4 GHz channel which would provide improved global soil moisture information (Kerr et al. 2001). The Soil Moisture and Ocean Salinity (SMOS) mission is currently being implemented by the European Space Agency. This instrument will extend the aperture synthesis approach pioneered by ESTAR to two dimension and will make dual polarized measurements at a range of angles. With these data it is expected to be able obtain not only soil moisture but also vegetation water content at a 50 km resolution (Wigneron et al. 2000).

Snow Cover and Water Equivalent

Snow cover can be detected and monitored with a variety of remote sensing devices. The greatest number of applications have been found in the visible and near infrared region of the electromagnetic spectrum. Because of Landsat and SPOT frequency of observation problems, many users have turned to the NOAA polar orbiting satellite with the AVHRR, which has a resolution of about 1 km in the 0.58-0.68Fm red band. The frequency of coverage is twice every 24 hours (one daytime pass and one nighttime pass). The major problem with the NOAA-AVHRR data is that the resolution of 1 km may be insufficient for snow mapping on small basins. Data from the MODIS instrument on NASA's EOS satellites with 250 m resolution in two visible bands will partially alleviate this problem. Despite the various problems mentioned, visible aircraft and satellite imagery have been found to be very useful for monitoring both the buildup of snow cover in a drainage basin and, even more importantly, the disappearance of the snow covered area in the spring. This disappearance or depletion of the snow cover is important to monitor for snowmelt runoff forecasting purposes. It has been recommended that the optimum frequency of observation of the snow cover during depletion would be once a week (Rango 1985). Depending on the remote sensing data used, it could be very difficult to obtain this frequency. Certain snowmelt-runoff applications have been possible with as few as two to three observations during the entire snowmelt season (Rango 1985).

Passive microwave data provides several advantages not offered by other satellite sensors. Studies have shown that passive microwave data offer the potential to extract meaningful snowcover information, such as *SWE*, depth, extent and snow state. SSM/I is a part of an operational satellite system, providing daily coverage of most snow areas, with multiple passes at high latitudes, hence allowing the study of diurnal variability. The technique has generally all-weather capability (although affected by precipitation at 85GHz), and can provide data during darkness. The data are available in near-real time, and hence can be used for hydrological forecasting. There are limitations and challenges in using microwave data for deriving snow cover information for hydrology. The coarse resolution of passive microwave satellite sensors such as SMMR and SSM/I (~25km) is more suited to regional and large basin studies, although Rango et al. (1989) did find that reasonable *SWE* estimates could be made for basins of less than 10,000 km^2. The AMSR to be launched on NASA's EOS satellite, AQUA, in 2002 will provide a wider range of wavelengths and with better spatial resolution than what is currently available (Schmugge et al. 2002).

Water Quality

Remote sensing techniques for monitoring water quality depend on the ability to measure these changes in the spectral signature backscattered from water and relate these measured changes by empirical or analytical models to water quality parameters. The optimal wavelength used to measure a water quality parameter is dependent on the substance being measured, its concentration, and the sensor characteristics.

Major factors affecting water quality in water bodies across the landscape are suspended sediments (turbidity), algae (i.e., chlorophylls, carotenoids), chemicals (i.e., nutrients, pesticides, metals), dissolved organic matter (DOM), thermal releases, aquatic vascular plants, pathogens, and oils. Suspended sediments, algae, DOM, oils, aquatic vascular plants, and thermal releases change the energy spectra of reflected solar and/or emitting thermal radiation from surface waters which can measured by remote sensing techniques. Most chemicals and pathogens do not directly affect or change the spectral or thermal properties of surface waters so they can only be inferred indirectly from measurements of other water quality parameters affected by these chemicals.

Remote sensing has been used to measure chlorophyll concentrations spatially and temporally. As with suspended sediment measurements, most remote sensing studies of chlorophyll in water are based on empirical relationships between radiance/reflectance in narrow bands or band ratios and chlorophyll. A variety of algorithms and wavelengths have been used successfully to map chlorophyll concentrations of the oceans, estuaries and fresh waters. While estimating chlorophyll by remote sensing technique is possible, studies have also shown that the broad wavelength spectral data available on current satellites (i.e., Landsat, SPOT) do not permit discrimination of chlorophyll in waters with high suspended sediments (Dekker and Peters 1993, Ritchie et al. 1994) due to the dominance of the spectral signal from the suspended sediments. Research on the relationship between chlorophyll and the narrow band spectral details at the "red edge" of the visible spectrum (Gitelson et al. 1994) has shown a linear relationship between chlorophyll and the difference between the emergent energy in the primarily chlorophyll scattering range (700-705 nm) and the primarily chlorophyll absorption range (675-680 nm).

While current remote sensing technologies have many actual and potential applications for assessing water resources and for monitoring water quality, limitations in spectral and spatial resolution of many current sensors on satellites currently restrict the wide application of satellite data for monitoring water quality. New satellites (i.e., SEAWIFS, EOS, MOS, IKONOS, etc.) and sensors (hyperspectral, high spatial resolution) already launched or planned to be launched over the next decade will provide the improved spectral and spatial resolutions needed to monitor water quality parameters in surface waters from space platforms (Schmugge et al. 2002).

Landscape Features

The landscape features most important to hydrology are landscape roughness, land cover and leaf area.

Landscape roughness

Roughness refers to the unevenness of the earth's surface due to natural processes (i.e., topography, vegetation, erosion) or human activities (i.e., buildings, power lines, forest clearings). Roughness affects transport of hydrometeorological fluxes between the land surface and atmosphere as well as below the surface, i.e., infiltration and water movement. Roughness is often separated in different complexities related to its effects on land surface-atmosphere dynamics. The complexities are 1) vegetation and urban roughness where the horizontal scale is relatively small, 2) transition roughness between landscape patches (i.e., plowed field next to a forest), and 3) topographic roughness due to changing landscape elevations. These complexities and scales have different effects on wind, heat, and water movement and are difficult to measure in the field at large scales. Lidar, Synthetic Aperture Radar (SAR), Digital Elevation Models (DEM), and photogrammetry are among the remote sensing techniques that have been used to measure landscape surface roughness properties over large areas.

The need for accurate and rapid measurements and assessments of land surface terrain features to estimate the effects of land surface roughness on hydrometeorological processes led to the application of lidar distancing technology from an aircraft-based platform (Ritchie and Jackson 1989, Ritchie 1996). Satellite platforms have also been employed (Harding et al. 1994).

The first applications of the airborne lidar altimeter were to measure topography (Link 1969) and sea ice roughness (Robin 1966). Lidar altimeters can measure long topographic profiles quickly and efficiently

Vegetation canopies are an important part landscape roughness that are difficult to measure by conventional techniques. Airborne lidar measurements provided accurate measurements of canopy top roughness (Figure 8a), heights (Figure 8b) and cover (Ritchie et al. 1992, Ritchie et al.

1993, Weltz et al. 1994). Scanning lasers (Rango et al. 2000) can provide a three-dimensional view of canopy structure needed understand canopy roughness.

Land cover

Landsat data is the used in developing the landuse and land cover for the agricultural scene. Multi-temporal coverage acquired during the crop season is used for developing the classification. A minimum of four images acquired during the entire season is desirable for creating signature files for the classification. A set of ground data is collected to do the validation of the classification, choosing predominant crops as well as the vegetation cover that may cause confusion in the classification. The base classification is created using four visible and Near- IR bands (Landsat 3, 4, 5 & 7) for the various classes in the scene. The signatures for each land cover class are developed using non-supervised classification for each set of AOI (Area of Interest). Signatures based on all AOI's are combined into one signature file for each area and used within supervised classification. The supervised classification is performed and an accuracy table is generated based on classification performance where ground data was acquired (Doraiswamy et al. 2001).

Leaf area index

The first step toward the development of the canopy level leaf areas index (LAI) is the development of an accurate land cover map. MODIS imagery (250 m resolution) is used in developing an LAI image for every clear acquisition available during the growing season. The one dimensional canopy reflectance model, SAIL (Scattering by Arbitrarily Inclined Leaves) (Verhoef 1984), provides simulated canopy reflectance in the direction of the sensor. The SAIL model calculates reflectance in the sensor direction as a function of canopy parameters and acquisition and sun angles and requires canopy parameters as leaf angle distribution, single leaf reflectance and transmittance and soil reflectance. The model run in the inversion mode using canopy reflectance as input simulates the canopy LAI. The model simulated LAI is calculated by minimizing the mean square differences of measured and simulated canopy reflectance for the visible and near infrared bands (Doraiswamy, et al. 2002). The model is run separately for each of the crop types in the scene and integrated to get the scene LAI.

Summary/Conclusions

Remote sensing provides a means of observing hydrological state variables temporally and over large areas. This paper summaries the state of the art of estimating from remote sensing land surface temperature, near-surface soil moisture, snow cover/water equivalent, water quality, landscape roughness and land cover.

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New Patented Technology to Protect and Improve Water Quality in Lakes

Raymond Bauer

Abstract

New technology, called the Surface Water Protection System (SWPS)[1], developed by Eco Boom, Inc. of New York, can be an effective tool to help solve pollution problems associated with stormwater runoff. A certain fabric geotextile mat, when fabricated and deployed to specifications, can control Biodegradable Organic Materials (BOM) and cause the depletion of most microorganisms, releasing odorless gas. The fabric filter material has small openings in sufficient number to permit passage of a predetermined volume of water per unit of time, but small enough to block particulate biodegradable organic matter of 20 microns or larger. The SWPS filter curtain prevents passage of the particulate matter and non-organic turbidity-causing particles. As a result of the increased microbial density, the microorganisms or bacteria decompose and consume the BOM. The quality of the water, as it subsequently passes through the filter curtain and enters the main body of water, is nearly devoid of the microorganisms and particulates. This rapid rate of consumption does not occur naturally in large bodies of water due to the lack of microbial density. When there is no discharge into the sedimentation basin, the water is calm except for any wind-induced wave motion. Following a rain event, colloids will be in suspension. It is theorized these minute particles will have Brownian motion due to thermal variations. The colloids will collide and coagulate with an increase in mass, and aggregate. These aggregates subsequently fall out of the water column and become sediment, which also fails to pass through the filter curtain.

[1] Patent issued 2-12-2002 "Method and Apparatus to Improve Water Quality in Substantially Enclosed Bodies of Water" by Raymond Bauer.

Keywords: biodegradation, microbial density, consumption

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Water Quality and Quantity II

Herbicide Contamination and Transport in Northern Missouri and Southern Iowa Streams

Robert N. Lerch, Paul E. Blanchard

Abstract

Herbicide contamination of streams has been well documented, but little is currently known about the specific factors affecting watershed vulnerability to herbicide transport. The primary objectives of this study were: 1) to document herbicide occurrence and transport from watersheds in the northern Missouri/ southern Iowa region in an effort to quantify watershed vulnerability to herbicide transport; and 2) to compute the contribution of this region to the herbicide load of the Missouri and Mississippi Rivers. Grab samples were collected under baseflow and runoff conditions at 21 hydrologic monitoring stations between April 15 and July 15 from 1996 to 1999. Samples were analyzed for commonly used soil-applied herbicides (atrazine, cyanazine, acetochlor, alachlor, metolachlor, and metribuzin) and four triazine metabolites. Estimates of herbicide load and relative losses were computed for each watershed. Median parent herbicide losses, as a percentage of applied, ranged from 0.33 to 3.9%; loss rates that were considerably higher than other areas of the United States. Watershed vulnerability to herbicide transport, measured as herbicide load per treated area, showed that the runoff potential of soils was a critical factor affecting herbicide transport. Herbicide transport from these watersheds contributed a disproportionately high amount of the herbicide load to both the Missouri and Mississippi Rivers. Based on these results, this region of the Corn Belt is highly vulnerable to hydrologic transport of herbicides from fields to streams, and it should be targeted for implementation of management practices designed to reduce herbicide losses in surface runoff.

Keywords: herbicide transport, watershed vulnerability, hydrologic soil group, corn and soybean herbicides

Introduction

In the Corn Belt region of the United States, herbicide contamination of streams is widely recognized as one of the major environmental impacts of row crop production. Field runoff represents the primary hydrologic mechanism responsible for herbicide transport from agricultural fields to streams (Wauchope 1978, Leonard 1988). Over the last decade, several studies have overwhelmingly demonstrated that row crop production leads to frequent detections and potentially harmful levels of commonly used corn and soybean herbicides and their metabolites in stream water (Thurman et al. 1992, Richards et al. 1993, Lerch et al. 1998, Blanchard and Lerch 2000). Maximum herbicide concentrations and frequency of detections follow herbicide application to fields in the spring (Thurman et al. 1992, Blanchard and Lerch 2000). The inopportune coincidence of herbicide application with intense spring rainfall events creates a critical herbicide loss period during the second and third quarters of each year (Richards et al. 1993, Clark et al. 1999).

There are many factors affecting herbicide transport to streams, and they can be organized into four general categories: 1) intrinsic factors - soil and hydrologic properties and geomorphologic characteristics of the watershed; 2) anthropogenic factors - land-use and herbicide management; 3) climate factors - particularly precipitation and temperature; and 4) herbicide factors - chemical and physical properties and formulation.

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One of the most important intrinsic watershed characteristics affecting herbicide transport is the runoff potential of its soils, which is primarily determined by soil particle size distribution (i.e., soil texture) and topography. Hydrologic soil group (HSG) categories represent one way to rank runoff potential of soils, and they have been shown to be valid indicators of regional hydrology and watershed vulnerability to herbicide transport (Blanchard and Lerch 2000). Soil texture directly affects water infiltration rate and, therefore, it largely determines the dominant hydrologic pathway of the watershed. Watersheds dominated by clayey soils, or those with prominent argillic horizons, will typically have low infiltration rates, high surface runoff volume, and will be vulnerable to surface transport of herbicides. Those watersheds with a predominance of silty or sandy soils will have high infiltration and percolation rates, low surface runoff volume, low vulnerability to surface transport of herbicides, but high vulnerability to nitrate leaching.

Studies conducted by the U.S. Geological Survey (USGS) National Stream Quality Accounting Network (NASQAN) have been used to estimate the herbicide load of the largest sub-basins to the Mississippi River (Clark et al. 1999). While the NASQAN data has provided useful information about herbicide transport in these large sub-basins, there remains a gap in our knowledge regarding the herbicide contribution of smaller sub-basins within the Missouri and Mississippi Rivers. This information would contribute to a more detailed understanding of which areas transport the greatest herbicide load to the large river systems, and it will provide a basis for prioritizing implementation of best management practices (BMPs) to the most vulnerable watersheds.

The primary objectives of this study were to document herbicide occurrence and to estimate herbicide transport from 21 watersheds encompassing the northern Missouri/ southern Iowa region (Figure 1) in an effort to quantify watershed vulnerability to herbicide transport. An additional objective was to compute the contribution of northern Missouri and southern Iowa streams to the herbicide load of the Missouri and Mississippi Rivers.

Methods

Stream sampling

The study area included 21 sites, 20 streams and one reservoir outlet, draining approximately 56,700 km^2 in northern Missouri and southern Iowa (Figure 1 and Table 1). Stream samples represented watershed areas ranging from 21,276 to 1,796,154 ha, with a median area of 115,500 ha (Table 1). Grab samples were collected at USGS hydrologic monitoring stations (Figure 1 and Table 1) between April 15 and July 15 from 1996 to 1999. Samples were collected under baseflow and runoff conditions. A single sample was collected from the main flow paths for the smallest streams, and for the larger streams, sub-samples were collected at two or three locations in a transect across the stream. All samples were transported to the lab on ice and filtered through 0.45 µm nylon filters within 48 hours of collection. For the larger streams, all sub-samples were mixed to create a single composite sample just before filtration. The fewest number of samples collected at any site was five and the most was 11, with an average of eight samples per site per year. Only seven sites were sampled in 1996 (Table 1), and data from these sites were only used for computing annual herbicide loss as a percent of applied.

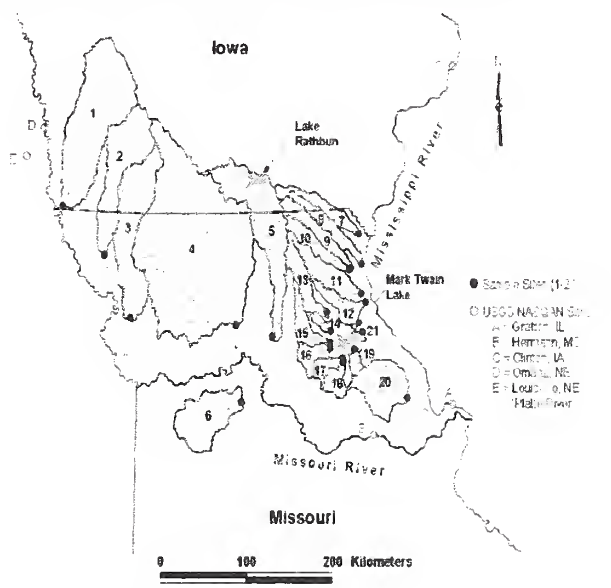

Figure 1. Watersheds of included in this study. Entire area encompasses ~56,700 km^2. See Table 1 for list of watershed names.

Table 1. Sampled watersheds and watershed drainage areas.

Watershed	Site Number	Years Sampled	Watershed Area (ha)
Nishnabotna River	1	1997-1999	730,161
Nodaway River	2	1997-1999	393,200
Platte River	3	1997-1999	454,439
Grand River	4	1997-1999	1,796,154
Chariton River	5	1997-1999	489,167
Blackwater River	6	1997-1999	289,744
Fox River	7	1997-1999	102,980
Wyaconda River	8	1997-1999	103,230
North Fabius River	9	1997-1999	115,526
Middle Fabius River	10	1997-1999	100,152
South Fabius River	11	1996-1999	156,558
North River	12	1996-1999	92,130
North Fk Salt River¶	13	1996-1999	119,286
Crooked Creek¶	14	1996-1999	21,276
Middle Fk Salt River¶	15	1996-1999	86,556
Elk Fk Salt River¶	16	1997-1999	67,488
Long Branch Creek¶	17	1997-1999	47,129
South Fk Salt River¶	18	1996-1999	59,050
Lick Creek¶	19	1996-1999	26,943
Cuivre River	20	1997-1999	240,570
Salt River§	21	1997-1999	608,650

¶ Sub-basins of the Salt River.
§ Salt River site was only used for computing herbicide load to Mississippi River.

Chemical analyses

Samples were analyzed for the following commonly used corn and soybean herbicides and selected triazine metabolites: acetochlor, alachlor, atrazine, cyanazine, cyanazine amide, deethylatrazine, deisopropylatrazine, hydroxyatrazine, metolachlor, and metribuzin. For all analytes, terbutylazine was added as a surrogate to 100 or 200 mL samples which were then concentrated using 500 mg C_{18} solid-phase extraction cartridges. Herbicide concentrations were then determined by gas chromatography (GC) with N-P or mass spectral (MS) detection or high performance liquid chromatography (HPLC) with UV detection. Only hydroxyatrazine and cyanazine amide were analyzed by HPLC. Detection limits, using GC/MS or HPLC/UV, and frequencies of detection are given in Table 2.

Table 2. Detection limits and frequency of herbicide detection (average of 1998 and 1999).

Herbicide or Metabolite	Detection Limit	Frequency of Detection
	ng L^{-1}	%
Atrazine	3	100
Deethylatrazine	4	99
Deisopropylatrazi	8	95
Hydroxyatrazine	50	99
Cyanazine	9	89
Cyanazine Amide	20	85
Acetochlor	6	91
Alachlor	3	92
Metolachlor	2	99
Metribuzin	8	87

Land and herbicide use estimates

Land use was based on land cover data produced from 30 meter resolution Landsat Thematic Mapper spectral data. Both the Iowa and the Missouri data sets were developed using nominal 1992 data. Planted row crop acreage was estimated for each watershed by combining the land cover data with county-level row crop data from the USDA-National Agricultural Statistics Service (USDA-NASS 1997-2000). The key assumption in this approach is that the distribution of row crops was uniform within a county. The suite of herbicides analyzed in this study pertained only to corn, soybeans, and sorghum. Therefore, herbicide use estimates were computed for each watershed by using the estimated acreage of these crops within each watershed combined with state level annual reports of agricultural chemical usage (USDA-NASS 1997-2000) for each relevant crop. These reports include planted crop acres, the percentage of crop acres treated by a given herbicide, and its average annual use rate for each state.

Load calculations

Herbicide loads for all sampled watersheds, and the NASQAN sites (Figure 1), were computed on a daily basis for the critical loss period of April 15 to June 30 of each year. Herbicide concentrations for non-sampled days were estimated by linear interpolation between measured concentrations (Clark et al. 1999). In addition, boundary conditions were assumed for extrapolating herbicide concentrations to the beginning and ending of the critical loss period. All herbicide and metabolite concentrations were assumed to be zero on April 1 and July 31. Since the NASQAN project performs year-round monitoring, no assumptions were required with respect to beginning or ending concentrations. Measured or estimated concentrations were multiplied by the average daily stream discharge to estimate daily load.

Results

Study area characteristics

Land-use within the study area is predominantly agricultural, with an average of 96% of the watershed areas in row crop or forage production. Corn and soybeans account for about 80% of the row crop production. Sorghum is a significant crop only in the eastern portion of the study area within Missouri. Row cropping intensity ranged from 22% to 77% of the watershed area. In general, row cropping intensity is high in the western watersheds (sites 1-3), low in the central watersheds (sites 4-6), and intermediate in the eastern watersheds.

The HSG represent four general soil categories (A to D), with HSGA having the lowest and HSGD the highest runoff potential. Since runoff potential is critical to surface transport of herbicides, it follows that watersheds dominated by HSGC and D would be the most vulnerable to herbicide transport and subsequent stream contamination. Within the study area, the runoff potential of soils increases from west to east (Figure 2). Sites 1-3 represent watersheds with low runoff potential soils mainly within HSGA or B. Sites 4-6 represent a transition in which the soils become progressively higher in runoff potential from west to east. Sites 7-21 represent watersheds with the highest proportion of soils within the HSGC and D categories. These watersheds have >75% of their terrace and upland areas within the two highest runoff potential categories.

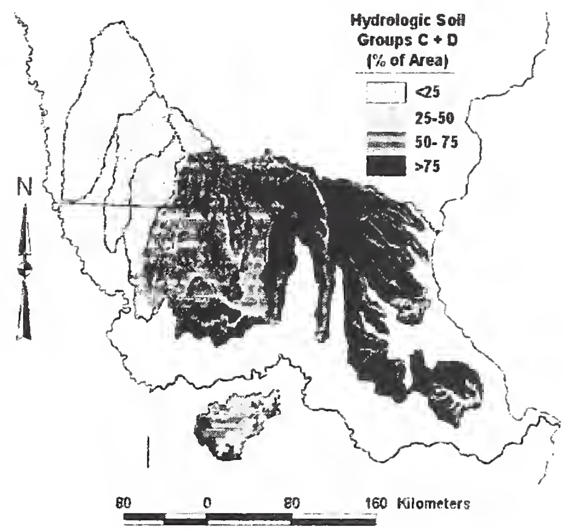

Figure 2. Percentage of watershed area in Hydrologic Soil Groups C and D.

Relative herbicide losses

Relative herbicide losses were computed as a percent of the herbicide applied within the watershed. This approach removes the dominant influence that watershed size and herbicide usage have on absolute loads, but it relies on herbicide usage estimates of unknown accuracy (USDA-NASS 1997-2000). Relative herbicide loss estimates for 1996-1999 showed that the triazine herbicides had the highest losses, with peak atrazine or cyanazine losses exceeding 10% of applied in five watersheds (sites 11-15) in 1996 (Figure 3). Median herbicide losses were: 3.9% for atrazine, 2.3% for cyanazine, 2.1% for metribuzin, 2.0% for metolachlor, 0.50% for acetochlor, and 0.33% for alachlor (Figure 3). These median losses were 2.8 to 40 times higher than median herbicide losses reported by Capel et al. (2001) for 43 USGS

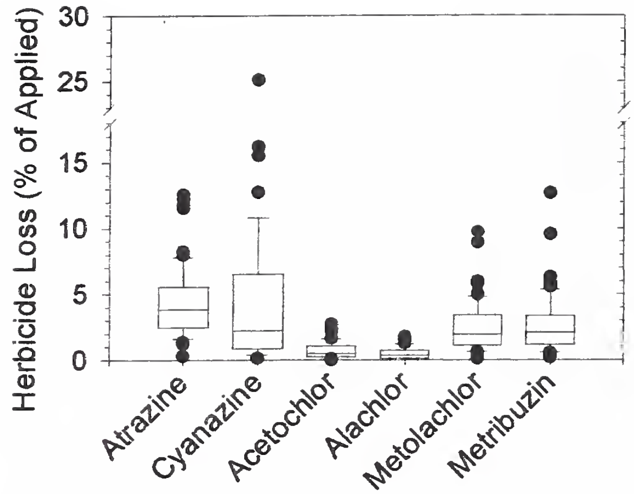

Figure 3. Relative herbicide loss, expressed as percent of applied, for 1996-1999.

National Water-Quality Assessment Program sites and their review of several hundred studies published in the international literature.

The relative herbicide losses reported in Figure 3 represent conservative estimates of actual herbicide loss for three reasons: 1) metabolites were not included (when atrazine metabolites were included median losses increased from 3.9% to 4.5% of applied); 2) loss was computed for three months of the year; and 3) the assumed boundary conditions for extrapolation were conservative, particularly for the triazine herbicides.

Areal herbicide losses and watershed vulnerability

Areal herbicide loss, expressed as the sum of all herbicide and metabolite losses on a treated area basis, provides a direct measure of watershed vulnerability to herbicide transport (Figure 4). This approach to calculating relative herbicide losses, like loss as a percent of applied, also removes the influence of watershed size and herbicide usage, but it has the additional advantage of using land-use data that has been verified for accuracy. Herbicide transport from treated fields occurred at the lowest rates in the northwestern (sites 1-3) and southeastern (sites 18-20) watersheds, and the highest loss rates occurred in several of the northeastern watersheds (sites 7-9, 11, 13, and 15) and in site 6 (Figure 4). The areal loss rates generally follow the pattern of HSGC and D soils (Figure 2), in which herbicide loss rates tend to increase with increasing runoff potential of soils (Figure 2). Of the seven watersheds in which areal loss rates were 350 g/ha or greater, six of them (Sites 7-9, 11, 13, and 15) have >75% of their area comprised of high runoff potential soils within HSGC and D categories. Soils within these most vulnerable watersheds have claypans or pronounced argillic horizons that restrict water percolation and increase surface runoff.

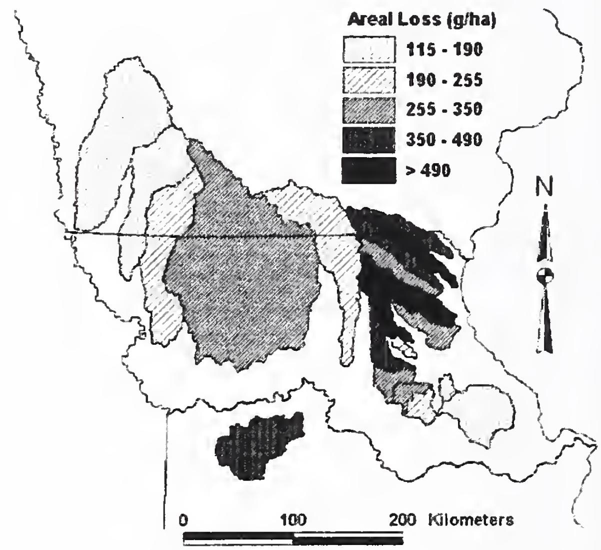

Figure 4. Watershed vulnerability to herbicide transport, expressed as the sum of all herbicide and metabolite losses on a treated area basis.

Contribution to the herbicide load in the Missouri and Mississippi Rivers

Herbicide transport from the six Missouri River tributaries (sites 1-6) contributed a disproportionately high amount of the herbicide load to the Missouri River at Hermann, MO (site B) from 1997 to 1999 (Figure 1 and Table 3). These six tributaries account for only 3.1% of the drainage area and an average of 14% of the stream discharge within the Missouri River basin. Averaged over the three years, this region of the Missouri River basin contributed just over three-fourths of the metribuzin load, approximately one-third of the atrazine, cyanazine, and acetochlor loads, and about one-fifth of the metolachlor load. The small area of the Missouri River basin monitored in this study was obviously a major contributor to the herbicide load within the entire basin. This indicated that the

Table 3. Contribution of Missouri and Mississippi River tributaries to the herbicide load at Hermann, MO (Site B) and Grafton, IL (Site A).*

Watershed	Area	Discharge	Atrazine	Cyanazine	Aceto-chlor	Alachlor	Metol-achlor	Metribuzin
	% of Missouri River at Hermann, MO							
MO River Tributaries†	3.1	14	38	37	30	12	21	78
	% of Mississippi River at Grafton, IL							
MS River Tributaries‡	3.4	3.2	7.4	9.1	2.3	7.5	5.4	31

* Average of 1997-1999.

† Missouri River tributaries represent the sum of Sites 1-6.

‡ Mississippi River tributaries represent the sum of Sites 7-12 and 21.

combination of high herbicide inputs and moderate to high runoff potential of the soils resulted in the disproportionately greater herbicide loss rates of these six watersheds compared to other intensive cropping regions within the lower Missouri River basin.

The Mississippi River tributaries (sites 7-12 and 21) also contributed a disproportionately high amount of the herbicide load to the upper Mississippi River at Grafton, IL (site A) (Figure 1 and Table 3). These seven watersheds represent 3.4% of the drainage area, and an average of 3.2% of the stream discharge within the upper Mississippi River basin (Table 3). Metribuzin transport was the most highly disproportionate of the herbicides measured, accounting for an average of 31% of the load at Grafton, IL. Atrazine, cyanazine, alachlor, and metolachlor transport were also disproportionately high from this region, but much less so than metribuzin. Acetochlor transport was disproportionately low relative to stream discharge when averaged over the three years. Because of the greater similarity in land-use of these watersheds to the upper Mississippi River basin, the contribution was generally not as disproportionate as that computed for the Missouri River basin.

Conclusions

Relative herbicide loss, as a percentage of applied, was considerably higher within the northern Missouri/ southern Iowa region compared to other areas of the U.S. or Corn Belt. The computation of areal herbicide loss rates, on a treated area basis, provided a direct measure of watershed vulnerability to transport. Areal loss rates were strongly correlated to soil runoff potential. The most vulnerable watersheds are dominated by claypan soils or soils with pronounced argillic horizons that increase surface runoff. Herbicide transport from the northern MO/southern IA region contributed a disproportionately high amount of the herbicide load to the MO and MS Rivers from 1997 to 1999. Atrazine, cyanazine, metolachlor, and metribuzin transport were disproportionately high from both the MO and MS tributaries. This region of the Corn Belt is highly vulnerable to herbicide transport, and many of these watersheds should be targeted for BMP implementation to reduce herbicide loading to streams.

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A Derived-Distribution Approach to Estimating Daily Loads of Sediment in Coastal Plain Streamflow

Robert Hubbard, Joseph Sheridan, David Bosch

Abstract

Standards for allowable total maximum daily loads (TMDLs) of sediment are being assigned to selected streams and rivers across the US. Estimates of total daily loads (TDLs) of sediment are required for comparison with TMDLs for those streams which are deemed to have impaired water quality due to sediment, and are required for formulating plans to improve these streams. A TMDL is defined as a daily stream loading rate (e.g. kg day^{-1}). However, actual stream sediment data are collected in terms of concentrations (e.g. mg L^{-1}), thus requiring estimates of streamflow to make the necessary rate and load conversions. The USDA-ARS, Southeast Watershed Research Laboratory (SEWRL) in Tifton, GA has collected over 30 years of hydrologic and climatic data from the 334 km^2 Little River Watershed (LRW). The LRW is typical of heavily vegetated, slow-moving stream systems of the southeastern Coastal Plain. Field studies quantifying dissolved and suspended loads in LRW streamflow have been conducted during the 30 year period (1974-1978, 1984-1986), but a continuous record of measured loads does not exist. A "derived distribution" (DD) approach is being tested to estimate TDLs of total solids (TS) and suspended sediment (SS) loads in LRW streamflow. A "derived distribution" (DD) is the frequency distribution of the dependent variable that is "derived" from the distribution of independent variables through a monotonic functional relationship between the dependent and independent variables. This paper presents results of coupling a derived flow distribution with mean SS and TS concentrations to estimate TDLs for the LRW of the southeastern Coastal Plain.

Keywords: streamflow, sediment loads, TMDLs, Coastal Plain watersheds

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Introduction

Water quality of streams, lakes or other water bodies may be degraded by excessive amounts of sediment in surface runoff or base flows. Heavy loads of SS in streamflow reflect erosion from cropland or from roads and skid trails associated with clear cutting of forest land, both of which are of serious environmental concern. Numerous studies have reported sediment concentrations and loads for a variety of drainage systems (Long and Bowie 1963, McGuiness et al. 1971, Griffiths 1982, Neff 1982, Carling 1983), along with information relating loads to rainfall intensity and duration, runoff amount, drainage area, or land use (Dragoun and Miller 1966, Dendy and Bolton 1976, Costa 1977, Ostry, 1982).

Total daily loads (TDLs) of sediment must be assigned to watersheds across the US to quantify total assimilative daily loads and maintain water quality, to identify streams with impaired water quality (TDLs exceeding TMDLs), and for formulating plans to improve water quality (Bonta 2002). A TDL is a daily load rate (e.g., kg day^{-1}), but field data are collected in terms of concentrations (e.g., mg L^{-1}), thus requiring flow rates to make the necessary conversions. When discrete samples are obtained, often flow rates are not measured, or a qualitative description of flow is made (e.g. "low", "high", "numerical rank", etc.).

A "derived distribution"(DD) is the frequency distribution of the dependent variable that is "derived " from the distribution of the independent variable through a monotonic functional relationship between the dependent and independent variables. Derived distributions have identical probabilities for corresponding independent- and dependent-variables values.

The USDA-ARS, Southeast Watershed Research Laboratory (SEWRL) in Tifton, GA has collected over 30 years of hydrologic and climatic data from the 334 km^2 Little River Watershed (LRW). In addition, streamflow samples for determination of total solids (TS) and suspended sediment (SS) were collected from LRW flow-measurement locations for 1974 –1981, and again from 1984 –1986.

This paper presents the results of using a DD approach to estimating TDLs in LRW streamflow based on the flow duration probability curves and data from the TS and SS studies.

Methods

Study area description

The Little River Watershed is located within the Tifton Upland of the Southern Coastal Plain physiographic region of the southeastern USA, and typifies the topography, geology, soils, and land use of that region (Yates, 1978). The Tifton Upland lies within the outcrop area of the Miocene series Hawthorn Formation. The Hawthorn Formation is continuous and considered to be an effective aquiclude for the Tifton Upland (Stringfield 1966).

Upland soils are classified primarily as fine-loamy (or loamy), siliceous, thermic Plinthic Paleudults (Calhoun 1983), generally having infiltration rates in excess of 5 cm hr^{-1} (Rawls et al. 1976). Internal drainage of upland soils is good to very good. Bottomland soils adjacent to drainage networks are primarily loamy, siliceous, thermic Arenic Plinthic Paleaquults with some Fluvaquents and Psammaquents (Calhoun 1983). Drainage of bottomland soils is poor to very poor with water standing on the surface during portions of the year. Major soil series are the same from the upper, or headwater areas, to downstream portions of LRW, although relative percentages vary somewhat between watersheds.

Topographically, the region has been described as an area of floodplains, river terraces, and gently sloping uplands, with moderately wide interstream divides separating relatively broad valleys. The divides are nearly level, gently sloping, or undulating. Valley bottoms are nearly level and typically swampy, while valley sides are gently sloping. Channel slopes are generally < 0.1 %, whereas upland side slopes generally range up to 5 % (Yates 1978). The low-lying, poorly drained areas (the most prominent feature of the Coastal Plain region) are characterized by short hydroperiods, an alluvium substratum, and mixed tree and shrub-type vegetation (Kitchens et al. 1975). Drainage systems such as these have been variously termed river floodplain swamps (Kitchens et al. 1975), floodplain wetlands (Kibby 1978), seasonally flooded wetlands (Johnston et al. 1984), forest wetlands (Leitman et al. 1983), and blackwater swamp systems (Wharton 1978).

The LRW (Figure 1) has mean sea level elevations ranging from 85 to 122 m. Small watersheds in this region typically have intermittent flow, with nonflow periods generally occurring in the late summer or fall.

Hydrologic measurements and streamflow sampling

The LRW was instrumented by USDA-ARS beginning in 1966 for Coastal Plain hydrologic research. Hydrologic data available includes both precipitation and streamflow observations. Streamflow stage was recorded at seven flow-measurement locations where horizontal, broad-crested weirs with V-notch center sections were located in conjunction with highway bridge or culvert installations. Weir ratings were based on extensive stream discharge measurements at each site (Yates 1978, Mills et al. 1984).

Streamflow samples were collected at LRW flow-measurement locations from August 1974 through August 1978 for determination of total solids (TS) concentrations (Sheridan and Hubbard 1987). The 1974 to 1978 LRW data contained information on TS transported in streamflow from Coastal Plain watersheds, but information was not available for that period on the relative proportioning of the dissolved and suspended components. Data on SS were available from later environmental and water quality studies conducted on LRW from 1979-1981 (unpublished data). Although this

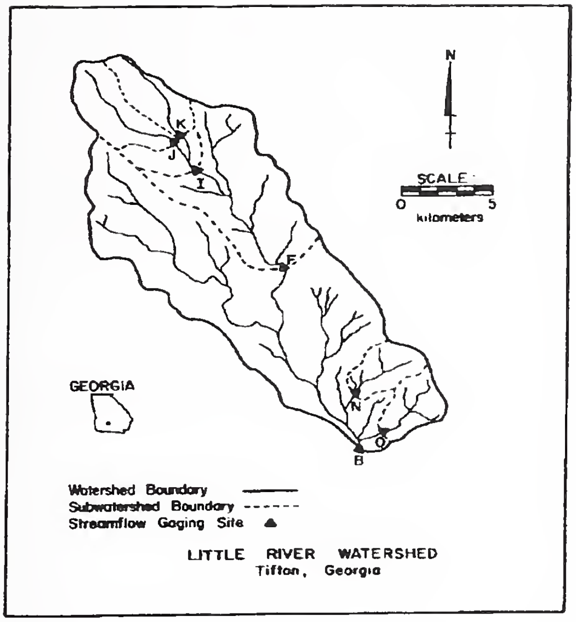

Figure 1. Map of Little River Watershed.

SS data was not from the same period, it was indicative of the magnitude of the suspended fraction of TS for these watersheds.

Total solids concentrations in runoff from the study areas for 1974-1978 averaged 96 +- 81 mg L^{-1}, with a maximum observed value of 859 mg L^{-1}. Total solids concentrations were examined by hydrologic seasons (March-May, June-August, September-November, December-February). The TS concentrations were significantly lower for March-May and December-February than for June-August and September-November. The March-May and December-February months also had the greatest per unit area runoff, indicating that dilution of transported solids by increased rainfall and runoff was the dominant effect.

The relationship between TS concentrations and rate of flow at the time of sampling was examined by plotting TS concentration vs. instantaneous flow rate converted to a per unit area basis (Figure 2). Correlation analysis showed that the relationship between TS concentrations and discharge rate was poor ($r = -0.16$, $p<0.0001$, $n = 1872$). Elevated values of TS concentrations were observed only at relatively low-to-moderate flow rates (Figure 2).

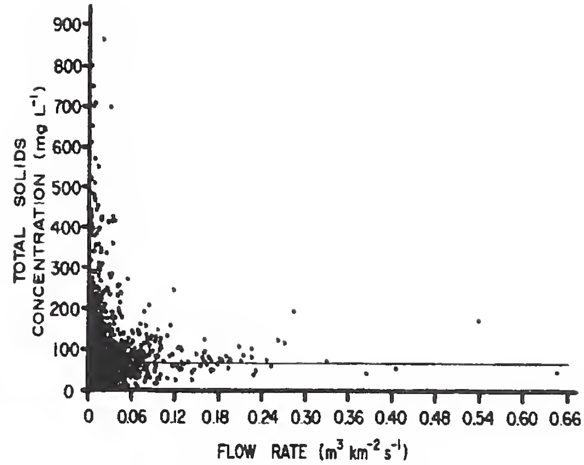

Figure. 2. Total solids concentrations vs. instantaneous per unit area flow rate.

The mean SS concentration in streamflow from LRW for 1979-1981 was 15 +- 20 mg L^{-1} (Sheridan and Hubbard 1987). Examination of the data by hydrologic seasons showed that average SS concentrations were significantly higher from March-August, when agricultural activity was greatest. A plot of SS concentration vs. instantaneous per unit area flow rate at the time of sampling is shown in Figure 3 for 1979 to 1981. Suspended sediments were poorly correlated with the instantaneous per unit area rate of flow ($r = 0.22$, $p <0.0001$, $n = 1608$).

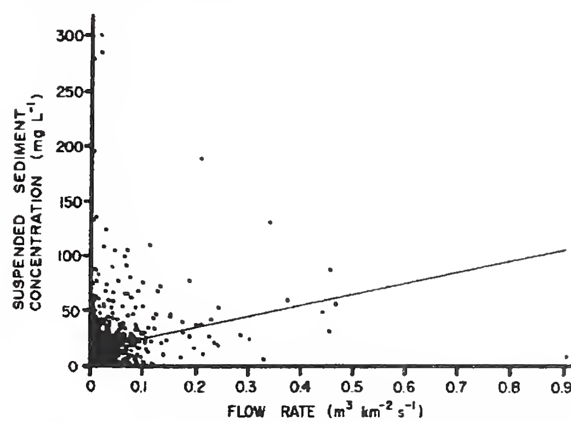

Figure 3. Suspended sediment concentration vs. per unit area flow rate.

The TS and SS concentrations for the LRW from the 1974-1981 study compared favorably with a limited number of measurements (U.S. Geological Survey 1973-1981) for a 1720-km^2 Coastal Plain drainage adjacent to the LRW (the Alapaha River near Alapaha, GA). Mean SS and dissolved solids concentrations for 17 streamflow sanples from the

Alapaha between January 1973 and August 1981 were 29 and 62 mg L^{-1}, respectively. The LRW SS concentrations were also compared to published SS data for Coastal Plain streams in Georgia (Perlman 1985). The average SS concentration for 1769 samples (33 sampling sites in seven river basins) for streams heading within the Coastal Plain physiographic region of Georgia was 13 mg L^{-1}. The low average SS concentration in streamflow from other watersheds in the Coastal Plain region indicated that the low SS concentrations observed on the LRW were typical of the region (Sheridan and Hubbard 1987).

Observed SS concentrations for 1984-1986 ranged from 1-137 mg L^{-1}. Mean SS concentrations from subwatersheds B, F, and K were 14, 17, and 14 mg L^{-1}, respectively. Based on Duncan's multiple range test (SAS Institute 1985), differences in mean SS concentrations between the three subwatersheds were not significant. This confirmed the earlier 1979-1981 finding that SS concentrations in this region of the Coastal Plain are generally independent of watershed size due to the characteristics of the watershed drainage and transport systems (Sheridan and Hubbard 1987). Suspended sediment concentrations showed only a slight increase with increasing per unit area instantaneous discharge.

Results

The long-term (30+ years) LRW mean daily streamflow records were analyzed to develop a regional flow duration curve (FDC). The FDC is a cumulative frequency curve that gives the daily mean discharge that is equaled or exceeded as a percentage of the total record period. The USGS statistical program SWSTAT (Flynn et al. 1995) was used to develop FDCs for the LRW watersheds. The FDCs were then normalized by the respective watershed mean annual flow (MAF) and averaged to produce a regional FDC.

The regional flow duration curve for the LRW with 95% confidence intervals is plotted in Figure 4. Flows ranged from approximately 26,620,000 L min^{-1} at the 1 % probability level to 5800 L min^{-1} at the 99% probability level. This means that based on analyses of 30+ years of flow records on the LRW, flows in excess of 26,620,000 L min^{-1} occur less than 1 % of the time, while flows in excess of 5800 L min^{-1} occur 99% of the time.

The upper confidence interval at the 1% flow probability level is flow in excess of 33,260,000 L min^{-1} while the lower confidence interval at the 1% flow probability level is flow in excess of 20,000,000 L min^{-1}. At the 99% flow probability level the upper confidence interval is flow in excess of 15,000 L min^{-1}. Calculated values for the lower confidence interval became negative by the flow probability level of 75% and hence are not plotted.

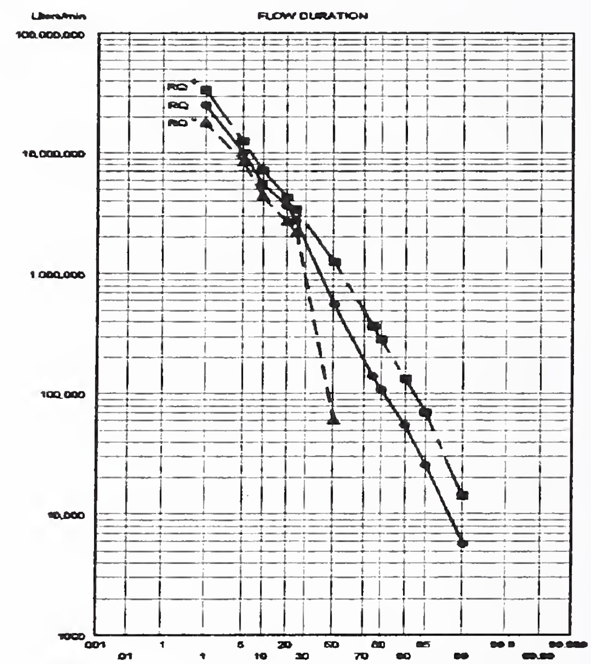

Figure 4. Regionalized flow duration probability curve with confidence limits.

Total daily loads (TDLs) were calculated using the regional flow duration curve by multiplying flow rate by mean SS or TS concentration. This in effect assigns a probability (% exceedance or risk) of exceedance of daily SS or TS loads in Coastal Plain stream systems.

For SS we multiplied the flow duration information by the mean value of 15 mg L^{-1}. For TS we used the mean value of 96 mg L^{-1}. We then converted the data to a TDL by summing for 24 hours. Table 1 shows the results for both SS and TS.

Calculated TDLs of SS ranged from 125 kg day^{-1} at RQ 99 to almost 575,000 kg day^{-1} at RQ1. Calculated TDLs of TS were 6.4 times as great. At RQ50 (mean annual flow rate) the SS load over a one year period computes to 736 mg yr^{-1}. This compares with the T-value (4 tons acre^{-1}) of 4494 mg yr^{-1}. In other words our computed annual losses based on our assumption of MAF rates and measured mean SS concentrations showed that SS loads for LRW would be only about 16% of estimated annual T-value losses. The TDL of SS, the pollutant of primary concern to environmental regulators, is quite low in southeastern Coastal Plain streamflow as compared to that of streams of other regions of the country.

Table 1. Total daily loads of suspended sediment and total solids for the Little River Watershed.

Flow Probability	TDL of SS	TDL of TS
	Kg day^{-1}	Kg day^{-1}
RQ1	574,992	3,679,976
RQ5	215,519	1,379,328
RQ10	133,188	852,409
RQ20	77,701	497,290
RQ25	61,256	392,004
RQ50	11,854	75,872
RQ75	3,022	19,346
RQ80	2,408	15,409
RQ90	1,203	7,698
RQ95	605	3,871
RQ99	125	798

Conclusions

A DD approach was used to develop probabilities of exceedance (1 to 99%) of daily loadings of SS and TS transported in streamflow from southeastern U.S. Coastal Plain watersheds. The DD was developed using a normalized regional flow duration curve and results of previous regional studies on concentrations of SS and TS from gauged agricultural watersheds in the Coastal Plain region.

Results indicate that this approach to developing daily stream loadings as required for TMDLs is effective for applications in the Coastal Plain region and provides probabilities of exceedance, or risk, information useful for water quality management. This approach provides land managers and regulators with a tool for predicting TDLs based on risk probabilities calculated from actual flow rates and sediment concentrations.

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Long-Term Effects of N Fertilizer on Groundwater in Two Small Watersheds

M.D. Tomer, M.R. Burkart

Abstract

Changes in agricultural management can minimize leaching of NO_3-N. But then how much time does it take to improve groundwater quality? This study was conducted in two small, first-order watersheds (30 and 34 ha) in southwest Iowa. Both were kept in continuous corn from 1964 through 1995, but one received large fertilizer-N applications, averaging 446 kg ha^{-1} y^{-1}, between 1969 and 1974. This study's objective was to determine if NO_3-N from these large applications persisted in groundwater. In 1996, transects of piezometer nests were installed, deep cores were collected, then water levels and NO_3-N concentrations were measured each month. In 2001, 33 water samples were collected and analyzed for tritium and stable isotopes. The watershed that received large N applications had greater NO_3-N concentrations in groundwater and stream baseflow, by 8 mg L^{-1}. Groundwater time-of-travel estimates and tritium data support persistence of NO_3-N from fertilizer applied 30 years ago. "Bomb-peak" precipitation (1963-1980) most influenced tritium activity in groundwater beneath toeslope positions, and deep groundwater was dominated by pre-1953 precipitation. Data from cores suggest NO_3-N may take 30 years to percolate to groundwater below the watershed's divide. Stable isotope data suggest runoff/infiltration processes contribute greater recharge and mixing of groundwater below the toeslope. Therefore historical and current practices affect NO_3-N concentrations in groundwater near the stream. It may take years to quantify impacts of management systems implemented in 1996 by monitoring groundwater. In many areas, changes in agricultural practices may take decades to fully impact groundwater quality.

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Keywords: nitrate, groundwater quality, BMP evaluations, tritium, stable isotopes

Introduction

Agricultural land use has been linked with loadings of nutrients to ground and surface waters. Agricultural practices can be modified to reduce these loadings through nutrient management, crop rotation, and other practices (e.g. Kitchen and Goulding 2001). Once a management change is made, the quality of runoff waters may improve rapidly, particularly if erosion is reduced. However, most streamflow is comprised of baseflow that originates from groundwater. The timing of groundwater responses to changes in agricultural management is difficult to predict and depends on many factors. Groundwater quality in the Midwest US has been affected by NO_3-N leaching (Burkart and Stoner 2001). As changes are implemented to address this problem, the time needed to improve groundwater quality must be better understood. If response times are underestimated, research on BMPs may be too short-term, causing groundwater quality benefits to be underestimated. Targeted timelines to achieve improved water quality may also be set too optimistically. This paper demonstrates the time frame that may be required to realize the full benefit of BMPs on groundwater quality.

This study took place in two watersheds (W1 and W2) of the Deep Loess Research Station (DLRS) in southwest Iowa (Figure 1; Karlen et al. 1999). Research at this site, begun in the 1960s, focused on erosion and nutrient balances under corn production (Karlen et al. 1998). Both watersheds were under continuous corn and conventional tillage from 1964 through 1995, and showed similar hydrology (Kramer et al. 1999). But experimental N-fertilizer applications occurred between 1969 and 1974, when W1 received an average 446 kg N $ha^{-1}y^{-1}$ and W2 received an average 172 kg N $ha^{-1}y^{-1}$. This experiment was used to assess NO_3-N movement in deep soils (Schuman et al. 1975, Alberts et al. 1977, Alberts and Spomer 1985), and in baseflow (Burwell et al. 1976). New crop rotations were established in 1996, when W1 was placed in a corn-soybean rotation, and W2 was placed in a six-year, contour-strip rotation with corn, soybeans, corn, and three years of alfalfa (Figure 1). With both rotations, only corn receives N fertilizer. The difference in current farming practices is hypothesized to cause a difference between the two catchments in groundwater and stream-baseflow NO_3-N concentrations. But if NO_3-N from large N applications from 1969-1974 persist in groundwater, it will be difficult to test this hypothesis directly. This study determined if these large N applications made 30 years ago could persist in groundwater and baseflow.

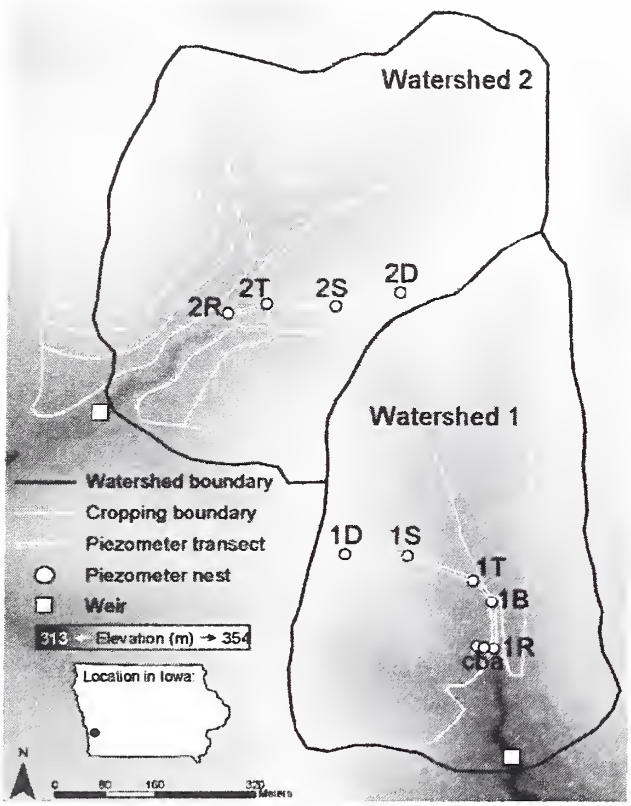

Figure 1. Map of watersheds W1 and W2, showing topography, piezometer-nest transects, weirs, and cropping boundaries.

Methods

In 1996, piezometer nests were installed in transects, at divide (D), mid-slope (S) toe-slope (T), and riparian valley (R) positions of both watersheds. These nests are identified by watershed number and position (Figure 1). Additional riparian piezometers were installed in W1 in 1999, and are denoted as 1B, 1Rb and 1Rc (Figure 1). Piezometer-nests comprise three transects, identified by watershed (W1, W2 and $W1_{rip}$ for the riparian transect in W1), that each portray an expected path of groundwater flow. During drilling, cores were taken to the depth of glacial till, which is an aquitard limiting the downward flow of groundwater beneath the deep loess. Cores were analyzed for bulk density and NO_3-N. Materials encountered during coring included loess at upland (D and S) positions, and alluvium from reworked loess at lower (T and R) positions. Water levels and groundwater NO_3-N concentrations were measured on a monthly basis. Three lines of evidence helped to determine if past N applications have had a persistent effect on groundwater nitrate. These lines of evidence were based on hydrologic measurements, spatial trends in isotope chemistry, and spatial and temporal trends in NO_3-N concentrations in water and deep sediments.

The hydraulic conductivity (K_s) of the saturated zone was measured by conducting slug tests, and a geometric mean K_s value was calculated for each type of deposit (till, sand at the till interface, loess, and alluvium). Positional survey data, coring descriptions, water levels, and K_s values were used with the Darcy equation to estimate groundwater travel times along the three transects. Hydraulic gradients were based on water levels for June 2001 and April 2002, when the highest and lowest average water levels were observed. An effective porosity of 0.2 was applied to calculate pore velocities, which were divided into the transect distances to obtain travel time estimates. Total porosities averaged 0.42, and an effective porosity of about half this value was selected to represent the mean (center of mass) movement of a solute plume through the groundwater.

In June 2001, 33 water samples were collected for isotopic analyses of $\delta^{18}O$, $\delta^{2}D$, and ^{3}T. Samples were taken from 30 piezometers, from stream baseflow passing the weir of each watershed, and an aggregate sample was collected from four, 1.8-m-depth suction lysimeters to represent recent precipitation. A record of annual tritium activities in precipitation was constructed to represent the age distribution of tritium activities (TU) in groundwater recharge at the site. We obtained TU data for annual precipitation from IAEA (1992), IAEA/WMO (2001), and Simpkins (1995). Records from monitoring stations at Lincoln NE, St. Louis MO, and Ottawa ONT were used to compile a continuous tritium input record from 1953 to 1999. Correlations between these stations, and a time-trend of local data provided two methods of estimation, which gave good agreement. Once these precipitation records were constructed, the expected tritium activity of each year's precipitation in 2001 was calculated, based on a half-life of 12.43 years (Gonfiantini et al. 1998).

Results and Discussion

Estimated groundwater velocities between piezometer nests averaged 13.5 m y^{-1}, and varied from 5.3 to 27.1 m y^{-1}. Travel time estimates (Table 1) varied according to the length of each transect and changes in hydraulic gradients. Along the W2 transect, larger gradients occurred with higher water levels measured in June 2001. But along W1, larger gradients occurred during April 2002. Between 64 and 82% of the groundwater travel times occurred above the toeslope positions. Travel times are considered conservative because vertical and horizontal gradients are present, causing actual travel distances to be greater than horizontal distances.

Table 1. Transect lengths and estimated groundwater travel times using hydraulic gradients from two sets of measurements.

Transect	Length (m)	Travel time (y) based on water levels from: Jun 01	Apr 02
W2	277	22.7	25.9
W1	337	36.1	31.0
$W1_{rip}$	28	1.4	2.5

When isotopic decay was applied to constructed tritium-precipitation records, expected residual tritium activities in 2001 indicated that: 1) waters that fell as precipitation within 20 years of sampling cannot be differentiated; 2) water with ages between 20 and 40 years would be indicated by tritium activities exceeding 12 TU; and 3) water at least 45 years old is indicated by small tritium activities (<3 TU). Tritium activities of the 33 water samples ranged from 0.8 to 18.5 TU. Small tritium activities (< 3 TU), indicating the oldest water, occurred in six of the 33 samples and were always in the deepest groundwater (Figure 2). Tritium values exceeding 12 TU, showing an influence of 20-40 year old precipitation, occurred in 9 of the 33 samples. In groundwater these larger tritium activities always occurred at midslope (1S, 2S) and at or near toeslope (shallow at 2T, 1Rc) positions (Figure 2). Values of intermediate tritium activity (3-12 TU) were typically found at or below toeslope positions (1T, 1B, 1Ra, 1Rb, deep at 2T, 2R), indicating recent or mixed-age waters. Samples collected from the weirs had tritium activities of 11.1 and 12.5 TU for Watersheds 1 and 2, respectively, which are considered similar. The baseflow is probably of mixed origin, with a weak influence of 20-40 year-old waters.

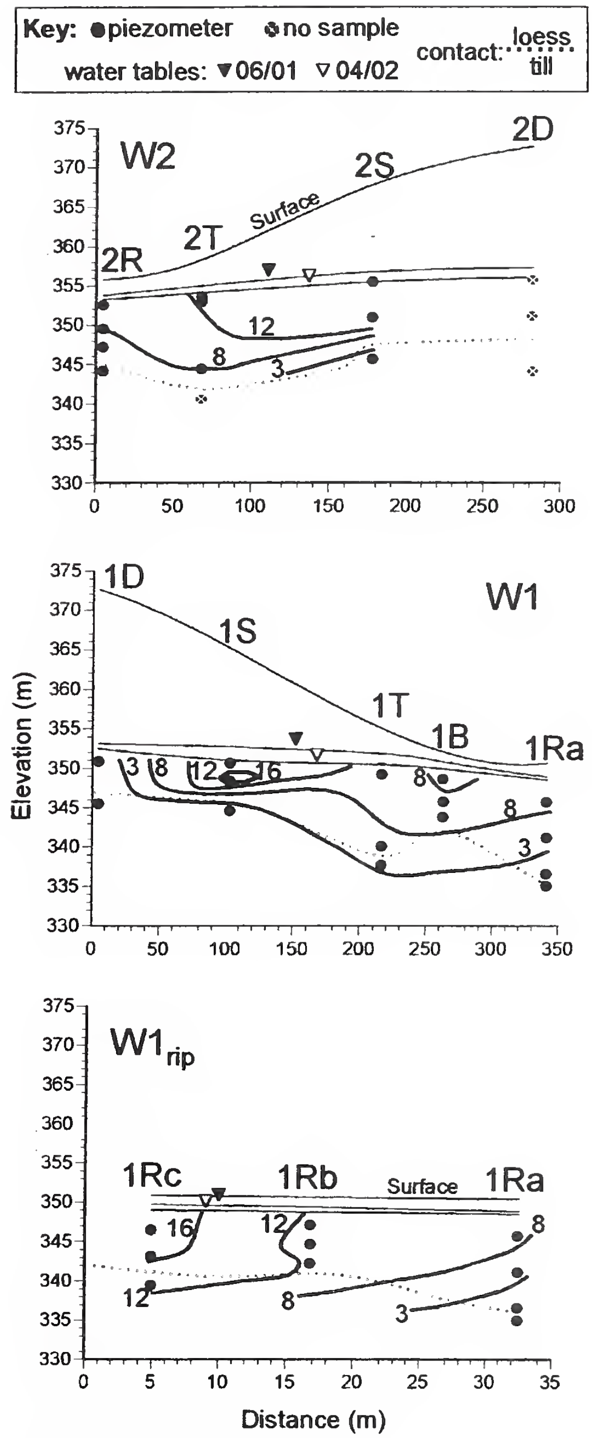

Figure 2. Cross sections showing variations in groundwater tritium activity (TU) along three transects during June 2001.

Stable isotopes also showed noteworthy results. The average $\delta^{18}O$ was –7.1‰ (range –5.6 to –8.9‰) and the average $\delta^{2}H$ was –49.2‰ (range –39.6 to –60.9‰), similar to other Midwest data (Simpkins 1995, IAEA 1992). Groundwater contains mixed waters with fairly small isotopic variability. But despite their limited variation, the isotope data did follow a distinct spatial pattern. Groundwater beneath the upper landscape positions (D and S) was isotopically enriched compared to toeslope (T) or riparian valley (R) positions ($p < 0.01$, based on a t-test). This indicates differences in processes or sources affecting groundwater according to landscape position. Probably, seasonal changes in runoff and infiltration act to segregate recharge waters according to landscape position. Snowmelt and cold spring rains are isotopically depleted, and occur when there is little plant cover. These depleted waters would be most prone to runoff from upper slopes and then infiltrate near the toeslope. This means that riparian valley groundwater is affected by recent precipitation, as well as groundwater being contributed from upslope.

A large increase in sediment NO_3-N concentrations (ug $^{-1}$) was observed in the deep cores obtained during 1996 at position 1D (Figure 3), centered near 17.5 m depth. This pattern was not observed in any of the other deep profiles, and only 25 out of 323 samples collected at the other positions showed NO_3-N concentrations exceeding 4 ug g^{-1}. This 17.5 m depth, when plotted with depths of peak NO_3-N concentrations from cores collected from Watershed 1 during 1972, 1974-76, 1978, and 1984 (Schuman et al. 1975, Alberts et al. 1977, Alberts and Spomer 1985), shows a strong linear relationship ($r^2 = 0.98$) with cumulative baseflow since 1969, when the large N applications in Watershed 1 began. (Figure 3). This relationship suggests that NO_3-N from these large applications moved through the unsaturated zone in response to hydrologic fluxes through the watershed's subsurface. Given the inferred movement of this NO_3-N pulse to depth, one would anticipate that, at lower-elevation landscape positions, this NO_3-N would have percolated into the saturated zone before this 1996 core sampling. This would explain why large sediment concentrations were not observed at depth at lower W1 positions. Also, at position 1D, one would expect increases in groundwater NO_3-N as this NO_3-N pulse moved from the sediment into groundwater (Figure 3). Such an increase was observed and its timing was consistent with baseflow volumes measured during the late 1990s.

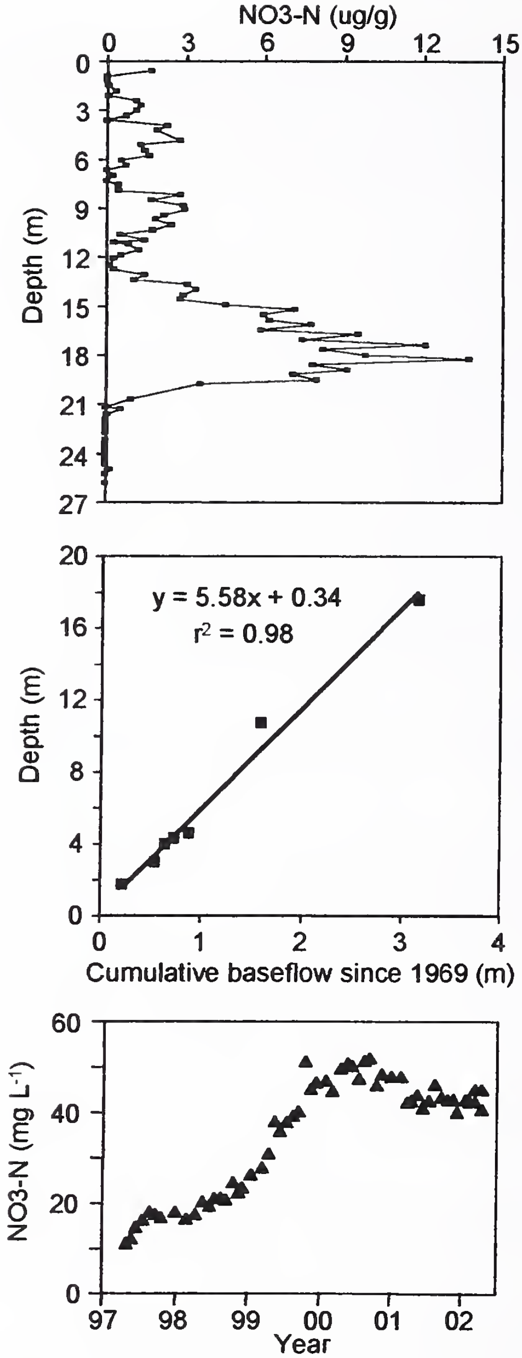

Figure 3. Top: Concentrations of NO_3-N in sediment with depth at position 1D measured in cores taken in 1996 (top). Middle: Depths of maximum NO_3-N concentrations, as measured in 1972, 1974-1976, 1978, 1984 (Alberts et al. 1975; Schuman et al. 1985), and 1996, are related to stream baseflow since large fertilizer N applications. Bottom: Increases in NO_3-N in the underlying water table were consistent with continued percolation of this pulse.

Concentrations of NO_3-N in baseflow were greater at the outlet of W1 ($p < 0.01$, based on a paired t-test), where an average of 20 mg L^{-1} contrasted 12 mg L^{-1} from W2. Three independent lines of evidence (groundwater travel, tritium, and N in sediment) suggest this difference could, at least in part, be due to large N applications from 1969 to 1974.

Conclusion

Groundwater time-of-travel estimates and tritium data both suggest that groundwater remains resident in these watersheds for more than 30 years. Furthermore, analyses of sediments from deep cores show that soil NO_3-N may take 30 years to reach groundwater in upper parts of these watersheds. Concentrations of NO_3-N in groundwater beneath upslope positions in Watershed 1 are still influenced by N applications made from 1969 to 1974. Stable isotope data suggest that in lower landscape positions, historical and recent land use practices affect current NO_3-N concentrations. It will be difficult to discern impacts of recent cropping changes between W1 and W2 by simply monitoring groundwater or baseflow. Monitoring of shallow unsaturated-zone waters may be the most reliable means to do this. Multiple lines of evidence suggest it takes at least several decades for subsurface water to travel from the divide to the stream. In many locations, changes in agricultural practices may take decades to fully affect improvements in groundwater quality.

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A Study on Nitrogen Emission from Agricultural Lands and Eutrophication on Nearby Surface Waterbody as an Impact

Mohammed Hossain, Md. Omar Faruk, Md. Waji Ullah, Ahmadul Hassan

Abstract

The use of agricultural fertilizer has been increasing to boost up the agricultural production to ensure the food security of large growing population of Bangladesh. The increased fertilization results into continuous degradation of water quality of the nearby water bodies. A study was conducted to determine the emission of nitrogen from agricultural lands and its impact as eutrophication using analytical techniques. The hydro-meteorological data were collected from the nearest meteorological station and water quality of the waterbody on certain parameters have been monitored by using DataSonde 4a, a water quality measuring instrument. The water and material balance approaches in combination with Geographic Information System (GIS) have been used to determine the agricultural runoff and thereby the amount of nitrogen released from the agricultural lands which is ultimately added into the waterbody. The study result reveals that about 5.35% (about 9 kg/ha) of nitrogen has been released from agricultural lands against an application of 356 kg/ha of nitrogen in a year. The nitrogen and phosphorus content in water, low N/P ratio, low secchi depth and plankton species diversity indicate that the water body remains eutrophic in nature throughout the year.

Keywords: nitrogen emission, agricultural lands, GIS

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Runoff and Sediment Losses from Annual and Unusual Storm Events from the Alto Experimental Watersheds, Texas: 23 Years after Silvicultural Treatments

Matthew McBroom, R. Scott Beasley, Mingteh Chang, Brian Gowin, George Ice

Abstract

Evaluating the potential impacts of intensive silvicultural practices on water quality is critical for establishing the long-term sustainability of contemporary forest management practices. From 1979 to 1985, a study involving nine small (~2.5 ha) forested watersheds was conducted near Alto, Texas in the upper western Gulf-Coastal Plain to evaluate the impacts then-current silvicultural practices on water quality. In the years following the study, silvicultural Best Management Practices (BMPs) including Streamside Management Zones (SMZs) and other erosion control practices evolved and questions arose about the applicability of earlier results to current practices. In 1999, these same watersheds were re-instrumented to evaluate the water quality effects of intensive silviculture using modern BMPs. Three years of pre-treatment data were collected to calibrate the watersheds. During the calibration phase, in June 2001, Tropical Storm Allison struck southeastern Texas, dumping almost 11.8 cm of rainfall on saturated soils in about 3 hours. This single storm event resulted in over 73% of the annual flow and over 95% of the annual sediment for 2001. In a little over three hours, the watersheds clearcut and chopped in 1980 generated over 2.5 times more sediment that the entire year following harvest and site-preparation. Comparisons of data from the 1979 Alto Watershed study with pre-treatment data from the current study suggest that these watersheds have a high potential for geologic erosion even with mature forest cover. Large natural variation in runoff and sediment makes it difficult to detect treatment effects for these forested watersheds.

Keywords: stream flow, water quality, sediment, silviculture, non-point source pollution, best management practices

Introduction

Concerns exist in the public policy arena about the potential effects of silvicultural non-point source pollution on water quality (USEPA 2000). Forested watersheds are generally associated with better water quality than waters draining watersheds of other major land uses (USEPA 1995). However, forest practices such as harvesting may temporarily increase sediment losses from harvested areas (Beasley et al. 2000). Detecting the impacts of forest practices on water quality is complicated by great spatial and temporal variability in runoff from forested watersheds. For example, stream sediment losses from undisturbed forested watersheds in the South are reported to range from a trace up to 717 kg/ha/yr (Yoho 1980).

During the 1970s and 1980s, a series of watershed studies were conduced to measure the effects of forest practices in the mid-South on non-point source pollution (Beasley 1979, Blackburn et al. 1986, Miller et al. 1988, Ursic 1986). A common forest practice at that time involved clearcutting naturally regenerated, mature, mixed pine-hardwood stands and replanting with genetically improved loblolly pine (*Pinus taeda*). Harvest efficiency (wood utilization) was not as high as

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today, with less topwood and poorer species utilization. As a consequence, management of large quantities of residual logging slash required mechanically intensive site preparation methods such as shearing, windrowing, and burning or roller-chopping and burning to prepare sites for replanting. Furthermore, BMPs at that time did not call for SMZs on many smaller headwater streams, so sites were prepared and planted across these streams.

Stands regenerated in the 1970s and early 1980s are now being harvested. Clearcutting of these stands results in very little residual logging slash due to better harvest efficiency and stand homogeneity. Therefore, mechanical treatments such as shearing and windrowing are used less extensively. Contemporary practices involve increased use of herbicides and fertilizers. BMPs now include SMZs on intermittent and many ephemeral streams, and these types of BMPs have been shown to effectively reduce silvicultural impacts on water quality (Lynch and Corbett 1990, Arthur et al. 1988). Williams et al. (1999) reported a ten-fold decrease in sediment from logging with BMPs when compared to logging without BMPs. According to Carraway et al. (2002), BMPs were voluntarily implemented on about 91.5% of silvicultural operations in Texas.

The Texas Intensive Silviculture Study (TexIS) was initiated in 1999 to evaluate the effects of contemporary silvicultural practices on water quality in small headwater catchments. Nine small (~2.5 ha) watersheds that were studied by Blackburn et al. (1986) were re-instrumented. This affords a unique opportunity to contrast water quality effects of contemporary intensive forest practices with practices from two decades ago.

This paper will examine the water yields and sediment losses measured from 1999 through 2001 on these forested watersheds. Comparisons will be made with data collected from 1980 through 1985 on these watersheds following clearcut harvesting and site preparation. An extreme hydrologic event, Tropical Storm Allison, will also be examined and an overall comparison with pre-and post-harvest conditions will be conducted.

The Alto Experimental Watersheds

Nine small watersheds (2.57 to 2.72 ha) in the Neches River Watershed, about 16 km west of Alto in southwest Cherokee County in East Texas, are used in this study. This area has a humid, sub-tropical climate with hot summers and cool winters. The average annual rainfall of 117 cm is fairly evenly distributed throughout the year, with April and May receiving a little more rainfall than other months. Topography is dominated by rolling hills, with watershed elevations ranging from 90 to 131 m and watershed slopes ranging from 4 to 25%. Mean channel slopes are about 19%, indicative of ephemeral or intermittent headwater reaches. Soils developed under mixed loblolly pine and hardwood forests from marine sediments of the Queen City Sand of the Eocene Epoch. Soils tend to be light-colored, well-drained, erodible, and generally have low fertility. Cuthbert and Kirvin soils (clayey, mixed, thermic typic Hapludults) compose about 78% of the watersheds.

Fully-stocked, unthinned loblolly pine plantations planted in 1981 dominated 6 of the 9 watersheds. The other 3 had younger plantations. Watersheds had full canopy closure with almost 100% soil coverage by forest vegetation and forest litter layers.

The original study reported by Blackburn et al. (1986) employed a randomized block design, with three watersheds being clearcut with shear, windrow and burn site-preparation (SW1, 2, 3), three being clearcut with roller-chop and burn site-preparation (SW5, 7, 9), and three serving as controls (SW4, 6, 8). The six treatment watersheds were clearcut in the summer of 1980, with site-preparation and planting occurring that same year. Measurement of stream flow and sediment continued until 1985, after which these watersheds were decommissioned.

In 1998, these same watersheds were re-instrumented. The concrete approach sections that were constructed on these watersheds in 1979 were refitted with 3-foot H-flumes. Stage recorders were installed, along with Coshocton wheel samplers. Stage is measured with a potentiometric float and pulley level recorder in the stilling well at the sidewall of the flume. A datalogger stores stage measurements in 5-minute intervals and initiates an ISCO autosampler during storm-runoff events. Water samples are collected with a 3-foot float arm, with the sampler intake at its midpoint. This allows for sample collection at 0.5 stream depth regardless of stage. Samples are taken at approximately a 30-minute interval during runoff events to represent the phases of the storm hydrograph. Samples are pumped into 1-L polypropylene bottles.

Samples are collected and taken to the laboratory immediately after each event for analysis of suspended solids using the gravimetric method. Immediately upstream of the concrete approach section is a sediment trap. Bed-load deposited in the sediment trap is collected and dry-weight determinations of bed-load are made after storm events as well. Total sediment loss is suspended sediment concentration from each ISCO sample bottle or composite (mg/L) times total flow for the representative time interval (L) plus bed-load (kg). This value is then divided by watershed area (ha) to get kg/ha of sediment loss. Sediment data were not available for part of one event on 01/29/1999, so sediment concentrations were estimated using regression with streamflow. Total rainfall is measured with a series of National Weather Service standard 8-inch non-recording rain gauges at each watershed, and tipping-bucket recording rain gauges are used for determining storm intensity and duration.

Annual Runoff

Precipitation

In the original Alto Watershed study, the pre-treatment year (1980) was an abnormally dry with only 79.1 cm of precipitation, almost 38 cm less than average (Blackburn et al. 1986). The first post-treatment year was wetter than average, with 129.8 cm, almost 13 cm greater than average. The next 3 years annual precipitation was close to average.

In the current watershed study, during the first year of the pre-treatment period, 1999, only 82.6 cm of rainfall were recorded, or 34.4 cm less that the long-term average (Table 1). The year 2000 was wetter; though most of the storms were low-intensity, long-duration events, resulting in little runoff. The year 2001 was very wet, with 170.6 cm of precipitation, or 53.6 cm greater than average.

Water yield

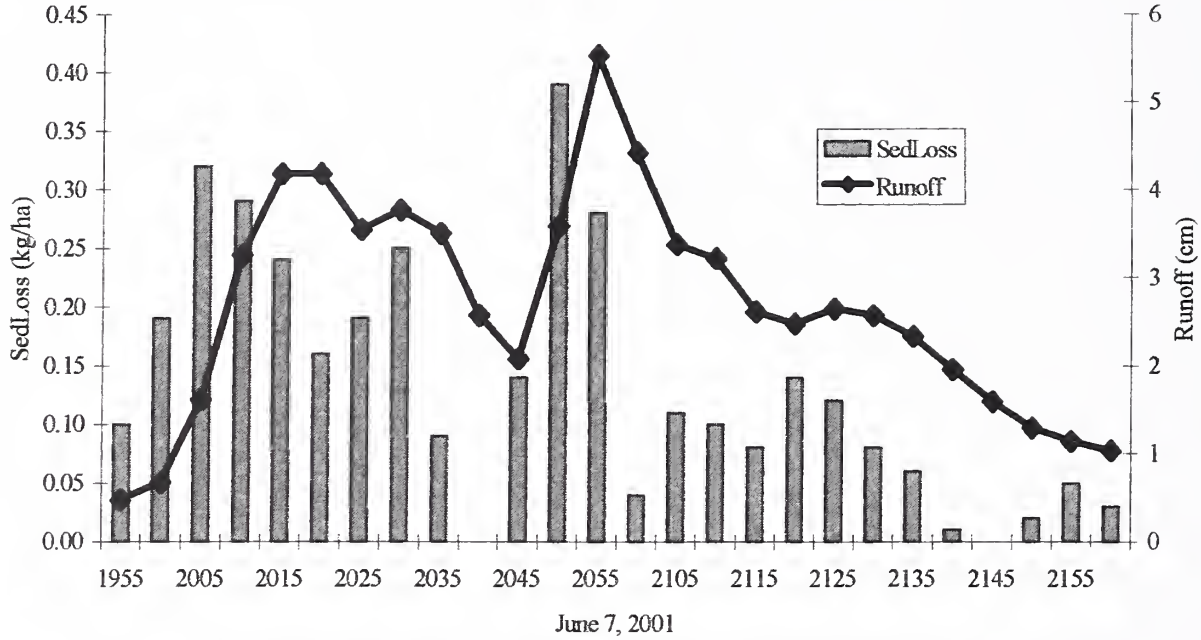
Water yield in these headwater streams is dominated by storm runoff, with baseflow making only a minor contribution to total annual water yield. Most runoff tends to occur during the winter and spring, when antecedent soil moisture is highest due to low evapotranspiration demand. Watershed runoff efficiencies tend to be low, with most precipitation being retained by vegetation and soils.

Prior to treatment in 1980, Blackburn et al. (1986) reported about 2.2 to 6.1 cm of runoff from these watersheds. 1980 was an abnormally dry year, with annual runoff efficiencies ranging from 4.1 to 5.6%. After treatment, first-year stormflow was greater from sheared (14.6 cm, or 11.2% of rainfall) than chopped (8.3 cm, or 6.4% of rainfall) or control (2.6 cm, or 2.0% of rainfall) watersheds. Streamflow returned to pretreatment levels by the third year after harvest.

Annual water yields measured in 1999 to 2001 tended to be comparable to those measured in 1980 (Table 1). Highest water yields were recorded in 2001, the year with the greatest annual rainfall. Rainfall efficiency was below 1% in 1999 and 2000, and between 3 and 5% in 2001. This higher efficiency was due to the effects of an extreme storm event as discussed below.

Table 1. Mean pretreatment rainfall, runoff, sediment loss, and flow-weighted sediment concentration by year and for Tropical Storm Allison by watershed treatment block* for the Alto Watersheds.

Block	Year	Rainfall (cm)	Runoff (cm)	Sed Loss (kg/ha)	Flow-weight conc. (mg/L)
	1980	79.12	3.31	115.33	348.4
Shear Block	1999	82.60	3.28	85.2	161.4
	2000	126.29	0.99	3.5	32.1
	2001	170.64	9.24	293.0	257.9
	Allison	11.81	6.44	277.7	349.3
	1980	79.12	4.42	28.66	64.8
Chop Block	1999	82.60	2.49	26.4	40.6
	2000	126.29	0.53	0.2	4.5
	2001	170.64	6.46	67.3	72.8
	Allison	11.81	4.66	66.4	98.4
	1980	79.12	3.45	62.19	180.3
Cntrl Block	1999	82.60	2.95	52.2	92.7
	2000	126.29	0.64	1.0	14.9
	2001	170.64	7.10	241.8	260.5
	Allison	11.81	5.81	227.0	288.4



* Watersheds are blocked by Blackburn et al. (1986) treatments; shear/pile windrow burn = SW1, 2, 3; roller chop and burn = SW5, 7, 9: control = SW4, 6, 8.

Figure 1. Runoff and sediment losses for small watershed 2 for Tropical Storm Allison for the period of most intense rainfall on 06/07/2001 (total rainfall = 16.3 cm) for the Alto Watersheds.

Sediment losses

Total sediment loss

Prior to the first commercial clearcut of these watersheds in 1980, sediment losses ranged from 55 to 530 kg/ha/yr. Pre-treatment sediment loss rates at Alto are higher than those measured in some other studies in the southeast following harvest. For example, Miller et al. (1988) reported mean first-year sediment losses of 237 kg/ha after clearcutting and site preparation in the Ouachita Mountains. In the Gulf Coastal Plain, streambeds and banks are comprised mostly of deposited sediments causing stream channels to be more susceptible to erosion.

After treatment, Blackburn et al. (1986) measured the highest first-year sediment loss rates yet reported in the western Gulf Coastal Plain, with 2,937 kg/ha/yr on the sheared and windrowed watersheds, 25 kg/ha/yr on the roller-chopped, and 33 kg/ha/yr on control watersheds. Clearcutting followed by shearing, windrowing, and burning resulted in 56.8% bare soil, thus increasing overland flow and erosion potential. For comparison, roller chop site-preparation resulted in only 15.7% bare soil while the control watersheds had 3.3% bare soil.

In the current study, annual sediment losses varied during the three pre-treatment years depending on precipitation (Table 1). As discussed below, extreme storm events were the controlling factor for total annual sediment yield. Pre-treatment sediment losses measured in the current study were comparable to those reported by Blackburn et al. (1986). Pre-treatment loss rates were nonetheless higher in 1980 on watersheds 2 (514 kg/ha) and 3 (313 kg/ha) than those reported by Miller et al. (1988) with 237 kg/ha and Beasley and Granillo (1988) with 264 kg/ha after clearcutting and site-preparation.

Sediment loss timing

Blackburn et al. (1986) measured the greatest sediment concentrations during the peak phase of the hydrograph, and the same was generally true for this study. For larger events, when more bottles were analyzed along the hydrograph, the greatest sediment loss rates were observed at or just prior to the peak of the hydrograph (Figure 1). During periods of higher flow, as streams approach bank-full, more water is in contact with the erodible channel cross-section, resulting in greater sediment loss rates even in undisturbed forested watersheds. Also, stream velocities are highest during this phase of the storm event, increasing the energy available for sediment detachment and transport from channels.

Unusual Storm Events

In the first Alto study, one relatively large storm event on 05/15/1980 produced most of the sediment loss and stream flow for the pre-treatment year. This single event (8.2 cm rainfall) resulted in 96 to 99% of the total annual pre-treatment sediment loss and 59 to 79% of the total annual streamflow. About 7.3 cm of rain fell during the 3 days prior to this event, saturating soils. Based on the 94-year precipitation record at Nacogdoches, Texas, located approximately 64 km east of the study site, this 1-day event would have about a 2-year return period (Chang et al. 1996).

The largest pre-treatment storm event for the current study was Tropical Storm Allison, which produced almost 30 cm of rain over a 3-day period from 06/05/2001 through 06/07/2001. Almost 10 cm of rain fell up to 06/07/2001, saturating soils, and during late afternoon of 06/07, 11.8 cm of rainfall fell over a 4-hour period with a maximum 1-hour intensity of 6.55 cm. This 3-day storm event had over a 100-year return period, with a maximum 1-day return period of about 20 years.

Tropical Storm Allison resulted in 71 to 76% of the annual streamflow and 92 to 98% of the total annual sediment loss for 2001 (Table 2). Allison resulted in more sediment loss than occurred in the entire post-treatment year for the clearcut and roller-chopped watersheds in 1981 and for the second-year losses for the sheared watersheds in 1982. Other studies in the Southeast have found that single storms often account for the majority of total annual sediment loss (Beasley 1979, Beasley 1984). Similarly, for the current study, a single storm on 01/28/1999 with about a 10-year 1-day return period resulted in over 95% of the total annual sediment loss (Table 2). In 2000, precipitation was more evenly distributed, yet about one-quarter to one-half of the annual sediment loss occurred as a result of a runoff event on 05/04/2000. This rainfall event had less than a 2-year 1-day return interval.

Discussion and Conclusions

The reactivated Texas Alto Watershed Study provides a unique opportunity to evaluate effects of contemporary silvicultural practices on water quality. Results from the calibration phase of the current study are consistent with pre-treatment results from the original Alto study in 1980, indicating that these watersheds have recovered from harvesting and site-preparation. Blackburn et al. (1986) found that treatment effects were greatly diminished by the second post-harvest year. Vegetative regrowth is rapid in the southeastern United States due to warm temperatures and evenly distributed annual rainfall, resulting in shorter recovery times than are observed in cooler, dryer regions. Storm flows and sediment losses from undisturbed watersheds were low, though losses at Alto were higher than those reported in other studies following harvest and site-preparation. Overland flow is rarely observed in these watersheds, with flow responses most likely dominated by subsurface macro-channel flow and variable source areas (Beasley 1976).

Table 2. Percentage of annual flow and sediment loss accounted for by the largest storm event in each pretreatment year by watershed treatment block* for the Alto Watersheds.

Watershed	Year	% Annual Rainfall	% Annual Runoff	% Annual Sed Loss
	1999	15.2	97.1	98.2
Shear	2000	7.9	27.0	56.9
Block	2001	6.9	70.8	94.1
	1999	11.9	92.4	95.3
Chop	2000	3.5	49.1	38.9
Block	2001	6.9	75.5	98.5
	1999	15.2	96.9	98.8
Cntrl	2000	4.8	37.2	26.9
Block	2001	6.9	73.4	92.0

* Watersheds are blocked by Blackburn et al. (1986) treatments; shear/pile windrow burn = SW1, 2, 3; roller chop and burn = SW5, 7, 9; control = SW4, 6, 8.

Most annual runoff and sediment loss results from a few larger events that occur during periods when soils are near saturation. Such storms may have a fairly high probability of occurrence, as was the case on May 15, 1980 when a 2-year 1-day storm generated 96 to 99% of the annual runoff on these watersheds. For the current study, a 10-year 1-day storm on January 28, 1999 resulted in over 90% of the annual sediment loss. Tropical Storm Allison had about a 20-year 1-day return period, and resulted in more sediment loss than was observed for the entire year after clearcutting and roller-chop site preparation on these same watersheds in 1981.

Since there is no evidence of sheet-flow, head-cutting, or gullying in these watersheds, it is reasonable to conclude that elevated sediment loss rates for forested watersheds results from erosion of channels that are incised into erodible marine sediments. Little rock- armoring is present, leaving channels exposed to erosive forces. Rock cover in streams can be a factor that reduces sediment loss rates in streams draining forested watersheds (Beasley et al. 2000). Increased flow rates result in increased channel erosion and thus higher sediment concentrations in stormflow.

Great variability is observed in runoff and sediment loss rates in these forested watersheds. These data from undisturbed watersheds highlights the need for non-point source pollution assessment and management strategies that reflect the temporal and spatial variability inherent in nature.

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Sources and Estimated Load of Bioavailable Nitrogen Attributable to Chronic Nitrogen Exposure and Changed Ecosystem Structure and Function

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Abstract

Bioavailable nitrogen is a limiting nutrient throughout the Eastern United States. Research demonstrates that exposure to large doses of nitrogen leads to deleterious environmental impacts. However, effects of chronic exposure to lower doses of nitrogen are not well known. Since 1998, we conducted an integrated multi-disciplinary study investigating ecosystem feedback associated with chronic exposure to low doses of nitrogen. In this nitrogen-limited ecosystem, the ability of the soil system to adapt to new nitrogen inputs was degraded by the first winter season when soil nitrate concentration (indicated by extract concentration) increased 590%. Increased concentrations of nitrate were especially evident on fertilized plots ($P<0.0001$) but also were measured on plots that were only fenced to manipulate herbivore abundance ($P=0.0443$). Changes to the plant, macro invertebrate, herbivore, and microbial communities each reduced ecosystem nitrogen use efficiency thereby producing a self-reinforcing positive feedback loop leading to conditions favoring greater concentrations of soil nitrate. Because all of these effects occur together, it is unclear whether the results seen in soil nitrogen chemistry are from one or some combination of them working together. What is clear however is that changes to trophic feedbacks associated with chronic exposure to low levels of bioavailable nitrogen resulted in an annual mean nitrogen excess of 105.6%. Although it was not our goal to measure mass balance for all nitrogen loss pathways, we estimate that ecosystems similarly exposed to excess or new sources of nitrogen may be at risk and are capable of leaching essentially all new nitrogen additions during seasons of dormancy. These experiments demonstrate that even the relatively small amount of nitrogen deposited in precipitation has the capacity to change multiple aspects of ecosystem structure and function.

Keywords: atmospheric deposition, bioavailable nitrogen, nitrate, ecosystem response, ecosystem management, trophic interactions

Introduction

Bioavailable nitrogen has been falling in the rain for decades in developed areas (e.g., Smil 1990, Vitousek et al. 1997). This is expected to continue indefinitely (Brimblecombe and Stedman 1982, Galloway et al. 1994, Vitousek et al. 1997). Only recently has attention been focused toward risks associated with low level exposure to nitrogen, such as that accompanying atmospheric deposition (Likens 1992). Nitrogen, particularly nitrate, easily moves from terrestrial ecosystems into surface and groundwaters, including lakes, streams, rivers, and estuaries (e.g., Baker 1992, Kahl et al. 1993, Peterjohn et al. 1996). To weigh risks and assess management options, it is important that a thorough understanding of the interactions and transport of nitrogen in terrestrial and aquatic

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ecosystems, and the atmosphere be developed (Jorgensen 2002).

The nitrogen cycle is well studied. Although many of the cycle's components are important to consider for nitrogen management, there are relative few that interact closely with atmospherically deposited nitrogen. In this study, we are most interested in those components that are directly affected by nitrogen deposition. Wedin and Tilman (1996) published data demonstrating that exposure to excess bioavailable nitrogen degrades ecosystems in a number of notable ways: 1) retained nitrogen decreases with increasing exposure, 2) biodiversity declines, and 3) plant C/N ratio declines. Of perhaps greater importance is the observation that most of the ecological response measured by Wedin and Tillman (1996) occurs in the first 100 kg/ha/yr of deposition. We call this the 'ecologically significant dose' (Figure 1).

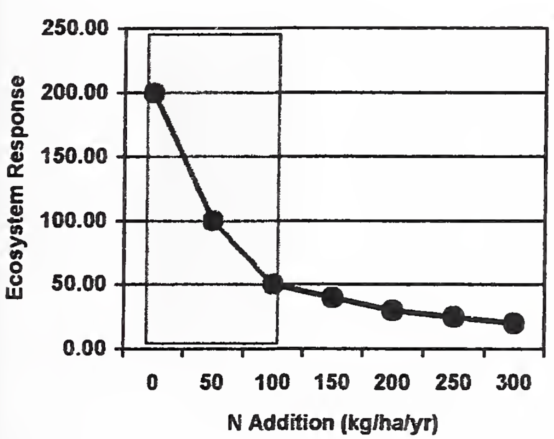

Figure 1. Data adapted from Wedin and Tilman (1996) demonstrating relationship between several ecosystem characteristics and nitrogen dose. Because most of the ecological response occurs with the first 100 kg of load, we call this the 'ecologically significant dose.'

Whereas atmospheric deposition in the Eastern United States is measured at approximately 10-30 kg/ha/yr NO_3 (National Atmospheric Deposition Program (NRSP-3)/National Trends Network 2002), it is evident that ecosystems throughout this region may be affected by exposure to atmospheric deposition of bioavailable nitrogen.

Ecosystems process deposition in a few ways. Deposition to terrestrial systems may be processed by biota (i.e., plants and/or microbes), escape to aquatic systems through runoff or percolation, or volatilize. Improved nitrogen management will occur where the probability of interaction with biota is high. The probability of interacting with biota changes in response to many environmental conditions, some of which (e.g., temperature and precipitation) are essentially outside of the scope of management intervention (Silva et al. 2002). But, many of the conditions are susceptible to management once their response is better understood (e.g., carbon availability).

Study Site

This study was conducted in southeastern Oklahoma at the Center for Subsurface and Ecological Assessment Research (CSEAR), operated by U.S. EPA, Robert S. Kerr Environmental Research Center, Ada, Oklahoma, USA. CSEAR is located in an area of interspersed old-field and oak-forest patches characteristic of the Cross Timbers ecotone, historically a mosaic of mixed grasslands and oak-dominated forest between the southern Great Plains and eastern deciduous forests of Texas, Oklahoma, and Kansas. Cultivation at CSEAR was abandoned at ca. 1950 and cattle grazing was halted in 1998.

Within a contiguous old-field, sixteen 40 x 40-m plots were established to investigate ecosystem interactions associated with additions of nitrogen and manipulations of herbivore populations. Plots were separated by a ca. 5-m mowed pathway. N availability and herbivory were manipulated in plots in a factorial experimental design such that 4 plots each received fertilizer only, herbivory manipulation only, a combination of fertilizer and herbivory manipulation, or neither. Plots were fertilized with granular 34% ammonium nitrate at an annual rate of 16.3 kg-N/ha/yr beginning in February 1999 and every 3 months thereafter. Herbivory was manipulated by a □2-m tall chain-link fence of 2.5-cm mesh that effectively excluded intermediate to large sized mammals while supporting greater abundance of small mammals inside fenced plots.

Methods

Two soil samples (separated by □10 m) were collected three times per quarter from each plot.* Samples were processed within 24 Hrs of collection. Samples were hand cleansed of gross contaminants (e.g., plant material, worms, stones) and homogenized. Two subsamples were taken from each sample. Subsamples were then extracted with 2-M KCl using a soil to extract ratio of 2:1 (Silva et al. 2003). Samples were centrifuged at 1800 rpm for 10 minutes at 15°C and filtered using pretreated (with 2-M KCl and deionized water) Whatman 42 filter papers. Extractions were completed within two days of sampling. Filtrates were analyzed for mineral-N using LACHAT QuikChem FIA.

* Note: sampling during spring 1999 was done differently and is not included in these analyses; complete sampling from July-December 2001 was interrupted and are not included in these analyses; however, in both of these cases the data points that exist are consistent with the data and interpretations presented.

Results

In this nitrogen-limited ecosystem, extractable nitrate concentrations never averaged more than 1 mg/l during the growing season and were always less than 2 mg/l during the dormant season on control plots. The introduction of an additional 16.3 kg/ha/yr of N deposition was initially successfully processed.

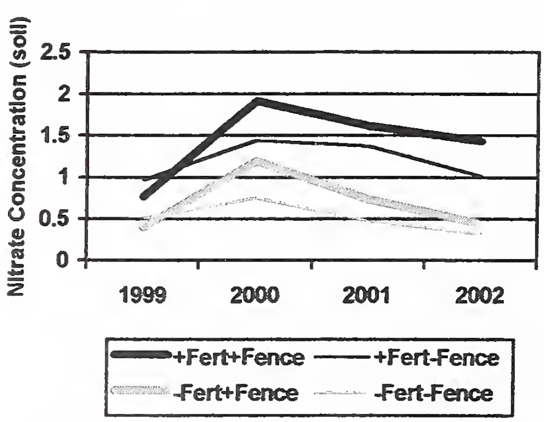

Figure 2. Soil nitrate concentration was consistently greater on fertilized plots. Concentration was also greater on fenced plots, apparently because of highly abundant small mammal populations in fenced plots.

However, this was short-lived and average soil nitrate concentration was 100.5% greater on fertilized plots (Figure 2. $P<0.0001$; $F_{1,360}=37.80$). Further, average nitrate concentration on fenced plots was 24.7% greater (Figure 2. $P=0.0443$; $F_{1,360}=4.07$).
The decline in total soil nitrate measured during 2001 and 2002 (Figure 2) is attributable to lower average concentration measured during winter when concentrations are greatest (Figure 3).

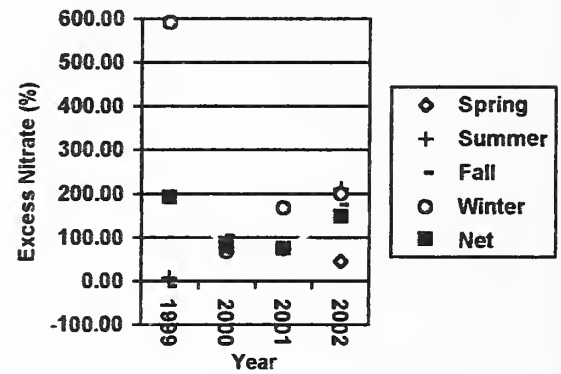

Figure 3. Excess nitrate on fertilized plots has changed dramatically throughout this experiment. This is primarily attributable to changes during winter. Amount of excess nitrate has increased during summer and fall, and stayed essentially the same during spring.

Average excess winter concentration declined from +590.6% during 1999 to +198.7% during 2002 (up from 2000 and 2001). At the same time, average excess concentration during spring remained essentially unchanged, but increased during summer from +6.8% in 1999 to +211.3% in 2002, and during fall from −11.3% in 1999 to +178.3% in 2002 (Figure 3). Mean excess nitrate concentration on fertilized plots was greatest in 1999 (+191.9%) and then declined to approximately +70% for 2000 and 2001, before increasing to +147.3% in 2002.

Excess soil nitrate concentration on fertilized plots ranged between +64.5% during spring to +135.4% during winter (Figure 4). Yearly average mean excess was +105.6%.

Our estimates lead us to expect that essentially all new deposition occurring during the dormant season (winter) is lost to watersheds and we are conducting further research better estimate this loss. However, that this loss is from lands that are considered undeveloped and at low risk is noteworthy.

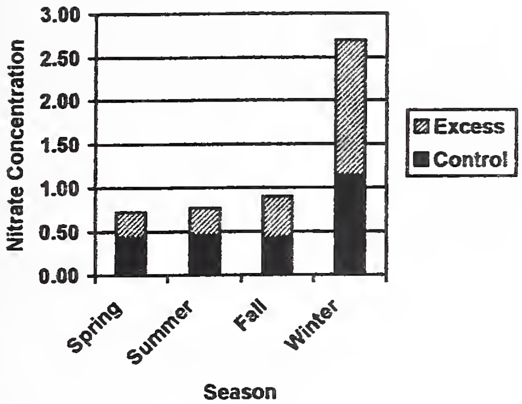

Figure 4. Excess soil nitrate peaks during winter. Averaged over a year, more than twice as much nitrate is available on fertilized plots compared to control plots.

Discussion

Our sites were unable to process the low amounts of nitrogen that we added. Although fertilization began in February 1999, differences among plot response to fertilization were not detectable until December when large differences were observed (Figure 3). Thus, fertilized plots successfully incorporated the initial nitrogen additions until the first dormant season when a substantial failure of the fertilized plots' ability to process nitrogen occurred. It appears to be this initial failure, or initial adaptation to new conditions, leading out of the dormant season that explains the relatively high concentrations measured during 2000 (Figure 2). Further adaptation may be lowering annual average soil nitrate concentration (Figure 2), but excess nitrate remains available during much of the year (Figure 3).

In this experiment, soil nitrogen chemistry is the final product of availability and trophic interactions. Elsewhere, we describe how chronic exposure to low doses of nitrogen on our study plots fundamentally change trophic/ecosystem interactions and lead to reduced nitrogen use efficiency (e.g., Jorgensen et al. 2002, Clark et al. 2003, Mayer et al. 2003, West et al. 2003). Although we expected change to site nitrogen processing to occur, we did not have a clear idea of how long it would take for them to be detectable. In fact, it took only a single growing season for changes to the soil's ability to adapt to nitrogen inputs to be evident and for changes to interactions among biota and ecosystem to be measurable.

This study demonstrates that even regions of the Eastern United States which receive seemingly modest or small amounts of atmospheric nitrate deposition may have undergone a long-term change to their ability to process further nitrogen inputs. Perakis and Hedin (2002) have hypothesized that current ecosystem nitrogen biogeochemistry in much of the Northern Hemisphere may be the product of an historical alteration to biogeochemical cycles that has not yet been identified or understood. We are currently investigating means of detecting such changes through insights from this research.

Finally, our data suggest that the inherent capacity of terrestrial ecosystems to process bioavailable nitrogen has been altered by past exposures to atmospherically deposited nitrogen. Although by themselves these data are insufficient to lead us to the conclusion that existing technologies will be unable to address currently observed environmental problems associated with nitrogen (e.g., Jorgensen 2002), they are consistent with that outcome.

Implications

This study provides evidence for several important phenomena:

1) soil nitrogen chemistry is altered after one growing season of exposure to low-level nitrate deposition.
2) soils are slow to adapt to new nitrogen inputs.
3) the ability of the ecosystem to respond to nitrogen inputs can be compromised by previous exposures.

Ecosystems may be at risk from doses of atmospherically deposited nitrate that are generally considered to be low or modest. As deposition of bioavailable nitrogen from the atmosphere can reasonably be expected to increase, it is prudent to identify and develop management options now to both restore ecosystems that are already compromised and to buffer affects to ecosystems that are at risk from new nitrogen inputs.

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Reaching Communities Across Arizona with Water Education

D. Phillip Guertin, Kim McReynolds, Susan Pater, Jeff Schalau, Kerry Schwartz, Sandra Sutton, Susan Ward, Deborah J. Young

Abstract

The University of Arizona Cooperative Extension, in collaboration with Arizona Department of Environmental Quality and other state and federal agencies, is developing a water education program for the residents of Arizona. The program addresses education needs for K-12 school levels, general adult education, and specific stakeholder groups. The paper will discuss the different facets of the extension program.

Keywords: watershed, water quality, nonpoint source pollution, outreach

Introduction

Arizona's increasing population continues to have an enormous impact on vast tracts of public and private land. Open space is being converted into housing developments, golf courses, and other recreational uses. This has put greater demands on the limited natural resources of these predominantly arid and semi-arid landscapes. In many areas of Arizona, ground water withdrawal is exceeding recharge. Agriculture, suburban development, recreation, grazing, and other activities are all influencing surface and ground water quality. This rapid urbanization permanently alters natural watershed characteristics. The factors cited above often give rise to contentious battles among land managers, ranchers, farmers, miners, developers, environmental activists, and recreational users. Each group has their own, often contradictory, visions of the land's fair, appropriate, and responsible use. Yet, each impacts water quality and quantity in its own way.

Nonpoint source pollution and sustainability of water supplies are the two primary water issues facing Arizonans. Stakeholders need to be involved in decision-making processes that affect their land and water resources. Watershed integrity is inextricably linked to varying land uses and the quality and quantity of water resources. Better land use decisions are the key to protecting the natural resources, community character, and long-term economic health of Arizona's communities.

Because land use is a principal issue, the people making land use decisions are our key target audience. In Arizona, this means bringing together municipal and county governments, public land managers, ranchers, miners, hunters, recreational users, environmental groups and other concerned citizens groups to discuss land use practices and develop watershed-based goals.

The Arizona Cooperative Extension is in the process of developing a water education program that will address the needs of an array of stakeholder groups in the state.
While some components such as Project WET and xeriscape education have been in place for many years two new components have began in the last year. This paper will outline the current status of the Cooperative Extension education program.

Water Education in Cooperative Extension

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The goal of Arizona Cooperative Extension water education programs is to provide knowledge to a target audience so that they can make "better," more informed decisions. It is the assumption that if people are aware of the consequences of their actions, personally and to the wider community, they will change their behavior.

The University of Arizona Cooperative Extension in the College of Agriculture and Life Sciences delivers research-based outreach and education to engaged stakeholders to foster better decisions to promote healthy and sustainable communities. Cooperative Extension faculty are based in all 15 counties and on six Indian reservations; Arizona residents have input on Extension programs that are delivered in their locale. Cooperative Extension uses volunteers to further disseminate their outreach and education.

There is increasing demand by Arizona residents for information and education on water issues and Cooperative Extension has been addressing this need. State-wide, Cooperative Extension faculty made 10,348 face to face contacts in 2002 in educational programs related to water resources. The University of Arizona research scientists are working on these issues as well; a communication network between campus water centers and state and federal agencies is in place. However, in the past the water education effort has been piecemeal, with many small individual efforts. The current goal is to organize the water education efforts to address the needs across the entire state.

The basic audiences to be addressed through education programs are: K-12, general adult education, and specific stakeholders groups. Three extension programs have been created and/or enhanced to address the education needs for the three audiences: Water Education for Teachers (WET) which addresses teachers of K-12 education, Master Watershed Steward Program (MWS) for general adult education, and Nonpoint Education for Municipal Officials (NEMO) which addresses elected and appointed officials, as well as specific stakeholder groups.

Project WET

Project WET is an international, interdisciplinary, water education program for formal and nonformal educators of students ages 5 to 18 administered in Arizona by the Water Resources Research Center and College of Agriculture and Life Sciences. Curricula, teaching aids, and materials are offered to school districts across the state. The curricula cover the properties of water, the water cycle, watersheds, groundwater, water quality, water rights, as well as an understanding of the importance of water to all water users.

Arizona Project WET facilitators conduct workshops where educators learn about Arizona's water resources by participating in fun, interactive, classroom-ready activities. The activities, developed and tested by teachers, are designed to develop critical thinking and build an understanding of concepts by experiential learning. The National Project WET Curriculum and Activity Guide, is a nationally acclaimed teaching resource. Other resources include a Nonpoint Source Curriculum for grades 9-12 and K-6, a Watershed Manager Guide, an Arizona WET Guide, Conserve Water Educators' Guide for grades 6-12, and a brand new Healthy Water Healthy People Guide with water testing kits.

As part of the education program, students learn to use the scientific process in activities like *H2Olympics*. In *Just Passing Through*, and *Incredible Journey* and dozens more activities, students use their own bodies as models for simulating processes. Students learn to organize and present data gathered from their own experiences using mathematical and graphical representations in activities like *Back to the Future* and *Get the Groundwater Picture*. Students are asked to analyze data and scientific reports as well as learning to come to consensus with people that hold different opinions in *A Grave Mistake* and *Water Bill of Rights*. At all levels the students develop critical thinking skills and learn about how water affects their communities and lives.

There were 370 Arizona Project WET educator workshop participants in 2002 affecting a reported 22,445 students per year. Many of the materials used in this program, designed for youth, can be adapted for adult audiences. Educators can arrange to receive Project WET resources by attending a workshop or purchasing materials at the National Project WET website. Teaching tools including groundwater flow models, nonpoint source pollution models, and Liquid Treasure History Trunks are available for check out in five different regions statewide. Project WET staff work with local groups (e.g., school systems, cities, and local resource organizations) to sponsor, plan and conduct water education events.

WET resources are made available at in-service teacher training seminars, through classroom presentations, and at most major environmental education events.

Master Watershed Steward Program

The Master Watershed Steward Program (MWS) began in Oregon and has been found to be a very successful approach to general adult education. MWS aims to increase the capacity of community members to identify and address water resource issues at local levels. The MWS program provides educational sessions and materials to help individuals understand how their watershed works. These volunteers then apply this knowledge of watershed stewardship by serving as community resource persons/educators to disseminate research-based watershed information, coordinate local projects, and assist in data collection in their communities. The program is modeled after the successful Master Gardener Program, an intensive and comprehensive, hands-on, learner-centered program that educates volunteers about horticulture; Master Gardeners then assist in the delivery of Cooperative Extension educational programs. Like the Master Gardener program, Master Watershed Stewards assist in the delivery of Cooperative Extension watershed/water resource education programs.

Both Yavapai and Cochise Counties have delivered pilot MWS courses in the last two years. A course typically consists of 40 hours of course work and two full-day field trips. Topics covered in the course include: hydrology, meteorology, geology, soils, climate, riparian and aquatic ecology, water law, best management practices, group organization, and funding sources. The primary focus is how these various topics affect water quality, water quantity, and watershed processes. Personnel from universities, governmental agencies, tribes, and non-governmental organizations may assist in delivering the lectures, field experiences, and hands-on activities.

Upon successful completion of the course, including a comprehensive final examination, participants are given Associate Master Watershed Steward Status. After contributing 40 to 50 hours of volunteer service to their community, delivering pre-approved, watershed-related education, they become Certified Master Watershed Stewards. MWS volunteer service has included working with local schools, field data collection, assisting a range of agencies in water/watershed projects, and organizing educational conferences.

Over the next three years the Arizona MWS program will be going statewide. A coordinator will be located at the main University of Arizona Campus in Tucson, Arizona. The state will be divided into four regions: Colorado River which includes the lower Gila and Bill Williams watersheds, Southeast which will include the Upper Gila watershed and tributaries such as the Santa Cruz and San Pedro watersheds; Northcentral which includes the Verde, Agua Fria, and Hassayampa watersheds, and Northeast which covers the Little Colorado and Upper Salt watersheds. The main campus coordinator also will provide support in the urban corridor between Tucson and Phoenix.

The statewide MWS program will review and adapt education materials to address Arizona needs and conditions, develop a web site to provide access to educational materials, assist in the MWS courses, and conduct workshops to train local facilitators. Each region will also have a coordinator. The responsibilities of the regional coordinators are to organize the MWS courses within their region and supervise and support the volunteers.

Nonpoint Education for Municipal Officials

The Southwestern United States, including the State of Arizona, is the fastest growing region in the country. Consequently, this once rural area is rapidly developing, and environmental issues that were once considered to be "eastern" problems must now be addressed. This includes water quality and nonpoint source pollution. Problems can no longer be neglected without evoking conflicts. As the result of increased population, there is a need to address health and quality-of-life issues that may result from contamination of water resources from nonpoint sources.

Nationally, the Nonpoint Education for Municipal Officials (NEMO) program has been very successful in helping to mitigate nonpoint source pollution. The goal of NEMO is to educate land use decision-makers to make voluntary actions that will mitigate nonpoint source pollution and protect our natural resources. Arizona Cooperative Extension at the University of Arizona, in cooperation with the Arizona Department of Environmental Quality, has recently initiated a NEMO program in the State of Arizona.

Arizona NEMO will consist of educational projects focusing on both water quality and quantity issues as identified by watershed characterizations. This will include topics such as comprehensive and integrated watershed planning, best management practices, water conservation, and riparian restoration. Part of the effort will include illustrating current and future conditions using geospatial technology (e.g. geographic information systems) to assist in the educational process. All stakeholders in a watershed will be included in the process. Educational projects will be tailored for the specific watershed conditions and water quality problems identified from watershed characterization.

Presently, NEMO programs have been concentrated in the eastern United States. The Arizona NEMO will adapt the NEMO approach to conditions in the semiarid, western United States. There are sharp differences between the eastern and western United States that must be considered for the successful application of the NEMO approach. First, the land ownership patterns are vastly different. The eastern United States is primarily comprised of private land, and the local land use authority is concentrated in municipal (village, town and city) government. In contrast, watersheds in the western United States are often a mosaic of different landowners and land use authorities. This mosaic may include municipal, county, state, and federal land owners, each with its own set of goals and policies. Land use in a watershed can range from intensive irrigated agriculture, open range grazing, residential development, and extensive wilderness areas, all within the same watershed. The western United States with more diversity, will require adapting methodologies to meet the conditions in each watershed that includes all of the local, county, state, tribal, and federal land use and water authorities into the process.

The second major difference between the eastern and western United States is the availability of water. In the eastern United States the supply of water is not the limiting factor and water quality is the chief concern. Watershed management focuses on nonpoint source pollution problems from private land. Municipal government uses regulatory (e.g. zoning) and incentive programs to mitigate the problems. In the semiarid, southwestern United States the water supply is limiting, and many natural resource problems are related to the lack of water, as well as water quality. The successful application of the NEMO approach will require integrating water supply and water quality concerns into the watershed planning process.

Arizona NEMO was initiated in October 2002. During the first year, the project concentrated on conducting watershed characterizations to support the watershed-based planning process in cooperation with Arizona Department of Environmental Quality (ADEQ). Arizona NEMO will work in partnership with ADEQ in developing watershed-based plans and serve as the education component during the implementation phase. In summer of 2003, Arizona NEMO will hire a coordinator who will be located on the main University of Arizona campus. The coordinator will contact stakeholder groups, develop an education needs assessment, and develop a web site to provide access to educational materials. Education material will be developed for the different stakeholder groups and workshops and seminars will be conducted starting in 2004.

In developing the education material, Arizona NEMO will stress the concept of Integrated Watershed Management and Planning (IWMP). The goal is to emphasize the linkages between water supply and quality and adapt concepts from the EPA Smart Growth program and National NEMO.

Summary

The goal of the Arizona Cooperative Extension is to deliver research-based, non-advocacy professional education to engage stakeholders and foster better water and land use decisions to improve water use, water quality and community sustainability. It is hoped that the education programs outlined above will serve as an instrument to guide research being conducted at our state universities to serve our stakeholders in local, state, tribal, and federal agencies and to the people of the State of Arizona. Wise decisions are only made when everyone has good and complete information.

Acknowledgments

Support for Project WET, Master Watershed Steward and Arizona NEMO has been provided by the Arizona Department of Environmental Quality. Without the support and cooperation of ADEQ many of these programs would not exist. The authors

appreciate the reviews of Cheryl Craddock and C.P. Patrick Reid.

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http://cals.arizona.edu/extension/

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EPA Smart Growth Program
http://www.epa.gov/livability

National Project WET (Water Education for Teachers)
http://www.projectwet.org

Nonpoint Education for Municipal Officials (NEMO) http://nemo.uconn.edu/national/

Oregon Master Watershed Steward Program
http://seagrant.oregonstate.edu/wsep/

First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Water Quality and Quantity III (Fire)

Impacts of Fire on Hydrology and Erosion in Steep Mountain Big Sagebrush Communities

Frederick B. Pierson, Peter R. Robichaud, Kenneth E. Spaeth, Corey A. Moffet

Abstract

Wildfire is an important ecological process and management issue on western rangelands. Major unknowns associated with wildfire are its affects on vegetation and soil conditions that influence hydrologic processes including infiltration, surface runoff, erosion, sediment transport, and flooding. Post wildfire hydrologic response was studied in big sagebrush plant communities on steep slopes with coarse-textured soils. Significant rill erosion was observed following both thunderstorm and rapid snowmelt events. Rainfall simulation and controlled overland flow techniques were used to study post-fire effects on infiltration, and interrill and rill erosion processes on burned and adjacent unburned areas. Results indicate that burn severity and the development of water repellent soil conditions play significant roles in determining infiltration and interrill erosion rates, particularly on shrub coppice dunes characterized by high surface litter accumulations. The most dramatic and long-lasting affect of fire was on rill erosion processes by reducing ground cover needed to slow and spread water as it moves across the soil surface. Ongoing research efforts are aimed at characterizing the hydrologic impacts of prescribed fire used as a tool to manage vegetation and mitigate the impacts of catastrophic wildfire events.

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Keywords: rangeland, fire, hydrology, erosion, infiltration, water repellent soils

Introduction

Fire is a natural component of the Intermountain sagebrush-steppe ecosystem (Wright and Bailey 1982) with return periods of 25 to 100 years, depending on community type and natural fuel load and distribution. Fuel and land management activities in the past century have disturbed fire frequencies putting wildland values such as soil and water quality at greater risk from devastating wildfires. Higher runoff rates from severely burned landscapes can lead to flooding and increased risk to human life and property. Increased soil erosion over natural levels following wildfire can lead to loss of soil productivity and a decline in rangeland health.

Hydrologic consequences of fire have been closely examined in forested ecosystems, but few studies have examined impacts of wildfire on rangeland hydrology. Many studies have shown an increase in runoff and erosion rates the first year following fire, with recovery to pre-fire rates generally within five years (Wright and Bailey 1982). Timing and extent of recovery is highly dependent on precipitation, slope and vegetation type (Branson et al. 1981, Wright et al. 1982, Knight et al. 1983, Wilcox et al. 1988).

The USDA-ARS Northwest Watershed Research Center (NWRC) has been investigating the interaction of wildfire and rangeland hydrologic processes on steep, coarse textured, sagebrush dominated landscapes in Idaho and Nevada. Research objectives are aimed at quantifying impacts of wildfire on infiltration capacity, runoff, and erosion, gaining insight into the processes involved and determining how long fire effects persist. The NWRC is also implementing a prescribed fire research program on Reynolds Creek Experimental Watershed (RCEW) to examine the hydrologic impacts of using prescribed fire as a vegetation and fuels management tool.

Study Areas

Study areas are located on the Boise Front immediately above the city of Boise, Idaho (Eighth Street Fire), and in the Pine Forest Range, 50 km south of Denio, Nevada (Denio Fire). Both sites have vegetation consisting of big sagebrush (*Artemisia tridentata*)/ Idaho fescue (*Festuca idahoensis*) communities. Some areas are characterized by increases in three-awn (*Aristida* spp), Sandberg's bluegrass (*Poa sandbergii*), and rabbitbrush (*Chrysothamnus* spp). Soils on both sites were derived from granite and consist of well drained, fine gravelly coarse sandy loams, on slopes of 35 to 45%. Both sites were severely burned by wildfire. More complete descriptions of study sites are presented by Pierson et al. (2001, 2002).

Methods

At both locations burned sites were compared to unburned sites with the same soil type, slope and vegetation as that found before the fire. Sampling on both burned and unburned sites was stratified based on coppice (areas strongly influenced by the existence of a shrub) and interspace areas (areas between shrubs primarily dominated by grasses and forbs) (Pierson et al. 1994). A portable oscillating-arm rainfall simulator with specifications as described by Meyer and Harmon (1979) was used to achieve intermittent rainfall similar to naturally occurring rainfall. Simulations were run on natural plots, 0.5 m^2 in size, without pre-wetting. Sampling was conducted in late summer when soil moisture for all plots was extremely low (<10% generally). The Boise Front sites were sampled one year following the fire and the Denio sites were sampled immediately following the fire. Rainfall was applied at a rate 67 mm hr^{-1} on the Boise Front and 85 mm hr^{-1} on the Denio sites. Runoff samples were collected at one- or two-minute time intervals throughout the 60-minute simulation and analyzed for runoff volume and sediment concentration. Infiltration capacity for each time interval was calculated as the difference between measured rainfall and measured runoff.

A flow regulator was used to apply rill flow rates of 7, 12 and 15 l/min to six randomly selected points at each site to simulate the rill development process. Flow samples were collected 4-m down slope of the release point. Suspended sediment samples for both rainfall and rill procedures were weighed, dried at 105°C then re-weighed to determine runoff rate and soil loss. Flow velocity in each rill was measured by releasing a concentrated salt solution into the rill and using electrical conductivity probes to estimate the mean travel time of the salt over a known rill length. Rill erosion sampling was done in early spring while soils were still moist. Vegetative cover and ground cover were also estimated for each rainfall simulation plot and rill using a combination of pin-point and ocular estimation procedures. Further description of the methodology can be found in Pierson et al. (2001, 2002).

Results and Discussion

Both the Boise Front and Denio study areas were severely burned with virtually all vegetation and litter being consumed during the fire. Bare ground for all burned sites was greater than 95% resulting in increased soil exposure to the erosive forces of raindrop impact and overland flow. Loss of surface litter, vegetative basal cover and associated microtopographic relief also reduced surface storage of water crucial for reducing runoff and increasing infiltration. Past studies have shown that fire can vaporize organic compounds on the soil surface and distill them downwards, creating concentrated water repellent layers within the upper the soil (DeBano 1981, Wright and Bailey 1982). Such water repellent soil conditions are generally temporary (DeBano et al. 1967, Robichaud 2000b), so that during the first minutes (or longer) of rainfall, water beads on or near the soil surface and quickly runs off the plot. Water repellency then deteriorates as rainfall continues, resulting in a gradual recovery in the infiltration capacity of the soil. The following results demonstrate the effects of these fire impacts on critical hydrologic and erosion processes.

Infiltration

The reduction in infiltration capacity due to the fire was concentrated on the coppice microsites for both study sites. Infiltration rates for coppices were reduced by over 30% in the initial stages of runoff (Figures 1 and 2). After one hour of rainfall, the initial fire-induced reductions in infiltration rates had recovered to a point where no significant differences in final infiltration rates could be detected (Table 1). Coppice microsites are generally dominated by heavy litter accumulations providing ample organic material for the creation of hydrophobic compounds that in turn create water repellent soils. Pierson et al. (2001) defined a soil water repellency index (WRI) to quantify the percent reduction in infiltration capacity during the initial stages of the infiltration process. Burned coppices exhibited an average of 16 and 32% reduction in infiltration due to water repellency on the Boise Front and in Denio, respectively (Table 1).

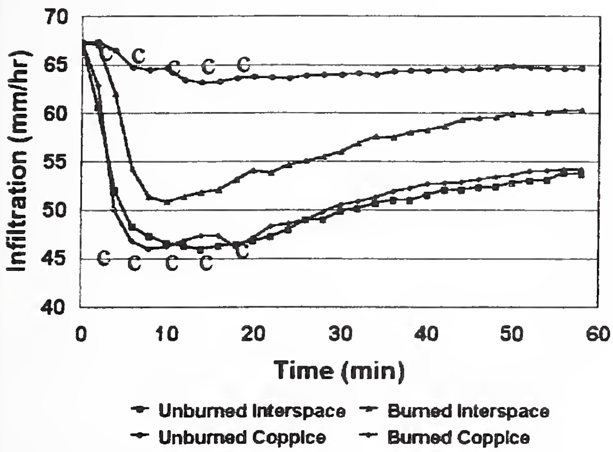

Figure 1. Infiltration rate for burned and unburned microsites, Boise Front, Idaho. Coppice rates within a level of time labeled with the same letter (C) are significantly different (P≤0.05).

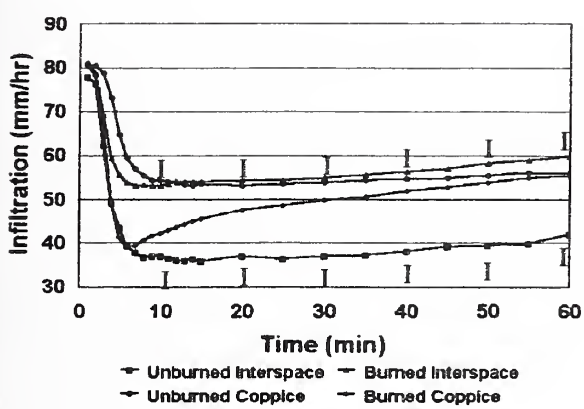

Figure 2. Infiltration rate for burned and unburned microsites, Denio Fire, Nevada. Coppice rates within a level of time labeled with the same letter (I) are significantly different (P≤0.05).

Under very dry conditions, unburned soils with sufficient waxy substances on the soil surface can also create water repellent soil conditions (DeBano et al. 1967, DeBano and Rice 1973). This phenomenon was observed for unburned interspace microsites at both locations. Unburned interspaces had soil water repellency values of 17% on the Boise Front and 9% in Denio (Table 1). Final infiltration rates for unburned interspaces were lower than all other burned or unburned microsites (Table 1).

Runoff

Fire had a significant, but small affect on the initiation of overland flow on all burned sites. Both sites showed a consistent and rapid runoff response to intense rainfall for all burned and unburned microsites. Runoff on burned coppices consistently began approximately 2 min after the start of rainfall. Runoff began within 5 min for all burned and unburned microsites (Table 1). Also, all microsites reached their peak runoff rates from 5 to 14 min following the start of rainfall (Table 1).

The greatest impact of fire in these steep coarse-textured sagebrush systems is not on generating runoff, but its effect on the dynamics of runoff as it begins to move down slope over the soil surface. A year after the fire on the Boise Front, significant rilling was observed following a short intense thunderstorm that flooded the town of Boise, Idaho. In Denio, significant rilling was observed on the burn following numerous rapid snow melt events during the winter following the fire. As a relative measure of how a burned hillslope responds to concentrated overland flow, varying rates of water were released at points on the landscape and the rate of discharge (l/min) was measured at 4 m down slope (Figure 3). At a release rate of 7 l/min, no water reached 4 m on the unburned sites. An average of 2 l/min reached 4 m on the burned sites after the fire and was nearly zero after one growing season. At a flow rate of 12 l/min, water was just beginning to reach 4 m on the unburned sites, while the burned sites flowed at over 4.5 l/min after the fire. Flow declined to less than 1 l/min on the burn after two growing seasons. At a release rate of 15 l/min, the unburned sites flowed at less than 0.3 l/min on the unburned sites. Flow on the burned sites dramatically increased to over 8 l/min after the burn and still flowed at 2 l/min after two growing seasons of vegetation recovery.

Table 1. Hydrologic and erosion variables for the Boise Front and Denio study sites. Values across a row followed by different letters are significantly different ($p<0.05$).

	Burned Coppice	Burned Interspace	Unburned Coppice	Unburned Interspace
Final Infiltration Rate (mm hr^{-1})				
Boise Front, ID	54.2a	60.4a	64.7a	53.7a
Denio, NV	55.5ab	60.0a	55.9ab	40.4b
Water Repellency Index (%)				
Boise Front, ID	16.3a	16.1a	3.3b	17.3a
Denio, NV	32.2a	11.6b	5.3b	9.4b
Time to Runoff (min)				
Boise Front, ID	2.5b	4.3a	5.4a	2.1b
Denio, NV	1.8b	2.0ab	3.0a	2.0ab
Time to Peak Runoff (min)				
Boise Front, ID	12.4a	12.8a	14.0a	13.6a
Denio, NV	5.6a	5.5a	7.8a	6.3a
Sediment/ Runoff Ratio (kg ha^{-1} mm^{-1})				
Boise Front, ID	3.5a	3.8a	1.2a	0.9a
Denio, NV	13.6a	8.7ab	5.1b	6.4b

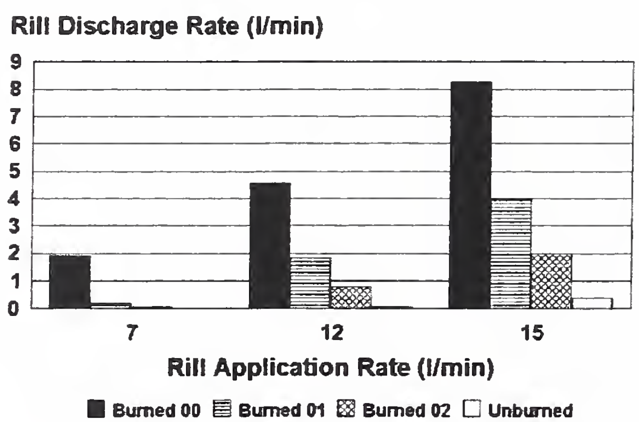

Figure 3. Rill discharge rate for burned and unburned sites on the Denio Fire, Nevada.

Vegetation, litter, rocks and other forms of ground cover create barriers that slow and spread water movement across the soil surface allowing more time for water to infiltrate over a larger surface area. Fire removes most of these barriers and allows the water to concentrate into rills where flow depth and velocity increase. This significantly decreases runoff response time and increases the runoff volume delivered to the bottom of a hillslope and into a stream channel. After the Denio fire, nearly all the soil ground cover was removed exposing over 95% bare ground compared to less than 5% bare ground on the unburned sites (Figure 4). It took two growing seasons and three winters for litter accumulation to reduce the amount of bare ground on the burned sites to near 50% (Figure 4). A ground cover of 60% is often used to gauge adequate hydrologic protection of a site (Gifford 1985, Robichaud 2000a). These results indicate that while 50% vegetation and litter cover does induce a significant reduction in overland flow, the site is not yet fully protected compared to unburned conditions.

Erosion

Fire had far greater impact on rill erosion compared to interrill erosion. Interrill erosion was relatively low for all burned and unburned microsites after both fires. The only significant fire effect observed was for burned coppice microsites on the Denio fire (Table 1). To examine the impact of the fire on rill erosion, the sediment concentration of the rill discharge collected 4 m down slope was used as a relative index to compare burned and unburned sites (Figure 5). No measurable sediment was collected on

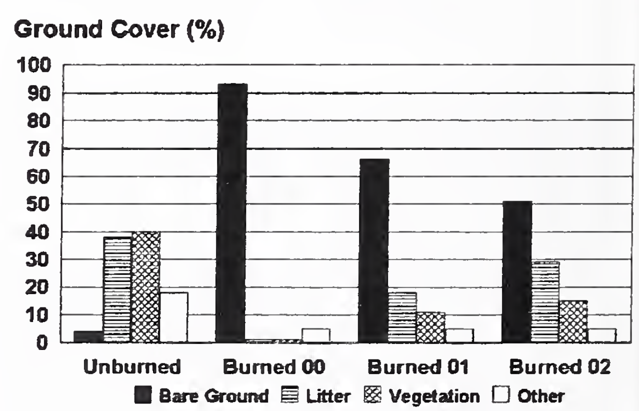

Figure 4. Ground cover for burned and unburned sites on the Denio Fire, Nevada.

the unburned sites at any of the flow rates applied. At a flow rate of 7 l/min, over 175 g/l of sediment were measured after the Denio fire and dropped to zero after two vegetation growing seasons. At a flow rate of 12 l/min, over 110 g/l of sediment were measured for the burned sites after the fire and near 10 g/l of sediment were measured after two growing seasons. At a flow rate of 15 l/min, burned sites exhibited sediment concentrations of over 75 g/l after the burn and less than 10 g/l after two growing seasons. The observed decline in post-fire sediment concentrations as the rill flow rates increased was due to the rills becoming "armored". Rills became sediment limited as the detachable sediment was removed by lower flow rates.

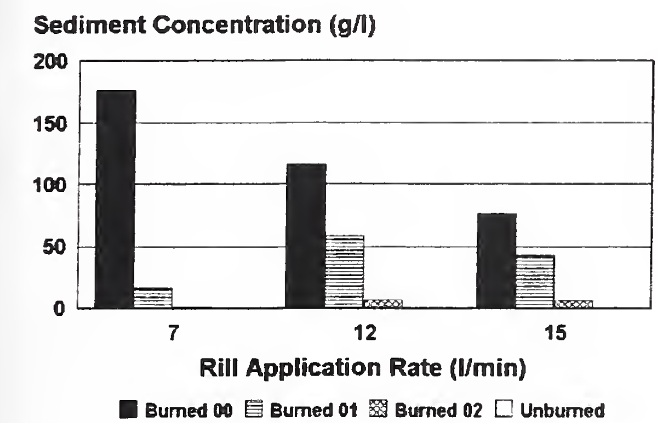

Figure 5. Rill sediment concentration for burned and unburned sites on the Denio Fire, Nevada.

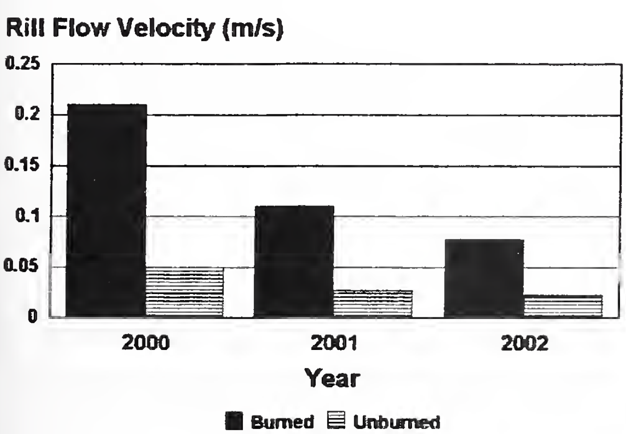

Figure 6. Rill flow velocity for a rill flow rate of 15 l/min for burned and unburned sites on the Denio Fire, Nevada.

The velocity of the water flowing in a rill is directly related to its ability to detach and carry sediment. As an example, the rill velocity at the 15 l/min release rate for the unburned sites was consistently under 0.05 m/s throughout the study (Figure 6). Such a velocity is apparently insufficient to detach and carry sediment down slope in a fully protected system with high ground cover of vegetation and litter. The velocity for burned sites was near 0.21 m/s after the burn and declined to 0.075 m/s after two growing seasons. This indicates that a velocity of 0.075 m/s can still cause rill erosion in a system protected by only 50% ground cover.

Implications

The results of these studies provide a relative index of the increased risk of runoff and erosion and the rate of hydrologic recovery following fire in coarse-textured sagebrush dominated systems. Fire in these systems can induce significant water repellency, particularly in areas dominated by shrubs with large accumulations of litter to fuel the intensity of the fire and provide the hydrophobic substances necessary to create water repellent soils. However, it is important to keep in mind that these systems have a certain degree of natural water repellency when they are extremely dry. This can confuse and confound the assessment of fire-induced water repellency often inventoried and used after wildfires to determine the need for post-fire mitigation. In general, the greatest impact of the fire is not on infiltration, but on overland flow dynamics and rill erosion. Whether burned or unburned, these types of sites are going to rapidly generate runoff under intense rainfall. The impact of the fire is that it leaves sites devoid of barriers to slow and store runoff once it begins to move. The water then concentrates and increases in velocity resulting in greater erosive energy. Water moves more rapidly down slope and ultimately into stream channels potentially causing downstream flood damage.

Future Research

The natural fire cycle of many rangeland ecosystems has been disrupted by past management and fire suppression activities. The result has been a dramatic increase in catastrophic wildfires across the western United States over the past decade. To reduce the threat of wildfire and return these ecosystems to a healthy state means to return them to a natural fire cycle. The NWRC in addition to the work on the hydrologic impacts of wildfire has established a long-term prescribed fire research program on RCEW in southwest Idaho. The goals of the research program are to: determine the short- and long-term effects of prescribed fire on runoff, erosion, water quality and vegetation and forage in rangeland ecosystems; develop pre-fire assessments for predicting post-fire soil, vegetation and hydrologic responses; and investigate interactions of prescribe fire and grazing for reducing the threat of wildfire. The NWRC intends to transfer research findings through decision support tools to help managers determine the risk of wildfire, to explore options for reducing such risk through the use of prescribed fire, and to predict post-fire impacts and optimize rehabilitation strategies.

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Potential Hydrologic Response to a Prescribed Fire on a Small Mountainous Watershed

Gerald N. Flerchinger, Patrick E. Clark

Abstract

Prescribed fire is often used to control invasive weeds, improve habitat, and deter wildfire. The Northwest Watershed Research Center plans to burn a heavily studied 26-ha watershed. This paper investigates the potential hydrological response to that prescribed fire. Changes in water repellency and infiltration capacity were assumed not to limit the low intensity snowmelt input to the basin. Percolation, subsurface flow and runoff during the first runoff season are influenced by the soil moisture deficit created by pre-burn vegetation conditions and will likely not be influenced greatly by the fire. A year of reduced evapotranspiration following the fire is necessary to reduce the soil moisture deficit and increase percolation beyond the root zone and subsurface flow to the stream. Results indicate significant changes in streamflow in this subsurface-flow-dominated watershed may not be observed until the second snowmelt season following the fire and could increase by 25%. These results are unlike watersheds dominated by overland flow and surface runoff where increased flows are more likely to occur during the first year following a fire.

Keywords: evapotranspiration, percolation, prescribed burn, runoff, rangeland

Introduction

Reintroducing fire as part of the natural cycle to control invasive weeds and improve habitat on rangeland is becoming an accepted practice where appropriate. Post-fire vegetation, water balance and streamflow responses, however, are not well understood. Many mid- to high-elevation watersheds of the semi-arid Intermountain West are ephemeral, being dominated by snowmelt, evapotranspiration and subsurface water flow. The Northwest Watershed Research Center (NWRC) conducted a series of studies on one such watershed, Upper Sheep Creek, which culminated in a ten-year water balance for the watershed. The NWRC in cooperation with the Bureau of Land Management currently has plans to burn the watershed in the fall of 2005. Assuming a likely burn scenario and probable vegetation recovery, this study examines the potential changes in evapotranspiration, percolation and streamflow within the watershed in response to vegetation changes following a prescribed fire using the SHAW model.

Field Setting

The Upper Sheep Creek Watershed is a 26-ha snowfed rangeland watershed located within the Reynolds Creek Experimental Watershed in the Owyhee Mountains of southwest Idaho, U.S.A (43°N, 116°W). Annual precipitation is approximately 508 mm, approximately 60% of which falls as snow. Spring snowmelt is the primary source of runoff from the basin. Streamflow typically begins in March/April, peaks in May and ceases in July. Nearly all water reaching the stream is subsurface flow; overland flow is seldom observed in the basin.

The site has considerable spatial variability in soils, vegetation and snow cover. Low sagebrush areas are located predominantly on the windswept southwest-facing slopes and are bare of snow for much of the winter. Lower portions of northeast-facing slopes are dominated by mountain big sagebrush and typically accumulate about a meter of snow during the winter. Aspen thickets are established on the upper portions of the north-facing slopes where large snow drifts form annually. Soils vary from shallow (30 cm) and rocky under low sagebrush to deep (>2 m) silt loam under the

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aspen. The geology of Upper Sheep Creek consists of variably fractured and altered basalt underlain by dense basalt at a depth of 20 to 30 m (Winkelmaier 1987, Mock 1988, and Stevens 1991).

Previous Studies

A detailed study of the Upper Sheep Creek Watershed was conducted by the USDA-ARS Northwest Watershed Research Center from 1984 through 1994. Numerous investigations have been conducted to define the geology of the watershed (Winkelmaier 1987, Mock 1988, and Stevens 1991) and to better understand the processes controlling the hydrologic response of this mountainous watershed (Cooley 1988, Flerchinger et al. 1992, Flerchinger et al. 1993, Deng et al. 1994, Flerchinger et al. 1994, Neale et al. 1995, Tarboton et al. 1995, Unnikrishna et al. 1995, Flerchinger et al. 1996, Luce et al. 1998).

Flerchinger et al. (2000) computed a ten-year water balance of the Upper Sheep Creek Watershed. Because of its spatial heterogeneity, the watershed was broken into three zones based on similarity in soils, vegetation, and snow accumulation. The three zones are referred to as low sagebrush, mountain big sagebrush, and aspen, which comprise 58.9, 26.6 and 14.5% of the watershed, respectively. A partial water budget was computed for each of the three landscape zones. Average annual effective precipitation for the watershed was 471 mm over the ten-year period. Runoff from the watershed averaged 30 mm and was linearly correlated ($r^2 = 0.52$) to effective precipitation above a critical threshold of approximately 450 mm. Simulated percolation of the water beyond the root zone using the Simultaneous Heat and Water (SHAW) model correlated extremely well with measured runoff. A regression equation between simulated percolation and measured runoff can be written as ($r^2 = 0.94$; root mean square error = 7.1 mm):

$$R = -12.3 \text{ mm} + 0.565\, PERC \qquad (1)$$

where R and $PERC$ are runoff and simulated percolation beyond the root zone, both in mm.

Post-Fire Regeneration

Vegetation studies during the ten-year study at Upper Sheep Creek determined that leaf area index (LAI) of the low sagebrush, mountain big sagebrush, and understory of the aspen site at the peak of the growing season is approximately 0.4, 1.2 and 1.0, respectively, based on point frame measurements. Leaf area index of the aspen canopy was 2.0. Vegetation characteristics used in the ten-year water balance simulation and assumed vegetation regeneration for a post-burn scenario are summarized in Table 1.

Table 1. Vegetation characteristics for unburned ten-year water balance and assumed vegetation regeneration for the first three years following a prescribed fire.

Landscape Area	Vegetation	Unburned		1st year		2nd Year		3rd year	
		LAI	Root depth (m)	LAI	Root depth (m)	LAI	Root depth (m)	LAI	Root depth (m)
Low Sagebrush	shrubs	0.4	1.0	0.4	1.0	0.4	1.0	0.4	1.0
Mountain Big Sagebrush	shrubs	0.72†	2.0	0.0	0.0	0.0	0.0	0.0	0.0
	grasses/forbs	0.48†	n/a	0.75	0.5	1.0	0.75	1.2	1.5
Aspen	aspen	2.0	2.0	0.2	2.0	0.4	2.0	0.6	2.0
	grasses	1.0	1.0	1.0	0.5	1.0	0.75	1.25	1.0

† Unburned analyses did not distinguished between LAI or rooting depth of shrubs versus grasses/forbs.

The low sagebrush zone (*Artemisia arbuscula*) has sparse vegetation with some grasses (*Poa secunda*) and considerable bare ground. Due to the lack of vegetation on this site, it is doubtful that it will carry a fire as is very typical of this ecosystem (USDA Forest Service 2003). Thus, post-fire conditions of the low sagebrush zone were assumed unchanged.

The mountain big sagebrush zone (*Artemisia tridentata vaseyana*) supports near complete cover of sagebrush, snowberry (*Symphoricarpos spp.*) and grasses. Pierson et al. (2001) compared burned and unburned areas at a site similar to Upper Sheep in northwestern Nevada. One year after the fire, grass and forbs replaced 35% of the total (shrub, grass and forbs) vegetation cover of burned coppice areas; percent cover of grasses and forbs within interspace areas recovered almost completely. This suggests a first-year post-burn LAI for Upper Sheep of approximately 0.75 consisting primarily of grasses and forbs. A first-year rooting depth of 0.5 m was assumed. Total LAI for the site was assumed to regenerate to pre-burn conditions within three years, consisting primarily of grasses with root depth extending down to a maximum depth of 1.5 m as suggested by Canadell et al. (1996) for bluebunch wheatgrass (*Agropyrun spicatum*). Mountain big sagebrush typically requires 15 years to recover (USDA Forest Service 2003), which is beyond the scope of this study and not considered a factor in short-term regeneration of the site.

The aspen zone consists of a thick stand of aspen (*Populus tremuloides*) and willow (*Salix spp.*) with an understory of mixed grasses. Although it can be difficult to initiate a crown fire in aspen, stands with abundant understory fuels can carry ground fires well under hot, dry fall conditions. Aspen have thin bark with little heat resistance and are easily top-killed by fire (USDA Forest Service 2003). Root systems of top-killed aspen send out suckers for several years after fire. Based on a typical ten-year recovery for aspen stands (USDA Forest Service 2003), the aspen was assumed to gain 0.2 LAI per year, with an unchanged pre-burn maximum root depth of 2 m. Grasses and forbs under the aspen were assumed to regenerate to their pre-burn levels within the first year.

Plant litter on the surface was assumed to be zero the first year following the study and gradually build to 2000 kg/ha of litter material with 40% ground cover by the third year for both the mountain big sagebrush and aspen zones. Pre-burn conditions are approximately 9000 kg/ha of litter material with about 90% ground cover resulting from ten years of exclusion from grazing.

Model Application

The original SHAW simulation for the ten-year water balance simulated 11 years (Sept 1983 through Sept 1994) to allow one year for the assumed initial conditions to equilibrate with climatic conditions. For this study, 11 separate simulations were conducted using the SHAW model, assuming a fire at the end of September for each respective year of the 11-year simulation. Vegetation regeneration scenarios presented in Table 1 were used in the model. Initiation of the growing season was adjusted each year based on snowcover depletion, but vegetation growth and regeneration were not adjusted for yearly weather variations. Changes in soil hydrophobicity and infiltration capacity were assumed not to limit infiltration of the low intensity snowmelt. Vegetation is secondary to the topographic influence on snow drifting in this watershed; given the unknown and complex effects of vegetation removal on drifting, the drift factors and effective precipitation determined by Flerchinger et al. (2000) were left unchanged for this study.

Results

Simulated annual evapotranspiration (ET) for the aspen and mountain big sagebrush zones are presented in Table 2 for the first three years following a fire. ET drops substantially the first year following the fire in response to vegetation removal. ET then increases each year, gradually returning to pre-burn conditions (within 2 mm) by the third year following a fire. Some third year ET estimates for the mountain big sagebrush are higher than pre-burn conditions due to higher soil moisture levels after two years of reduced ET (Table 2). This was not the case for the aspen because soil moisture deficits within the aspen zone are typically replenished with the spring melt season.

Simulated percolation for the mountain big sagebrush and aspen zones indicated relatively little response in percolation beyond the root zone for the first year following a fire, and percolation does not peak until the second year (Table 3). With a burn in the fall, spring runoff of the first post-fire year is influenced strongly by the moisture deficit created by pre-burn ET conditions. Thus, a response in percolation does not occur until the second post-fire snowmelt season. By then a reduced ET

Table 2. Comparison of annual evapotranspiration (mm) between unburned and 1st, 2nd, and 3rd year following a fire.

Year	Mountain big sagebrush zone				Aspen zone			
	No burn	1st Year	2nd Year	3rd Year	No burn	1st Year	2nd Year	3rd Year
1985	572	407	457	n/a	446	382	392	n/a
1986	582	377	425	529	491	391	436	469
1987	609	411	498	583	567	497	550	569
1988	409	312	364	471	516	435	496	510
1989	531	357	410	542	522	436	482	513
1990	491	381	435	529	526	491	500	523
1991	418	342	368	462	523	442	470	494
1992	307	267	292	349	511	433	496	509
1993	510	399	431	470	437	407	409	434
1994	491	303	361	449	500	430	475	494
Average	492	356	404	487	504	434	471	502
Percent †	100	72	82	99	100	86	93	100
p-value ‡	n/a	<0.001	<0.002	0.799	n/a	<0.001	0.001	0.035

† Percentage of annual ET compared to no burn
‡ Results of paired t-test comparing ET for respective post-burn years to ET with no burn

Table 3. Comparison of annual percolation beyond the root zone (mm) between unburned and 1st, 2nd, and 3rd year following a burn.

Year	Mountain big sagebrush zone				Aspen zone			
	No burn	1st Year	2nd Year	3rd Year	No burn	1st Year	2nd Year	3rd Year
1985	22	57	114	n/a	623	626	694	n/a
1986	32	84	191	156	815	833	894	864
1987	-30	-19	23	2	11	75	147	102
1988	-15	-15	1	-12	29	43	102	67
1989	-7	-2	28	61	612	616	688	642
1990	-6	-6	14	3	356	320	430	379
1991	-5	-5	-4	8	77	83	164	126
1992	-3	-3	-4	2	0	15	60	28
1993	52	65	89	110	754	725	808	769
1994	-12	-5	14	-6	143	137	202	180
Average	3	15	47	36	342	347	419	351
p-value †	n/a	0.052	0.019	0.032	n/a	0.558	<0.001	<0.001

† Results of paired t-test comparing percolation for respective post-burn years to percolation with no burn.

Table 4. Comparison of runoff (mm) between unburned and 1st, 2nd, and 3rd year following a burn.

Year	No burn	1st Year	2nd Year	3rd Year	Unburned Upper Bound 90% C.I.
1985	85	91	98	n/a	99
1986	93	102	123	111	108
1987	0	0	2	0	3
1988	0	0	0	0	7
1989	37	38	48	49	47
1990	16	13	25	19	29
1991	0	0	0	0	12
1992	0	0	0	0	7
1993	67	67	77	77	78
1994	0	0	7	2	15
Average	30	31	38	34 ‡	
p-value †		0.314	0.017	0.039	

† Results of paired t-test comparing runoff for respective post-burn years to percolation with no burn; t-test considered only those years where runoff occurred.
‡ 3rd year average computed using 85 mm of runoff for 1985.

season results in lower soil moisture deficits and increased subsurface flow.

Areal percolation (Table 3) translates directly to estimated runoff (Table 4) using the relationship established in Equation (1). Interestingly, a pairwise t-test comparing the first year runoff with unburned runoff shows no significant difference. Runoff is greatest the second year following the fire and somewhat less the 3rd year in this basin dominated by subsurface flow. A year of reduced ET is necessary following the fire to decrease the soil moisture deficit and significantly influence percolation through the root zone, subsurface flow and, ultimately, runoff from the basin.

Comparison of the upper bound of the 90% confidence interval in runoff for no burning in Table 3 with estimated runoff following a fire indicates that for years with measured runoff, the second year is generally near the upper bound of the 90% confidence interval. Therefore, a year with adequate runoff will be necessary to detect a response in runoff after a fire. The ten-year water balance study occurred during a period of below normal precipitation; long-term average precipitation for the watershed is around 508 mm compared to 471 mm for the ten-year study. Of the 18 years of runoff records from Upper Sheep, five of the six years with no runoff were within this ten-year period. Consequently, there is a greater likelihood of detecting an increase in runoff in response to a fire than the ten-year study might indicate.

Summary and Conclusions

Simulated ET was significantly less the first year following a fire compared to that for no burning as a result of vegetation removal, but gradually returned to pre-burn conditions by the third year following a fire. After a year of significantly less ET, percolation was significantly greater during the second post-fire year (Table 3). Increases in percolation during the first year following the fire were marginally significant compared to pre-burn conditions for the mountain big sagebrush zone but were not significantly different in the aspen zone. Percolation during the first spring snowmelt season was influenced by soil moisture deficits created by pre-burn vegetation conditions. A significant response in percolation to a fire was not observed until after a year of reduced ET and soil moisture deficit.

The delayed response in percolation translated directly into a delayed runoff response in this system dominated by subsurface flow. Runoff was not significantly changed during the first year following a fire. This observation may be specific to this and similar watersheds dominated by subsurface flow and is unlike watersheds dominated by overland flow and surface runoff where flows are more likely to increase during the first year following a fire.

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Fire Disturbance and Nitrogen Deposition Impacts at the Watershed Scale in Southern California

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Abstract

The mountains of southern California have some of the highest rates of atmospheric deposition in the world. Over the last two decades a consistent picture of unusually high levels of nitrate in streams has been painted using field sites at the San Dimas Experimental Forest (SDEF) and across southern California. Within the context of southern California, three major processes appear to drive the observed variability in nitrate concentrations in streams. First, chronic atmospheric nitrogen deposition leads to high nitrate concentrations. Second, the profound inter-annual variability in rainfall drives nitrate concentrations in streams. Wet years produce a more thorough flushing of nitrate from the watershed resulting in higher nitrate concentrations compared to dry years. Third, fire is a critical process for most ecosystems in southern California; its impacts vary from vegetation and hydrologic consequences to more subtle impacts on biogeochemical fluxes. The results of a prescribed fire experiment in 1984 at the San Dimas Experimental Forest indicate that fire initially causes elevated stream nitrate concentrations but that over the longer term watersheds with more frequent fire have lower nitrate export. This longer term decrease after fire is likely due to losses of nitrogen to the atmosphere during burning and to surface water export in sediment and dissolved load in the immediate post-fire environment. These losses are then followed by rapidly aggrading vegetation during the ensuing years of post-fire recovery that provides a sink for nitrate originating from either atmospheric deposition or N mineralization/nitrification.

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Keywords: nitrogen deposition, nitrogen saturation, chaparral, Mediterranean climate, fire response and recovery

Introduction

Atmospheric nitrogen deposition can adversely impact terrestrial and aquatic ecosystems through fertilization and acidification of ecosystems (Stoddard 1994, Aber et al. 1998, Fenn et al. 1998). The mountains of southern California receive some of the highest rates of atmospheric deposition in the world (~35 kg ha^{-1} $year^{-1}$) (Bytnerowicz et al. 1987). While the ecosystems downwind of the southern California metropolis are impacted by these high rates of deposition, the consequence of soil and aquatic ecosystem acidification is not generally one of the most critical problems (Fenn et al. 2003). Instead terrestrial and aquatic ecosystems in southern California are most affected by the fertilization effects of atmospheric nitrogen deposition and by the concurrent increase in stream NO_3^- concentrations (Meixner and Fenn 2003).

The response of ecosystems to increased atmospheric nitrogen deposition is not linear and can be affected by any number of factors including: history of disturbance, rate of atmospheric nitrogen deposition, climate, and ecosystem structure (Fenn et al. 1998). Among the types of disturbance that can effect long term ecosystem response to atmospheric nitrogen deposition are logging activities, wind blowdowns and fire (Vitousek et al. 1979, Lovett et al. 2000). Of these disturbances fire in particular is not well studied especially in terms of its long-term impact on ecosystem dynamics and biogeochemical fluxes (Wan et al. 2001). Given the current interest in re-introducing fire on the landscape as a land management practice, it will be important to understand the impact of fire on long-term biogeochemical fluxes from many types of ecosystems.

The San Dimas Experimental Forest offers an opportunity to investigate the long-term impact of fire on biogeochemical fluxes. Several natural burns have occurred in the last century as well as several prescribed burns conducted for research projects. The forest is impacted by high rates of deposition and the chaparral ecosystem of the forest has a quick recovery from fire, reaching climax structure in as little as 40 years. This pattern of rapid recovery to climax offers an opportunity to investigate the short and long-term impacts of fire on an ecosystem during a reasonable period of time as opposed to the much longer-term recovery from disturbance that occurs in more humid ecosystems. Additionally the highly variable annual precipitation of southern California offers an opportunity to investigate the impacts of climatic forcing on biogeochemical fluxes.

This paper answers three questions:

1) How do biogeochemical fluxes from a chaparral ecosystem respond after a fire and as the ecosystem recovers over the following 15 years?
2) What is the effect of inter-annual variability in precipitation on biogeochemical fluxes?
3) What part might fire management have in controlling the impacts of atmospheric deposition on chaparral ecosystems?

Methods

Site description

The San Dimas Experimental Forest (34° 12' N, 117° 40' W) was established in 1934 to study the interaction between chaparral ecosystems and water availability (Figure 1). The forest is subject to dramatic variation in annual precipitation and frequent wildfires (Meixner and Wohlgemuth this volume). Chaparral watersheds supply much of the groundwater recharge in southern California and it is one of the most extensive ecosystems in the American Southwest. In 1984 several prescribed burns were conducted that were intended to study the effect of fire on watershed hydrology, water quality and sediment transport (Riggan et al. 1994). These experiments took place against a backdrop of very high rates of atmospheric deposition (Riggan et al. 1985) and an existing disturbance history, as fire is a dominant ecosystem process in chaparral catchments (Minnich and Bahre 1995). This paper focuses on comparing nitrogen export from two catchments, Bell 3 (25.1 ha) which last burned in a wildfire in 1960 and represents a control and Bell 4 (15.9 ha) which was burned in the same wildfire and then burned under prescribed conditions in October of 1984 (Riggan et al. 1994). The data reported in this paper covers the time period from the fall of 1987 until the fall of 2001.

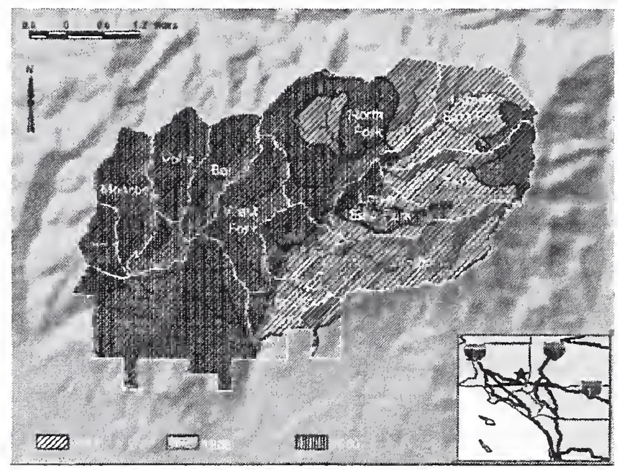

Figure 1. Location, catchment delineation and fire perimeters for the San Dimas Experimental Forest.

Water quality and streamflow measurement

Stream water samples were collected as often as four times daily using ISCO water samplers. Each sample was analyzed for NO_3^- using a Technicon AutoAnalyzer II. Based on past studies at San Dimas, stream concentrations of NO_3^- are equivalent to inorganic nitrogen flux since NH_4^+ concentrations are typically not detectable. Annual export and volume weighted mean (VWM) concentrations were calculated by integrating concentration values with streamflow. Streamflow in each catchment was monitored continuously using electronic data loggers with paper charts as backups. Streamflow records have only been quality checked for some of the years. For these years VWM concentration and export per unit area are reported. Additionally for all years the ratio of nitrate (NO_3^-) in streamflow of the burned catchment (Bell 4) to NO_3^- in streamflow of the control catchment (Bell 3) was calculated. When this ratio is greater than 1 the burned watershed had higher NO_3^- concentrations than the control.

Results and Discussion

In both watersheds stream NO_3^- concentrations vary wildly and have a high degree of correlation with streamflow, with higher concentrations during periods of high streamflow and lower concentrations during stream baseflow periods (Figure 2). Also noticeable in a temporal comparison is that the year closest to the 1984 fire (i.e. 1988) has the largest difference in NO_3^- concentrations between the two watersheds. In subsequent years NO_3^- concentrations were more closely matched between the two watersheds with more and more of the samples having lower concentrations on the burned catchment than on the control.

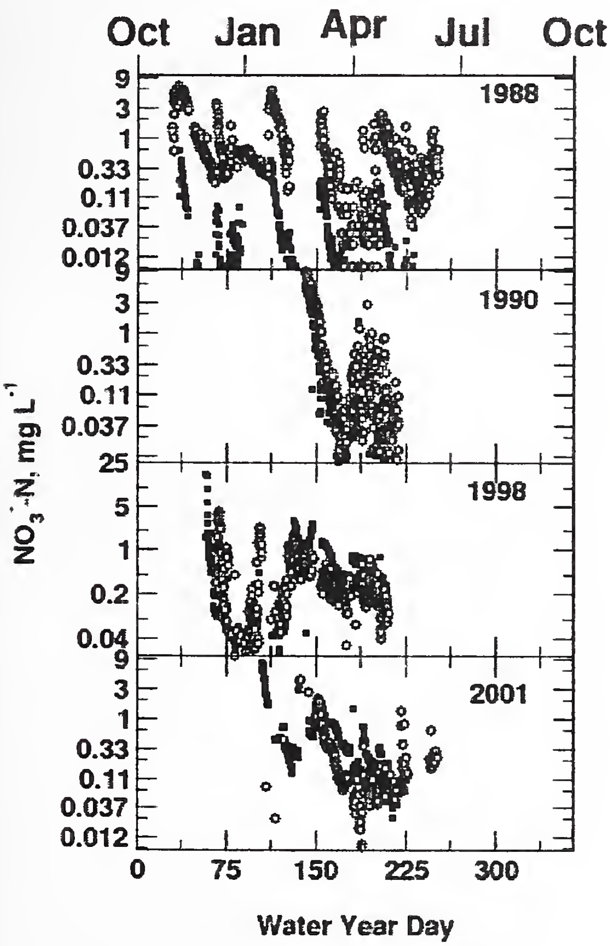

Figure 2. Stream NO_3^- concentrations for the watershed burned in 1984 (Bell 4, gray circles) and for the control watershed (Bell 3, black squares). Note that all y-axes are logarithmic.

The dramatic change in the difference in NO_3^- concentrations can be seen more clearly by looking at the ratio of concentrations between the burned (Bell 4) and unburned (Bell 3) catchments (Figure 3). Initially the ratios indicate that concentrations on the burned catchment are as much as 600 times those on the unburned, but as the experiment moves forward in time the ratio declines to being less than one for most of the last several years of the available data. This result is supported by looking at the number of simultaneous stream samples from the burned watershed that had lower concentrations than the control (Table 1). At the beginning of the period only 3 percent of the observations were lower on the burned catchment, but by the end of the period 85% of the stream samples on the burned catchment had lower concentrations.

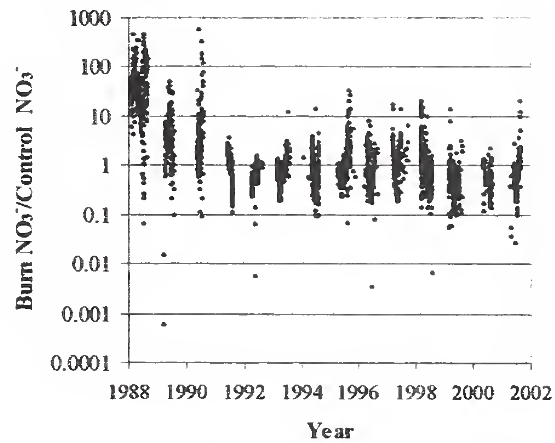

Figure 3. Ratio of NO_3^- in the burned catchment to that in the control catchment. Values greater than one indicate higher concentrations in the burned catchment and less than one indicate lower concentrations in the burned catchment as compared to the control catchment. Note logarithmic y-axis.

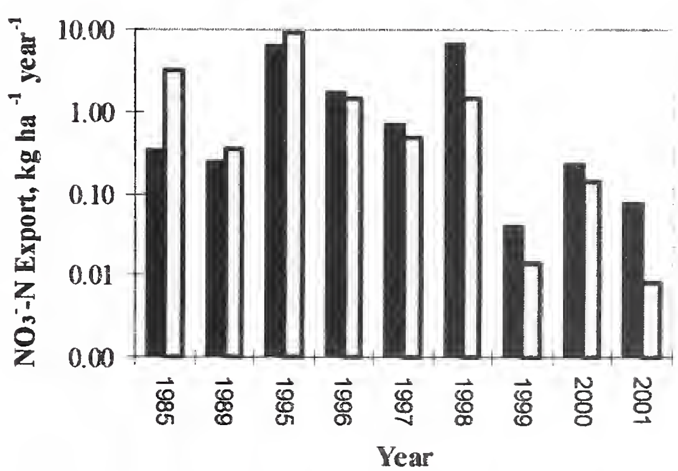

Figure 4. Nitrate export in each year with available streamflow data. Burned catchment in gray, unburned catchment in black. Note y-axis is logarithmic.

The change in concentrations for the individual samples is reflected in changes in annual export of NO_3^- (Figure 4) and in VWM results (Figure 5). At the beginning of the period NO_3^- export and VWM NO_3^- concentrations are higher on the burned catchment but by the end of the study period concentrations and export of NO_3^- are higher on the control catchment. Export per unit area is highly variable from one year to the next, mostly due to changes in streamflow related to the amount of rainfall and resulting streamflow in any one year (Figure 4). However the increases in export are further supported by increases in VWM NO_3^- concentration during the wetter years (Figure 5).

Table 1. Observed NO_3^- Burned < Unburned

	number of observations		
Year	Burn < Unburn	Total	Percent <
1988	10	303	3
1989	28	205	14
1990	18	169	11
1991	58	192	30
1992	281	405	69
1993	246	354	69
1994	119	172	69
1995	180	273	66
1996	188	247	76
1997	88	175	50
1998	299	415	72
1999	147	168	88
2000	79	95	83
2001	142	167	85

This increase in export in wet years and decrease in dry years indicates that there is a catchment hydrologic control on stream export of NO_3^-. This hydrologic control might be through shorter sub-surface residence times in the wet years and concurrent decreases in dentrification. Alternatively the inter-annual variability may be caused by year to year storage of inorganic nitrogen in the catchment in the dry years due to lack of water to move the NO_3^- out of the watershed (Vitousek et al. 1979). This second hypothesis is somewhat supported by the VWM data which shows heightened VWM NO3- concentrations in years like 2000 which followed dry years like 1999 (precipitation was similar in 2000 and 2001 but VWM concentrations were higher in 2000).

Fire recovery and biogeochemistry

Most studies of fire effects on ecosystem processes, hydrology or biogeochemical processes have focused on the immediate post-fire period and the large increases in streamflow, sediment export, nutrient export, and dissolved load that typically follow fires (e.g. Riggan et al. 1994, Townsend and Douglas 2000). However, as several hydrologists have shown, fire and ecosystem recovery can have long term impacts on water yield and thus might be expected to have an impact on other ecosystem processes (Kuczera 1987, Watson et al. 1999).

Our results illuminate several points about the effect of fire and the following recovery on biogeochemical fluxes from chaparral ecosystems. First, as noted above the immediate post-fire period has a noticeable increase in concentrations and export of dissolved nitrogen. Second, this increase persists for a period of up to 5 years in the post-fire environment. Third, the short period of increased export is followed by a longer period of decreased export in the burned system (Bell 4) compared to the unburned control (Bell 3).

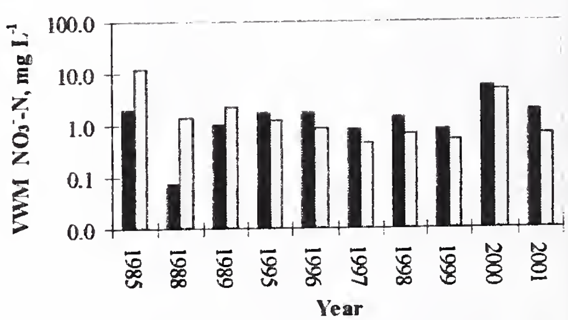

Figure 5. VWM NO_3^- for each year with available streamflow data. Burned catchment in gray, unburned catchment in black. Note y-axis is logarithmic.

In general it has been observed that canopy closure after a stand replacing chaparral fire takes five to seven years. Often times climax growth of chaparral vegetation can occur in as little as 20 years but may also take much longer. In both cases ecosystem response depends on precipitation and local site geology and soils (Horton and Kraebel 1955, Minnich and Bahre 1995). This pattern of vegetation recovery parallels the nitrogen export results presented here. Initially, after the fire has removed vegetation, NO_3^- export is dramatically increased (Figures 2-5 and Table 1). This increase may be due to several factors including: increases in streamflow in the post-fire environment (Loaiciga et al. 2001), decreased plant uptake and increased mineralization and nitrification rates, and loss of organic matter (Vitousek and Matson 1985). However, as the ecosystem recovered from disturbance and the vegetation grew back, nutrient export declined (later time periods in Figure 2-5) as the ecosystem sequestered more nitrogen and possibly other nutrients in response to the large losses that occurred during and after the fire. While studies of the long term effects of fire as an ecosystem disturbance are poorly known, other forms of ecosystem disturbance such as clear cut logging and other forms of vegetation removal have been studied in more depth (e.g. Likens et al. 1978, Vitousek et al. 1979). These studies have also shown an increase in nutrient export immediately following disturbance but a longer term decrease in nutrient export as the ecosystem recovers from the disturbance.

The data available in this study indicate another potential explanation for the pattern of higher concentrations of nitrate in the control catchment in the later years of our study. The control catchment in this study was the Bell 3 catchment that had last burned in 1960. Chaparral ecosystems burn on average every 40 to 60 years and therefore the Bell 3 watershed was at climax and ready to burn towards the end of this experiment. Other studies have indicated that as ecosystems approach their climax state that they will begin to leak nitrogen (Vitousek and Farrington 1997). With the high rates of atmospheric deposition in southern California, even rapidly aggrading catchments such as Bell 4 (burned) in the late 1990's still leak some inorganic nitrogen, possibly due to processes operating at seasonal transitions (Vitousek and Field 2001) and thus explaining the high concentrations of NO_3^- during periods of high streamflow. As the Bell 3 catchment approached 40 years since the last burn NO_3^- export and VWM NO_3^- concentrations may have increased as the vegetative N demand decreased with maturation of the chaparral ecosystem.

The interplay between the high rates of atmospheric deposition and fire history in the mountains of southern California indicates some possible challenges and opportunities for land managers. The effect of time since the last fire in chaparral ecosystems indicates that land managers need to take into account the immediate and long-term impacts of fire suppression, prescription, and management on catchment nutrient export. Chaparral catchments provide a water resource to southern California, and the short and long term effects of fire may have an adverse impact on the water quality of this resource. On the one hand fires in chaparral ecosystems cause dramatic short-term increases in nutrient export and NO_3^- concentrations, often above the federal drinking water standard of 10 mg L^{-1}. On the other hand fire suppression may lead to similarly high NO_3^- concentrations (Figure 2, 1998 unburned data).

Conclusions

This study provides preliminary evidence that long-term ecosystem recovery from fire in chaparral ecosystems leads to decreases in nutrient flux. This decrease in the long term follows an initial large flux of inorganic nitrogen in the immediate post-fire period. Export as well as VWM concentrations increase dramatically in wet years and are orders of magnitude lower in dry years. This inter-annual variability in export that is dependant on precipitation as well as antecedent conditions indicates that there is a hydrologic control on nutrient export from chaparral catchments. Finally, the influence of the post-fire recovery process and the hydrologic controls on nutrient export indicate some interesting trade-offs for land managers as they try to manage fire in chaparral ecosystems under high N deposition conditions. This study hopefully will help them balance the countervailing impacts of atmospheric deposition and fire recovery on the water quality consequences of these two activities.

Acknowledgments

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Water Quality and Quantity IV

Multi-Scale Assessment of the Extent and Effects of Calcium Depletion in Forest Soils of the Upper Delaware River Basin

Peter Murdoch, Rakesh Minocha

Abstract

The U.S. Geological Survey and the U.S. Forest Service have been testing collaborative monitoring strategies in the northern Delaware River Basin through a set of issue-focused parallel studies. In this study, the effect of soil calcium on tree condition and calcium concentrations in nearby streams were assessed using plot-scale, watershed-scale, and regional sampling approaches. Long-term records at research watersheds in the Catskill Mountains indicate a downward trend in stream calcium concentrations during the 1980s and 1990s. An experimental clearcut of a forest in a calcium-poor soil in the Catskill Mountains resulted in a significant stream export of soil calcium in the 2 years following harvest, and export remains elevated above pre-cut concentrations 6 years after harvest. Analysis of tree foliar chemistry indicates that tree health was negatively correlated with elevation and soil base-cation saturation at this experimental watershed. A survey of water quality in the nearest, forested first-order stream to the USFS Forest Inventory and Analysis (FIA) plots in the Delaware basin revealed a band of low-calcium streams that extends from the eastern Catskills south to the Delaware Gap region, then west to the western Pocono Mountains. Calcium concentrations in soils collected at FIA plots in the Delaware Basin indicate a similar regional pattern. The combination of intensive and extensive data collection, and integration of the forest-, soil-, and water-sampling programs of the USGS and USFS is providing a regional picture of the extent of soil calcium depletion and it's effects in the upper Delaware River Basin.

Keywords: calcium depletion, acid rain, integrated science

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Post-Fire Erosion Control Research on the San Dimas Experimental Forest: Past and Present

Peter M. Wohlgemuth

Abstract

The San Dimas Experimental Forest (SDEF) was established in the early 1930s to document and quantify wildland hydrology in the semiarid chaparral-covered steeplands of southern California. Concomitantly, the nearly seventy years of accumulated watershed research in this fire-prone ecosystem has produced invaluable information on post-fire erosion and the effectiveness and consequences of post-fire erosion control treatments. On average, first-year post-fire watershed sediment yield is 35 times greater than comparable unburned annual levels. This accelerated erosion can cause site degradation and threatens life, property, and infrastructure at the adjacent wildland/urban interface. To mitigate undesirable consequences of post-fire accelerated erosion, land managers have developed a program of hillslope and stream channel emergency rehabilitation treatments as erosion control measures. The SDEF has been the site on which many of these erosion control practices, both past and present, have been tested. In the 1960s, some labor-intensive treatments were shown to have no effect on reducing post-fire erosion. At the same time, more radical ground-disturbing treatments that were marginally effective in the short-term have persisted on the landscape and altered the subsequent sediment fluxes through these watersheds. In September 2002, virtually the entire SDEF burned in the Williams Fire. This allowed the implementation of a new series of emergency rehabilitation treatments for which the effects and consequences are largely unknown. Preliminary results suggest that an aerial application of polyacrylamide did nothing to reduce post-fire sediment yield, but that prefabricated wooden channel checks may be an effective post-fire rehabilitation tool.

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Keywords: erosion, fire, erosion control, post-fire rehabilitation

Introduction

Wildfire can dramatically alter the erosion response of upland landscapes. With the removal of the vegetation canopy and surface organic material, rainfall interception is reduced (Hamilton and Rowe 1949) and the denuded hillsides are subjected to unimpeded raindrop impacts (Wells 1981). In addition, the combustion of soil organic matter can create a subsurface water-repellent layer that restricts infiltration and promotes overland flow (DeBano 1981), enhancing sediment yield (Hamilton et al. 1954, Pase and Ingebo 1965, Heede et al. 1988). In southern California, first-year post-fire sediment yield is 35 times greater on average than comparable unburned annual levels (Rowe et al. 1954).

Post-fire erosion, sedimentation, and flooding are ongoing problems in the fire-prone ecosystems of the southwestern United States. The climatic patterns that produce highly flammable brush vegetation also generate weather conditions that promote high-severity wildfires. Accelerated post-fire erosion and flooding can threaten life, property, and infrastructure at the wildland/urban interface, where burgeoning population centers impinge on adjacent steep mountain fronts.

To mitigate undesirable consequences of post-fire accelerated erosion, land managers have developed a program of hillslope and stream channel emergency rehabilitation treatments as erosion control measures. The goal of these treatments is to cost-effectively protect the onsite and downstream values at risk until the native vegetation community can be reestablished. Unfortunately, the benefits of many of these erosion control measures have yet to be quantitatively demonstrated in rigorous field studies (Robichaud et al. 2000). However, some of the best post-fire erosion control research over the last half-century has been conducted on the San Dimas Experimental Forest.

Site description

Located in a front range of the San Gabriel Mountains about 45 km northeast of Los Angeles, California, the San Dimas Experimental Forest (SDEF) is a 6945 ha research preserve administered and operated by the USDA Forest Service, Pacific Southwest Research Station (Figure 1). With its headquarters at Tanbark Flat (34° 12' N latitude, 117° 46' W longitude), the SDEF has been the site of extensive hydrologic monitoring for nearly seventy years.

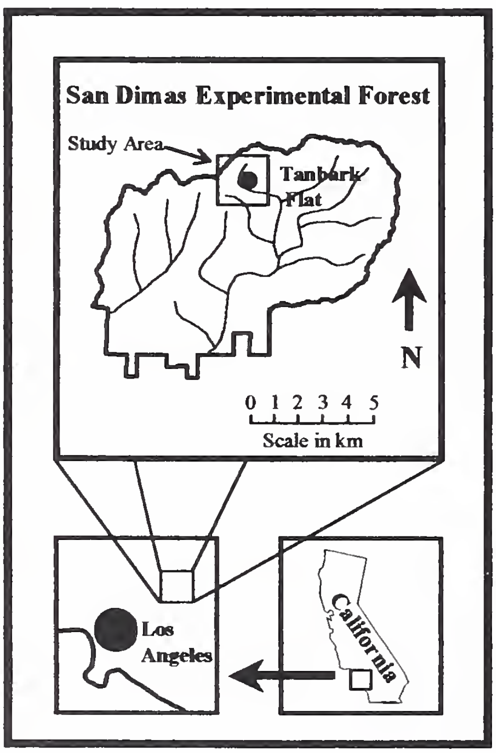

Figure 1. Location map of the San Dimas Experimental Forest.

Topography in the SDEF consists of a highly dissected mountain block with narrow, steep-walled canyons (slope angles average 68 percent) and steep channel gradients (average of 15 percent). Elevations range from 457 m to 1677 m. Bedrock geology is dominated by Precambrian metamorphics and Mesozoic granitics that produce shallow, azonal, coarse-textured soils (Dunn et al. 1988).

The SDEF experiences a Mediterranean climate, characterized by cool, moist winters and hot, dry summers. Mean annual precipitation, falling almost exclusively as rain, is 714 mm (62-year record), but rain during individual years can range from 258 to 1595 mm. Over 90 percent of the annual precipitation falls between the months of November and April, with 10 percent of the storms producing over 50 percent of the total rain (Wohlgemuth 1996).

Vegetation in the SDEF consists primarily of mixed chaparral. Plant cover on south-facing slopes ranges from dense stands of chamise (*Adenostoma fasciculatum*) and ceanothus (*Ceanothus* spp.) to more open stands of chamise and sage (*Salvia* spp.). North-facing hillsides are dominated by scrub oak (*Quercus berberidifolia*) and ceanothus, with occasional hardwood trees – live oak (*Quercus agrifolia*) and California laurel (*Umbellularia californica*) – occurring on moister shaded slopes and along the riparian corridors (Wohlgemuth 1996). Forest species, dominated by Big Cone spruce (*Pseudotsuga macrocarpa*), occur in the higher elevation eastern end of the SDEF (Dunn et al. 1988).

Post-fire erosion control treatments

Landscape-level post-fire erosion control treatments attempt to reduce and delay the onslaught of accelerated sediment yield that typically follows a wildfire. Many types of treatments, both on the hillslopes and in the stream channels, have been utilized over the years. These usually take the form of mechanical barriers to retain debris or enhanced ground covers to reduce the erosive power of rainsplash and overland flow. Modern treatments also include applications of chemical wetting agents or soil flocculants to promote infiltration. For a review and extensive discussion of post-fire rehabilitation treatments, see Robichaud et al. (2000).

Methods

The effectiveness of the different erosion control treatments on the SDEF was evaluated by comparing the sediment yield from small headwater catchments. Sediment was trapped and measured behind earth-filled dams. Sediment yields were calculated using an engineering end-area formula (Eakin 1939) based on repeated sag tape surveys of permanent cross sections (Ray and Megahan 1978). To normalize for catchments of different sizes, comparisons in sediment yield were made as cubic meters per hectare.

Johnstone Fire

The Johnstone Fire burned nearly the entire SDEF in July of 1960. Following the fire, twenty watersheds were selected for study that were as similar as possible in size (.75-2.5 ha), shape, and aspect. As there was no pre-treatment calibration of sediment yields between the study watersheds, inherent site differences in potential erodibility were assessed (based on slope angle, channel gradient, rockiness, and soil depth) and distributed evenly among the treatments.

Four seeding treatments and three types of mechanical barriers were tested as post-fire erosion control measures. The seeding treatments consisted of broadcast sowing a mixture of annual grasses at rates of 0.46 kg/ha and 3.66 kg/ha, as well as broadcast sowing a mixture of perennial grasses at rates of 0.82 kg/ha and 3.66 kg/ha (Rice et al. 1965). In addition, the areas sown to perennials were sprayed with strong herbicides to help establish these grasses by reducing competition from the re-growing brush species.

The mechanical treatments after the Johnstone Fire included side slope stabilization, contour trenching, and channel stabilization (Rice et al. 1965). Side slope stabilization consisted of planting barley in hand-hoed rows at 0.6 m vertical contour intervals. As the intent was to create closely spaced barriers to the overland flow of water and sediment, this was considered a mechanical rather than a vegetative treatment. Contour trenches were created by cutting slightly insloped horizontal platforms across the hillslopes with a bulldozer. These benches – intended to break up overland flow, increase depression storage, and promote infiltration – were established as close together as the terrain would permit (12 m apart on gentler slopes and 27 m apart on steeper hillsides). Channel stabilization was accomplished by building small gravity check dams roughly 30 m apart using soil cement. Although these dams would trap only a small wedge of transported sediment, the intent of the barriers was to serve as grade control structures that would prevent channel incision that could produce landsliding on the adjacent hillsides (Rice et al. 1965).

The four vegetative treatments were crossed with the three mechanical treatments plus their corresponding controls to yield a five by four matrix design with one watershed unit per experimental cell. Analysis of the data was performed by multiple linear regression (Rice et al. 1965).

Williams Fire

Nearly all of the SDEF burned again in the Williams Fire of September 2002. Although several post-fire erosion control treatments were tested after this burn, only two are reported here. A portion of the burned area was sprayed with polyacrylamide (PAM), a proprietary soil-flocculating agent. Applied by a helicopter, the intent of this treatment is to aggregate the fine soil particles, thus promoting infiltration and thereby reducing overland flow (Flanagan and Chandhari 1999), especially in areas of suspected water repellent soils. Other sections of the Williams Fire were treated with FlowCheck™ log structures in the stream channels. Manufactured by Forest Concepts, LLC from small diameter tree sections (see Figure 2), these prefabricated barriers were placed roughly 5-10 m apart along the stream courses. They were intended to serve as storage sites and grade control structures to prevent the scouring of channel sediment deposits by the accelerated post-fire runoff.

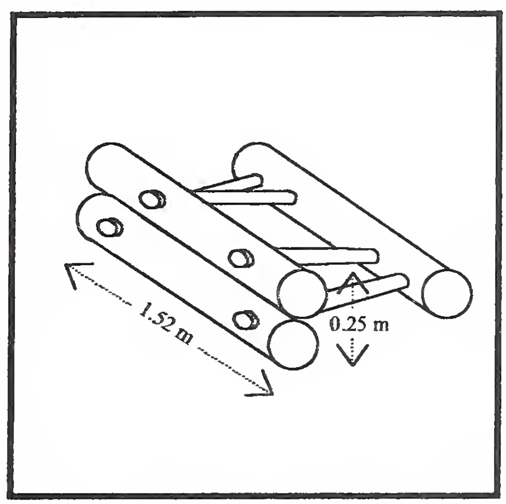

Figure 2. Schematic of a FlowCheck™ log structure.

Following the Williams Fire, four of the original twenty experimental watersheds were re-activated. One of these was sprayed with PAM, while an adjacent catchment was left untreated. After the Johnstone Fire, the PAM-treated watershed had check dams built in the channels, but the other had been subjected to side slope stabilization. Both these watersheds were seeded with annual grasses, although at different densities. A third watershed had FlowCheck™ log structures placed in the stream channels, while a neighboring catchment was left untreated. After the Johnstone Fire, the FlowCheck™-treated watershed was not subjected to any vegetative treatment, but the neighboring control had been seeded with annual grasses. Both of these watersheds had contour trenches carved into the hillsides. Table 1 shows the relationships between the Johnstone Fire treatments and the subsequent Williams Fire erosion control measures. As the modern treatments are unreplicated, the resulting comparisons are not necessarily generalizable.

Table 1. Watersheds used to test both the Johnstone Fire and Williams Fire erosion control treatments.

Vegetative Treatment	**Mechanical Treatment**		
	Side Slope Stabilization	Contour Trenches	Channel Stabilization
None		FlowCheck™	
Low Density Annuals	PAM Control	FlowCheck™ Control	
High Density Annuals			PAM

Results and Discussion

For these studies, sediment yield is the integrated output of debris from a watershed unit that received a uniform experimental treatment. A danger in re-activating existing study watersheds is the persistence of the previous treatments. The residual effects of erosion control measures from the Johnstone Fire may have influenced the results of treatments applied after the Williams Fire. In fact, the contour trenching and herbicide measures following the Johnstone Fire have altered the sediment fluxes through these small watersheds (Wohlgemuth 1996). However, the seeding of annual grasses, the side slope stabilization, and the channel stabilization has had no apparent effect on subsequent vegetation development or long-term sediment fluxes (Wohlgemuth 1996). Thus, in terms of the effects on present sediment fluxes, both sets of Williams Fire treatments – the PAM-treated watershed and its associated control, and the FlowCheck™-treated catchment and its associated control – had comparable Johnstone Fire treatment histories.

Johnstone Fire

The first winter after the Johnstone Fire was one of the driest on record, so few of the seeded grasses germinated and little sediment yield was produced. The seeding and herbicide treatments were repeated the following year, and the study area received nearly normal rainfall amounts. However, second-year sediment yield values indicated that none of the seeding treatments were effective erosion control measures (Rice et al. 1965). In contrast, side slope stabilization, contour trenches, and channel stabilization generated 35, 40, and 65 percent, respectively, of the sediment yield produced from watersheds without mechanical treatments. From this, Rice et al. (1965) concluded that measures designed to prevent the concentrated flow of water and the entrainment of sediment were the superior erosion control treatments.

Williams Fire

In the first winter after the Williams Fire, the study area received slightly below normal precipitation. Moreover, the amounts and disposition of the rainfall were nearly identical to the second year after the Johnstone Fire: early gentle rains, followed by a dry period, then more intense storms later in the winter season.

The sediment yield results for the two post-fire treatment comparisons are arrayed in Table 2. Based on these preliminary data and invoking the caveat of no replication, the PAM treatment appears to have had no effect as an erosion control treatment. Observations on the study area over the course of the winter revealed pervasive rilling on all watersheds, indicating substantial surface runoff. Although infiltration tests were not performed on the sites, presumably the PAM did not work as it was intended. In contrast, the FlowCheck™ log structures appear to have reduced watershed sediment yield. Virtually all of the 23 structures filled with sediment and only a few were subject to undercutting or side cutting. The savings in debris retention and the protection against channel incision could easily account for the differences in reduced sediment yield compared to the control.

Table 2. Sediment yield after the Williams Fire.

Treatment	Sediment yield (m^3/ha)
PAM	
Treated	34.2
Control	26.0
FlowCheck	
Treated	11.1
Control	34.7

Conclusions

Accelerated post-fire erosion and sediment yield is an ongoing problem in fire-prone Southwestern ecosystems. Land managers will continue to seek erosion control measures that are both cost-effective and environmentally benign. However, it is critical that prospective treatments are rigorously tested before their widespread application.

Many lessons have been learned over the past half-century of post-fire erosion control research on the San Dimas Experimental Forest. Some labor-intensive treatments have been shown to be ineffective at reducing erosion or would not be cost-effective at a landscape level. Some ground-disturbing measures of moderate efficacy continue to persist long after the post-fire emergency is over. Moreover, these treatments have altered the sediment fluxes through these small study watersheds. Preliminary results suggest that PAM did nothing to reduce post-fire sediment yield, but that FlowCheck™ structures may be an effective post-fire rehabilitation tool. Continued study will help assess the effectiveness and consequences of the untested treatments applied following the Williams Fire of 2002.

Acknowledgments

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Climate Variability, Fire, Vegetation Recovery, and Watershed Hydrology

Thomas Meixner, Peter M. Wohlgemuth

Abstract

The San Dimas Experimental Forest was established in 1934. A large database of fire history, and stream flow exists for several locations within the forest. San Dimas was selected as a perfect example of the hydrology, geology, and ecology of the mountains of southern California. As such the long-term data set provides the best examples available of rainfall-runoff relationships in a mountainous Mediterranean climate. One of the most important ecological processes operating at San Dimas is the frequent stand replacing fires that occur approximately every 40 years. The effect of fire on streamflow is a pertinent topic considering the current national discussion about changes in fire policy in the western United States. The data sets at San Dimas provide an opportunity to investigate what the short and long term impacts of fire are on water resources in chaparral ecosystems. In particular the fires of 1938 and 1960 provide an opportunity to investigate the effect of fire on streamflow response. Immediately after the fires the well-known fire-flood response of chaparral watersheds is noted with extremely large flood peaks possibly due to the combined effects of removal of vegetation and litter by fire as well as the presence of hydrophobic soils. However longer term impacts of fire are noticeable, lasting as long as 20 years and are most likely related to aggrading vegetation coverage and the linked increase in evapotranspiration. Around 1960 several watersheds were type converted to annual and perennial grasslands from their native chaparral, this increased streamflow to the present day and offers insight into the importance of deeply rooted vegetation on summer streamflow in seasonally dry climates.

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Keywords: Mediterranean climate, fire, streamflow, water yield, baseflow recession

Introduction

Fire and its effect on watershed hydrology and water quality has been a well studied process for many decades now (e.g. Hoyt and Troxel 1934, Loaiciga et al. 2001). These studies have typically focused on the immediate post-fire period and the large increases in flooding, sediment production and nutrient export (DeBano et al. 1998). While these processes are of obvious importance to assist decision makers in hazard avoidance during the post-fire period they are not the end of the story.

Ecosystem disturbances, such as fire, have short and long term impacts on ecological processes. Several studies in the last several decades have shown that while clear-cutting or stand replacing fires can cause immediate large increases in streamflow the longer term impacts of either disturbance are longer term decreases in annual streamflow (Kuczera 1987, Hornbeck et al. 1997). Additionally, vegetation type conversion experiments offer an important contrast to fire effects studies and permit a direct investigation of vegetation effects on ecological processes. Thus they offer a way to understand how fire influences watershed hydrology in the short and long terms (Bosch and Hewlett 1982).

In this study we investigate the following three questions:

1) What are the short and long-term effects of fire and vegetation type conversion on watershed hydrology?
2) What can these experiments tell us about the importance of the deeply rooted vegetation that is native to the chaparral and similar Mediterranean ecosystems?
3) In moving to a scale larger than those studied at San Dimas, what problems might be confronted?

Methods

Site description

The San Dimas Experimental Forest (SDEF) (34° 12' N, 117° 40' W) was established in 1934 to study the interaction between chaparral ecosystems and water availability (Figure 1). The Forest originally gauged a total of 17 catchments with area larger than 10 ha. Most of the stream gauging network was destroyed in a catastrophic wildfire in July 1960. Of the original gauged catchments, 5 catchments have been gauged continuously since the inception of the experimental forest. The forest is subject to dramatic variation in annual precipitation (Figure 2) and frequent wildfires. Chaparral watersheds supply much of the water that recharges to groundwater in southern California and it is one of the most extensive ecosystems in California. In 1984 several prescribed burns were conducted that were intended to study the effect of fire on watershed hydrology, water quality and sediment transport (Riggan et al. 1994).

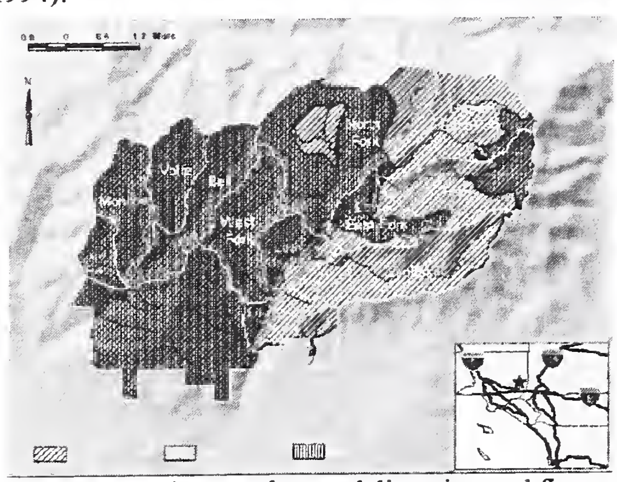

Figure 1. Location, catchment delineation and fire history for the San Dimas Experimental Forest.

Experimental framework

This paper focuses on comparing annual water yield and daily streamflow for several different natural and human induced experiments that have been conducted at SDEF (Table 1). First, the effect of a 1938 fire that only burned the three small Fern watersheds are assessed by comparing annual runoff from these catchments to the Bell 2 catchment that was not burned. Second, the Bell 1 and Bell 2 watersheds were type converted from native chaparral to grasslands in 1960 and 1958 respectively. The effect of this type conversion on water yield and daily baseflow recession will be analyzed by comparison to the Volfe catchment. Third, Volfe catchment streamflow prior to and following the 1960 fire was used to investigate the impact of the fire on annual water yield and baseflow recession. Fourth, we will look at the impact of a controlled burn conducted in 1984 on annual water yield in the Bell 4 catchment, which was burned, to annual water yield in the Bell 3 catchment which serves as a control for this experiment (Riggan et al. 1994). Finally, annual runoff from the SDEF catchments is compared to annual runoff from nine neighboring USGS gauges to assess the effect of scale on annual runoff (Table 2). Between catchment comparisons used runoff depth and similar time periods.

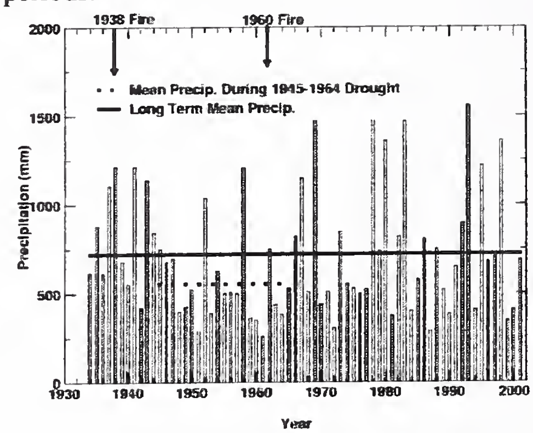

Figure 2. Annual Precipitation for SDEF. Note extended drought from 1945 to 1965.

Table 1. SDEF Catchment Descriptions

Catchment	Area (ha)	Burned	Treatment, year
Bell 1	31	1960	Grassland, 1960
Bell 2	40	1960	Grassland, 1958
Bell 3	25	1960	Chaparral
Bell 4	16	1960, 1984	Chaparral
Volfe	299	1960	Control
Fern 1	14	1938	No Treatment
Fern 2	16	1938	No Treatment
Fern 3	21	1938	No Treatment

Results

Unfortunately there is no available data for the pre-fire period for the Fern catchments but considering the similar size of the Fern catchments to the Bell 2 catchment they are expected to have similar or slightly lower annual runoff. The 1938 fire caused a significant increase (as much as 7 times) in annual streamflow from the three small Fern watersheds as compared to annual water yield from the Bell 2 catchment. This initial post-fire increase in water yield was short lived and disappeared by 1942. From 1942 until the end of the gauging record in 1957 the ratios of annual runoff indicate that runoff from the Fern's during this longer post-fire period was below what it would otherwise have been.

Table 2. USGS Gauges Used for Comparison

USGS Gauge	Area, ha
Fish Canyon	1647
San Antionio Canyon	4274
Sawpit	1373
Santa Anita	2515
Arroyo Seco	4144
Devil Canyon	1422
Little Dalton	705
San Dimas Canyon [1]	4740
Dalton Canyon	1875

1 San Dimas Canyon record combines records from above and below the San Dimas Canyon Dam.

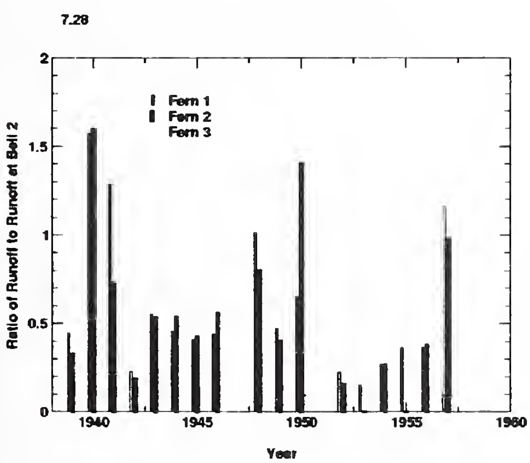

Figure 3. Ratio of annual runoff (mm) between three Fern catchments, burned fall of 1938, to annual runoff in the Bell 2 catchment.

The long-term impact of type conversion at SDEF was a persistent effect on annual water yield into at least the 1980s (Figure 4). Additionally the type-conversion has had its biggest impact on water yield in years with relatively low precipitation (less than 700 mm). Dry years such as these typically produce no runoff in mature chaparral watersheds (Figure 5). Further the effect of the type conversion is not isolated during the wet winter rainy season but is most prominent in increased streamflows during the summer and by a notably more gradual streamflow recession (Figure 6).

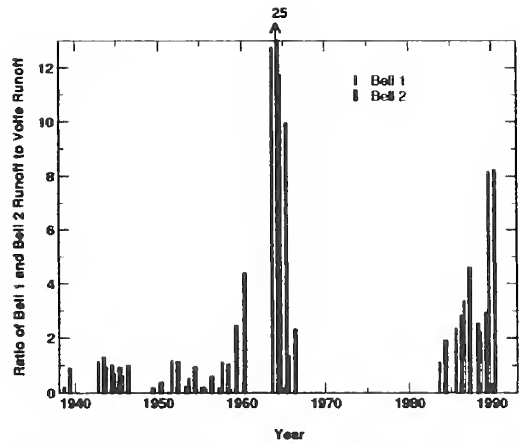

Figure 4. Ratio of annual runoff in Bell 1 and Bell 2 catchments to runoff from long-term control Volfe catchment.

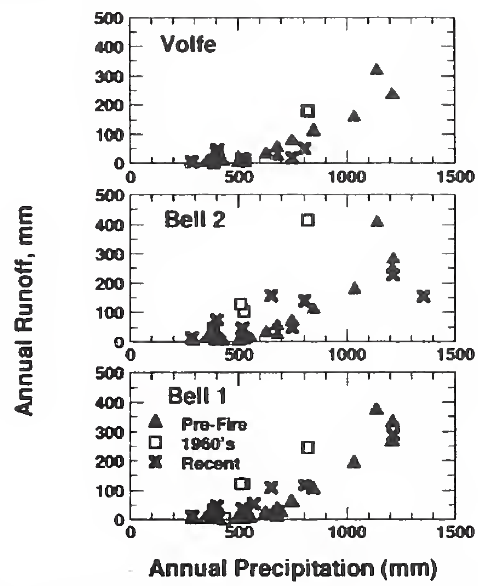

Figure 5. Annual runoff vs. annual precipitation for Volfe and Bell 1 and Bell 2 catchments

The response of streamflow in the Volfe catchment to the 1960 fire indicates an increase in water yield that is in part due to more gradual streamflow recession during the summer period (Figures 5 and 7). The cause of this increased summer streamflow is likely decreased evaporation due to decreased vegetation following fire or type conversion. The increase in streamflow is short lived and by the 1980s Volfe catchment streamflow returns to pre-1960 conditions.

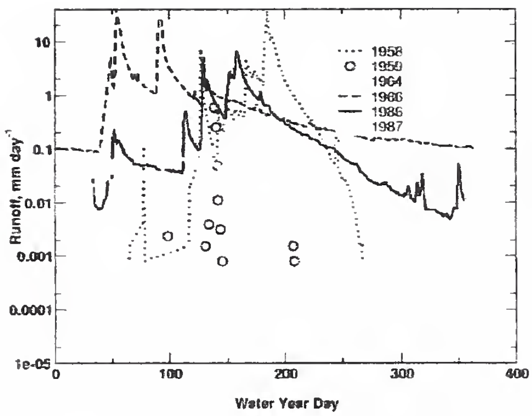

Figure 6. Daily streamflow for Bell 1 catchment (water year starts on October 1st). The years 1959, 1964 and 1987 were dry while 1958, 1966 and 1986 were wet years. Note more gradual streamflow recession in 1964 and 1966 as compared to 1958. Note that in 1986 and 1987 recession rates fall between pre-conversion conditions (1958 and 1959) and immediate post-conversion streamflow recession (1964 and 1966).

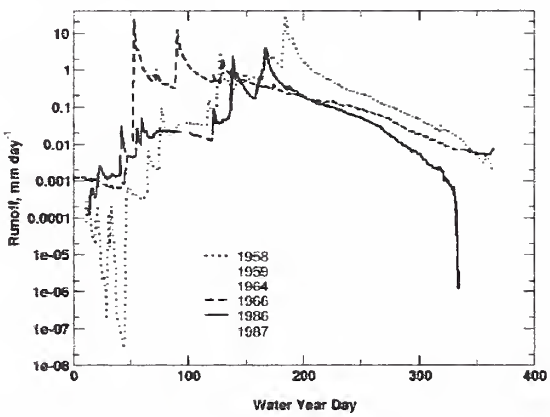

Figure 7. Daily streamflow for Volfe Catchment(water year starts on October 1st). Years are the same as for Figure 6.

The long pre-fire runoff data for both catchments indicates that runoff depth on Bell 4 is typically about 10% greater than runoff on Bell 3. After the 1984 prescribed burn in the Bell 4 catchment streamflow is greatly increased compared to the control Bell 3 catchment for 1985. The late 1980s and early 1990s indicate that this increase in water yield lasts until 1997. In 1998 annual runoff is greater in the control catchment (Bell 3) and this pattern persists and grows larger through the end of the data set.

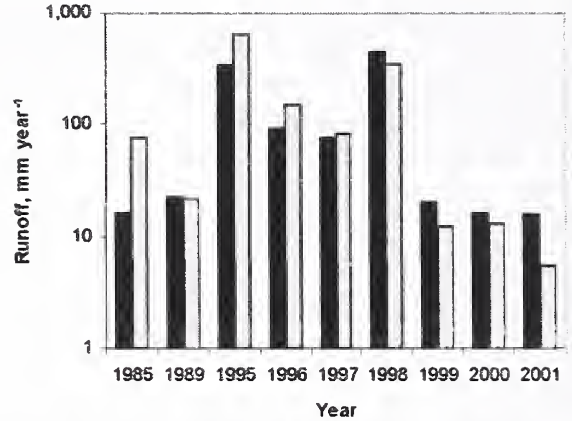

Figure 8. Annual runoff from Bell 3 (control, black) and Bell 4 (burned 1984, gray) catchments.

Annual runoff from the catchments of the SDEF increases with watershed area. The USGS gauges generally measure streamflow at a larger scale, but their streamflow records show an increase in streamflow with increasing watershed area but the runoff volume is significantly smaller than the SDEF catchments. The one USGS gauge that plots with the SDEF gauges is Little Dalton Canyon. This gauge is immediately adjacent to the SDEF with similar landscape position and geology to the catchments of the SDEF.

Discussion

Long-term effects of fire

The results indicate that fire, as reported many times previously, increases streamflow at daily and annual time scales e.g. (Hoyt and Troxel 1934). In this behavior fire as an ecosystem disturbance is not very different from other ecosystem disturbances such as clear cutting (Hornbeck et al. 1997). The results show that while the short-term impact of fire is to increase streamflow the longer-term affect is to decrease streamflow. This was evident in the results from the Fern catchments and from the recovery of the Bell 4 catchment from the 1984 fire (Figures 3 and 8).

The short-term increase in streamflow is due to decreased evapotranspiration (ET) losses and perhaps increased runoff during winter rain storms. Decreased ET is due to the loss of vegetation in the post-fire environment (Kuczera 1987, Hornbeck et al. 1997). The increase in winter runoff is possibly due to the loss of vegetation, surface cover and post-fire soil hydrophobicity (Letey 2001).

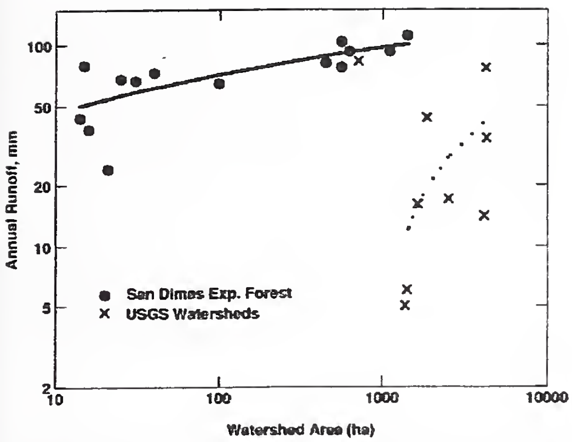

Figure 9. Annual runoff versus watershed area for San Dimas and USGS watersheds.

The results from the type-conversion experiments indicate that the type conversions induce an increase in flow during summer periods that is similar to the increases in stream baseflow due to a fire. Furthermore the type-conversions have caused a persistent increase in streamflow for well over 40 years but at a level somewhat decreased from the initial period. This result indicates that the large number of type-conversion experiments conducted globally during the 1950s and 1960s (Bosch and Hewlett 1982) should be revisited to see if the effect of conversion has persisted. The persistence of the type-conversion effect is likely dependant on ecosystem properties such as the ability of the original vegetation species to reinvade after the conversion. Native chaparral species have been slow to reinvade the converted catchments due to low seed dispersal and the dependence on fire for germination of many of the species (R.A. Minich, personal communication, Thanos and Rundel 1995).

Importance of deeply-rooted vegetation

The increase in summer streamflow from both the type conversion and after fires indicates that some process related to the vegetation is removed from affecting streamflow under the post-fire and post-conversion conditions. Most notably streamflow recession is more gradual with the absence of vegetation. Theoretically streamflow recession is treated as the draining of an aquifer. A given streamflow is associated with a given storage in that aquifer (Wittenberg and Sivapalan 1999). The implication of the more gradual baseflow recession observed is that there must be another mechanism draining aquifer storage when chaparral vegetation is mature. The most likely alternative draining mechanism to streamflow is ET. Others have shown that adding ET losses to the theoretical equations that model baseflow recession would cause more rapid decreases in streamflow than in the absence of ET (Weisman 1977). The vegetation of Mediterranean climates is widely noted for its deep roots that are used to survive the long hot, dry summers (Schenk and Jackson 2002). The data at SDEF offers a way to investigate the water demand of this vegetation; by comparing baseflow recession rates in the type converted systems to those in the control catchments.

Scale effects on water yield

The original work on paired catchment studies instigated much debate about whether the results at the ~100 ha scale would be observable at larger scales (Hoyt and Troxel 1934). The results here indicate some hazard in extrapolating from the results at SDEF to the larger southern California region and chaparral and Mediterranean ecosystems in general. The larger gauged catchments of the USGS gave reduced annual runoff amounts as compared to the headwater catchments of the SDEF (Figure 9). The reason for this difference may be increased recharge to deep hard rock aquifers, the USGS gauges are located on top of valley alluvial deposits, and losses of stream water to ET by riparian vegetation.

Relevance to California water resources

The results of this study are of critical importance to water resources management in California. Chaparral ecosystems are large in their extent and critical for groundwater recharge in southern California. Fire plays such a central role in chaparral ecosystems that even small hydrologic effects of fire and fire recovery would be felt throughout the water resources infrastructure of southern California. Past studies on the effect of fire on water yield in the chaparral have focused on the immediate increase in annual water yield (Turner 1991, Loaiciga et al. 2001). While relevant, the short-term positive of fire may be drowned out by the long-term effect of vegetation recovery noted here. This long-term loss of water from chaparral watersheds represents an adverse impact of fires in chaparral that is not well recognized and it could jeopardize water resources system reliability (Kuczera 1987, Watson et al. 1999). Additionally, it is likely that larger chaparral fire patterns of today have different catchment scale water yield effects than the more patchy nature of chaparral fires prior to large scale suppression during the last century (Minnich and Bahre 1995).

Conclusions

The results from this study indicate that type conversion has a long lasting positive effect on water yield from chaparral watersheds. This result may be ecosystem specific. While fire has a short term positive effect on water yield, longer term it causes a decrease in water yield. These two results indicate that fire suppression policy could have a beneficial impact on water resources through increasing return periods of fire or a negative impact due to the large scale of chaparral wildfires under suppression regimes. Finally, the research results at San Dimas need to be investigated from a broader regional perspective since comparisons with USGS gauges indicate a difference in process at the larger scales that are most relevant to water resources decision makers.

Acknowledgments

The United States Forest Service supported this research. We wish to especially thank Phil Riggan whose visionary 1984 experiment made a portion of this work possible. Additionally the many individuals who have helped transfer the SDEF streamflow data set into electronic form are to be thanked particularly Gwyn Dixon and Mike Oxford.

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Historical and On-Going Hydrologic and Sediment Transport Research at Little Granite Creek near Bondurant, Wyoming

Sandra E. Ryan, Mark K. Dixon, Kathleen A. Dwire, William W. Emmett

Abstract

Measurements of sediment flux and flow were made during the course of 13 runoff seasons in a small watershed near Bondurant, Wyoming. Begun in 1982 through the combined efforts of the U.S. Geological Survey and U.S. Forest Service, the database from Little Granite Creek represents one of the most comprehensive sources of information on stream transport processes available for an individual site. In August 2000 a wildfire burned portions of the watershed, creating an opportunity to monitor the impacts of fire on stream processes and water quality. Scientists from the U.S. Forest Service, Rocky Mountain Research Station initiated studies in 2001 to assess the magnitude of increased runoff and sedimentation generated by spring snowmelt and summer thunderstorms following wildfire. Additional work was initiated in 2002 on the post-fire dynamics of organic matter (dissolved, fine, and coarse), the status of aquatic macroinvertebrates and fish, and the re-establishment of vegetation in the riparian corridor. In this paper, we describe the monitoring effort in the Little Granite watershed and present preliminary results from the first year after burning. Such data are useful for planning future Burned Area Emergency Rehabilitation (BAER) efforts, validating erosion models (such as WEPP), and evaluating long-term sensitivity of aquatic systems to wildfire.

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Keywords: fire effects, runoff, stream sedimentation, water quality, large wood

Introduction

In August 2000, a fire in the Gros Ventre Wilderness Area on the Bridger-Teton National Forest in northwestern Wyoming burned most of the forest vegetation in the Boulder Creek watershed. This watershed constitutes nearly 40% of the area of Little Granite Creek where researchers from the U.S. Geological Survey [USGS] and U.S. Forest Service [USFS] collected data on sediment transport processes (bedload and suspended sediment) during 13 runoff seasons between 1982 and 1997 (Ryan and Emmett 2002). Other data collected include a flow record of several years, channel surveys, and data on dissolved loads and aquatic communities. Hence, the Boulder Creek Fire within the Little Granite Creek watershed presents a unique opportunity for evaluating the local and downstream effects of fire-related sedimentation relative to established baselines, as well as interactions among physical processes, recovery of riparian vegetation, and water quantity and quality. Further, most of the burn was contained within one sub-watershed whereas an adjacent sub-watershed remained largely unburned. The sub-watersheds are similar in size, aspect, and geology and enable a paired-watershed comparison of disturbance and recovery processes.

Watershed Description

Little Granite Creek, an upland contributor to the Snake River system, drains 21.1 mi^2 (54.6 km^2) of the Gros Ventre range near Bondurant, Wyoming, south of Jackson, Wyoming. The area is administered by the USFS, Bridger-Teton National Forest, Jackson Ranger District. Over half of the basin is in the Gros Ventre Wilderness Area. The two main tributaries of Little Granite Creek are Boulder Creek (8.0 mi^2/20.7 km^2)

and the upper basin of Little Granite Creek (7.6 mi^2/19.7 km^2). Approximately 80% of the forested area in Boulder Creek and less than 5% of that in upper Little Granite Creek burned in 2000 (Figure 1). Both basins face south and had similar pre-fire forest cover (Engelmann spruce, subalpine fir, and lodgepole pine). In the broader valley bottoms, riparian vegetation is dominated by willow species (*Salix* spp.), with an extensive herbaceous understory (Youngblood et al. 1985). In more confined portions, the floodplain overstory is dominated by conifers, with riparian shrubs occurring along the streambanks (Youngblood et al. 1985). Flow and rates of sediment transport are monitored at (1) the mouth of Boulder Creek (burned), (2) upper Little Granite Creek (unburned), and (3) Little Granite Creek above the confluence with Granite Creek (site of previous work) (Figure 1). Site 3 is approximately 2.5 river miles (4 km) downstream of the burned area. The elevations of the monitoring sites are around 6500 feet.

Runoff in the watershed is generated primarily by snowmelt and flows peak between mid-May and mid-June. While small to moderate thunderstorms are common in summer, they produce only minimal rises in the hydrograph. Mean annual temperature is 33.3°F (1.0°C) and mean annual precipitation is 20.53 inches (52.15 cm) at a nearby climate station in the vicinity of Bondurant, Wyoming (elevation 6,504 ft) (Western Regional Climate Center [WRCC]; National Climate Data Center [NCDC] Normals, 1961-90). Most of the precipitation falls as snow from November through March. Average annual snowfall measured at Bondurant between 1948 and 1999 is 134 inches (standard deviation ± 48.9 inches) (WRCC Monthly Total Snowfall).

The geology underlying the basin consists primarily of sedimentary formations (sandstone and claystone) of marine origin (Love and Christiansen 1985). Areas are prone to mass wasting from slow-moving earthflows, smaller-scale slumping, and soil creep. Upland resource uses include grazing and dispersed recreation (camping, hiking, horseback riding, and hunting). One USFS road parallels the stream for about 1.5 miles in the lower end of the watershed. Former land uses include coal extraction and a mining camp near the main stem above the confluence with Boulder Creek (Figure 1). These former and present uses are small in scale and their influence on sediment delivered to the channel is currently nominal.

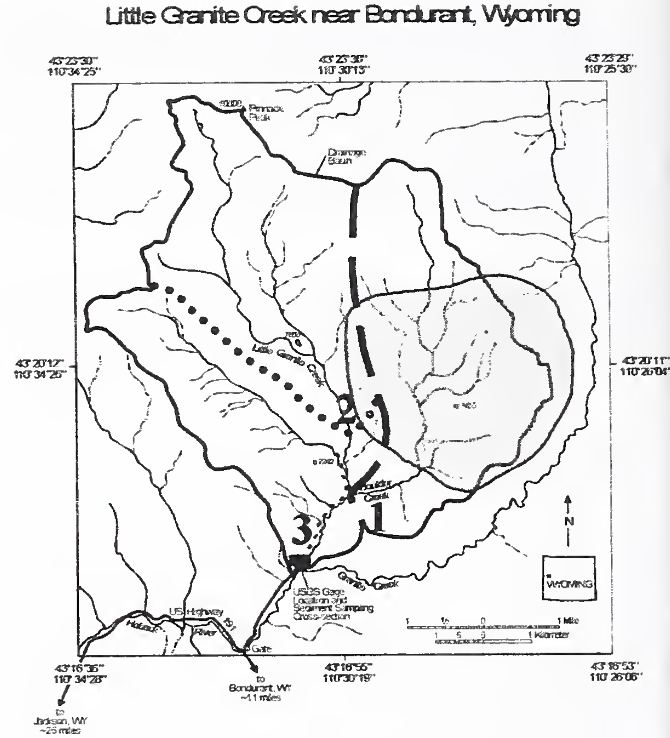

Figure 1. Map of Little Granite Creek watershed. Stipple pattern approximates the area burned by the Boulder Creek fire in 2000. An estimated 75% of the area burned with moderate to high intensity. The dark dashed line delineates the Boulder Creek watershed and the dotted line indicates the limits of the unburned watershed (upper Little Granite Creek). Numbers are described in text. The light dashed line represents an unimproved forest road.

The channel of Little Granite Creek ranges from step-pool in confined valley bottoms to pools and riffles in wider floodplains. Bed material ranges from gravel to small boulders, with less than 10% sand-sized grains located primarily at the channel margins and in small patches in the lee of larger particles. Banks are composed of sand, gravel, and cobbles overlain by fine sand and organic matter. The primary sources of sediment are scour from the channel bed and banks. External sources of sediment come from mass wasting, including active earthflows from unstable hillslopes, and slumping from undercut terraces and road cuts. Baseline concentrations of suspended sediment transport at high flows are relatively high (between 100 and 1000 mg L^{-1}) because of the underlying marine sedimentary formations. By comparison, similar measurements in streams in forested areas draining granitic terrain are rarely greater than about 100 mg L^{-1} (Andrews 1984, USDA Forest Service unpublished data).

Experimental Plan

Our basic approach in this watershed study is comparative, contrasting sedimentation processes, water quality, and riparian condition in burned and unburned drainages of the same watershed over time and against baseline data. The overall objectives of the Little Granite Creek study are listed below. However, we address only the first four objectives in this paper.

- Document changes in runoff patterns following wildfire.
- Compare pre-and post-fire rates of sediment flux at the historical site to determine the magnitude of increases in sedimentation following wildfire.
- Determine the relationships between temperature, snowmelt runoff, precipitation, stormflow response, and rates of suspended sediment transport in burned and unburned watersheds.
- Describe the fate and movement of large wood in the channels and floodplains in burned and unburned areas.
- Document changes in sediment storage, bed particle size, and embeddedness in burned watersheds.
- Quantify the transport and retention of organic matter (dissolved, fine particulate, and coarse particulate) in burned and unburned watersheds.
- Determine the relationships between sediment movement and post-fire recovery of riparian plant communities.

Monitoring methods

Flow stage is monitored at the three sites using automated stage recorders from April to November (or until the sites are no longer accessible). Frequent discharge measurements (>30 per season) are made at each site so that well-defined stage-discharge relationships are established (Nolan and Shields 2000). Precipitation and temperature are monitored at five locations in the watershed, both near the gaged sites and at higher elevations. There is also a SNOTEL (snow telemetry) site near the confluence between Little Granite and Granite creeks from which longer-term trends in temperature and precipitation data may be extrapolated.

Samples of transported bedload were collected at the three sites using a thick-walled Helley-Smith bedload sampler with a 3 x 3" (0.076 x 0.076 m) opening (Helley and Smith 1971). Typically, measurements were taken once per day at each site during the snowmelt runoff season. Samples of stream water were collected at each site using an ISCO™ automated sampler. The samplers were programmed to collect water once every 4 hours between May and October in the first 2 years post-fire, producing a very detailed record of sedimentation patterns from over 2000 samples per site. Suspended sediment concentration [SSC], fine particulate organic matter [FPOM], and a "sand-silt split" (at 63-μm) are obtained from these samples. Additional water grab samples were collected for analysis of dissolved organic matter [DOM] (USEPA 1987). Samples of coarse particulate organic matter [CPOM] were acquired for a range of flows from drift nets.

In an effort to improve and simplify methods for monitoring SSC, we deployed an Optical Backscatter Sensor [OBS-5] at the lower site in 2002 to measure turbidity for comparison with our measured concentrations. Turbidity is also used as a qualitative indicator of water quality and light penetration for aquatic studies.

Several stream reaches were selected for long-term monitoring of changes in sediment storage. Fourteen reaches (10 in Boulder Creek and 4 in upper Little Granite Creek), each 100 m (330 ft) in length, were monumented and surveyed in 2001-02. Within these reaches, equally spaced channel cross-sections and the longitudinal profile were surveyed using a total station surveying instrument. Pebble counts (Wolman 1954) using 200 grains were used to characterize the grain size distribution of the channel surface. Embeddedness, or the extent to which bed particles are covered by silt and sand, was estimated using a "view bucket" whereby the percentage of particles covered with fines is determined using a gridded template (Wayne Minshall, Professor of Ecology, Idaho State University, personal communication). The redistribution and recruitment of

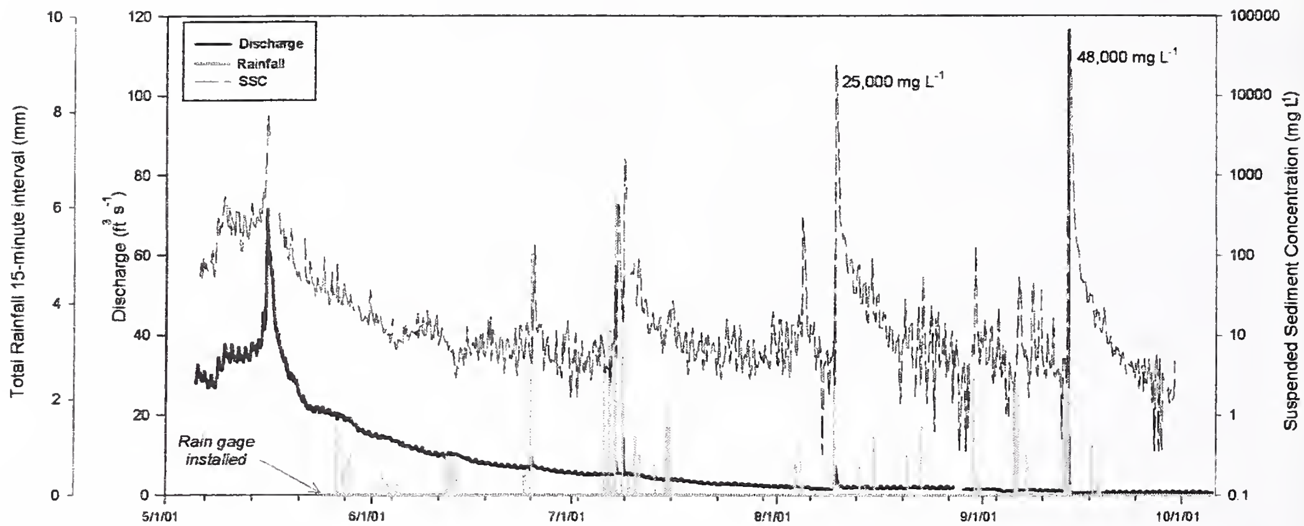

Figure 2. Boulder Creek rainfall intensity (at high elevation station), discharge, and SSC measured in 2001. Note that SSC is on a log scale, so differences in low concentrations appear exaggerated while higher concentrations appear muted.

large wood (tree trunks) within the channel and on the floodplains is monitored through repeat surveys. About 600 pieces of large wood were tagged, surveyed, and classified for stability during 2001 and re-surveyed in 2002. The stream survey work is repeated annually to monitor gross changes in channel morphology and movement of large wood over time. Vegetation (overstory, shrubs, seedlings, and vascular plants) growth and re-growth in burned riparian areas are monitored using permanent transects and quadrats within several of the reaches.

Results

The results described here are primarily from 2001 because, as of this writing, analyses of 2002 data are largely in progress. Similar to areas throughout the western United States, flows in Little Granite Creek were relatively low in water year 2001. Discharge on the main stem exceeded bankfull for only a few hours during a prolonged rainstorm during snowmelt runoff in May. Measured rates of suspended sediment transport were about 5x higher on the rising limb of the snowmelt hydrograph relative to pre-burn values, while there were no substantial differences observed for the falling limb. Rates of bedload transport during snowmelt runoff were similar to those measured before the fire.

During mid-summer, there were a few low to moderate-intensity thunderstorms that raised the Boulder Creek hydrograph by a small amount (Figure 2), similar to runoff patterns prior to burning. However, there were often large increases in suspended sediment associated with these storms. Rainfall rates of 1-6 mm in 15 minutes produced increases in SSC between 1-2 orders of magnitude (Figure 2). Concentrations returned to baseline values with a few hours or days during these smaller "blackwater" events. By contrast, no measurable increase in SSC was observed in the unburned watershed during these periods.

Two larger rainstorms in late summer produced the highest SSC measured at Little Granite to date. A brief (<15 minutes) but intense rainfall (~2 inches hr^{-1} or 50 mm hr^{-1}) on August 9 generated a few fine-grained mudflows from gullies within the burned area (Figure 3). Suspended sediment concentration was about 3 orders of magnitude greater than baseline values and it took over a week for these values to return to baseline (Figure 2). By contrast, SSC increased by 1 order of magnitude in the control watershed, returning to baseline within 8 hours. A second series of storms on September 13 produced a substantial spike in flow and SSC in streams below the burned areas (note: bedload was not measured during either storm). Flows on Boulder Creek exceeded bankfull and were greater than those measured during snowmelt runoff (Figure 2). Suspended sediment concentration peaked at about 48,000 mg L^{-1} which is about 4 orders of magnitude greater than baseline values whereas SSC in the control watershed was 1,300 mg L^{-1}, or about the same peak

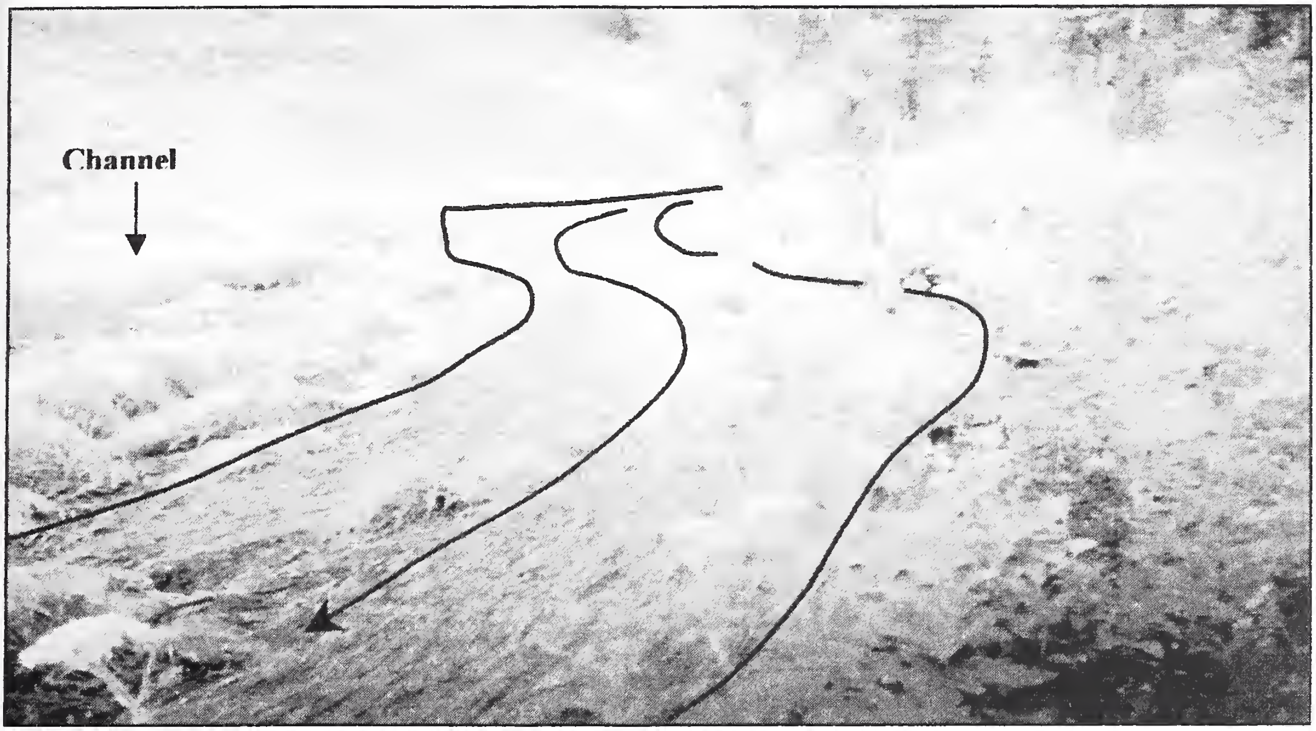

Figure 3. Photograph of the track of a fine-grained mudflow emanating from a gully in an area burned by the Boulder Creek wildfire (taken in August 2001). The channel upstream of this location plugged with debris causing the mudflow to move on to the floodplain and rejoin the channel downstream of this location. Similar tracks were observed after rainstorms on August 9 and September 13, 2001.

concentration observed during snowmelt runoff. The event in Boulder Creek was witnessed by several hunters camping nearby who later reported thunderous noises, numerous "gully washers," and rapid increases in flow. During reconnaissance, we noted areas with shallow (~6 inches or 0.15 m) soil slips, scoured gullies, fresh mudflow tracks, and out-of-bank flow marks. Additionally, several wood jams that had been tagged for monitoring earlier in the summer had been dismantled and large tree trunks could be traced up to 0.5 km (0.3 miles) from the site where they were originally surveyed. Channel cross-sections within the burned area showed localized zones of deposition and scour which were frequently associated with the rearrangement of large wood (Figure 4).

With each storm and subsequent blackwater event, the bed of Boulder Creek and Little Granite Creek below the burned area became increasingly infused with fines, consisting of a mixture of burned organic matter and inorganic materials. Measurements taken in 2002 indicated that the degree of embeddedness in Boulder Creek ($92\% \pm 4\%, \overline{x} \pm 1\,\text{s.e}$) was substantially greater than that observed in the control watershed ($59\% \pm 2\%$).

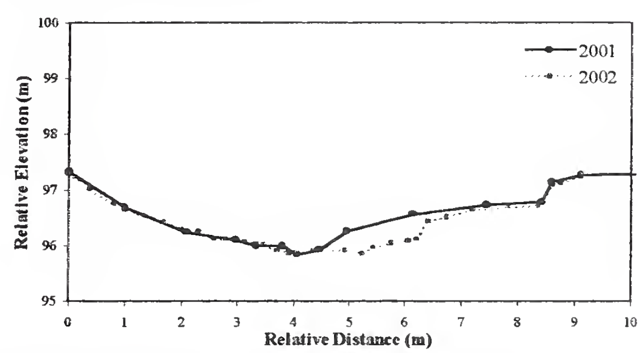

Figure 4. Example of change in cross-section observed between summer of 2001 and 2002. This particular cross-section is located downstream from a wood jam and the scour likely occurred during the rearrangement of wood during the September 13 storm. Vertical exaggeration is approximately 1:1.

Future Work

We plan to continue monitoring runoff, stream sedimentation, and morphologic changes at Little Granite Creek, in addition to following the re-growth of vegetation and recovery of riparian areas over time. Such results are useful to forest managers from several disciplines for addressing critical issues concerning resource fragility and system recovery following wildfire. Hydrologists and vegetation managers need information on hydrologic changes following wildfire and their expected impacts in different parts of the watershed in order to define areas where rehabilitation efforts may be most effective. Downstream users (municipal and agricultural) need information on local and off-site impacts of wildfire on stream water quality and quantity. Ecologists need to identify areas most prone to heavy sedimentation or scour and how such changes may affect aquatic communities. Over longer time frames, managers must consider the overall fragility of the watershed in order to prescribe adequate protection for burned areas. Through integrated monitoring of recovery in streams, such as that demonstrated by the long-term project in Little Granite Creek, we can begin to address these concerns and project how far into the future the effects of wildfire may be sustained by an aquatic system.

Acknowledgments

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Quick Response Small Catchment Monitoring Techniques for Comparing Postfire Rehabilitation Treatment Effectiveness

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Abstract

Increased runoff and erosion commonly occur after wildfires with the onset of precipitation events. Various erosion mitigation treatments are often used after wildfires to reduce flooding and sedimentation. The effectiveness of these treatments has not been well documented in the literature; therefore we undertook a rapid response approach (within four weeks following fire suppression) to install small catchment monitoring systems to compare treatment effectiveness. A paired watershed approach uses two adjacent and similar catchments (5-20 ac) after wildfires, treating one catchment and using the other catchment as a control. We developed a rapid response monitoring system that can be installed in a few weeks to monitor sediment yield and runoff response. These systems are usually left in place for three to five years.

Each installation has a complete weather station and electronic measuring devices to record streamflow and sediment accumulation in a storage basin. The sediment basins are cleaned out manually after each storm event in order to relate the event (intensity, amount and duration) to runoff and sediment yield. The data is automatically transmitted each day via cell phone or radio transmission to our computer server, thus making the data available daily on our web page. We have installed six paired catchments to date in Colorado, Washington, two in California, and two in Montana. Preliminary results suggest that 1) first year storm events produce the largest runoff and sediment response and 2) treatment effectiveness is less with high intensity short duration storm events. This rapid response protocol allows for quick installation of a monitoring system to provide an assessment of treatment effectiveness.

Keywords: erosion, sedimentation, paired watersheds, instrumentation, data logger

Introduction

Soil erosion after wildfires in forest and range environments is often a land management issue due to effects of sediment on water quality and downstream values at risk. Rainfall-induced soil erosion commonly occurs after wildfires. Various methods have been used to estimate sediment yields after wildfire; however, few studies have actually measured postfire erosion (Robichaud et al. 2000). Additionally, postfire mitigation treatments used to reduce runoff and erosion have not been rigorously evaluated to determine if they are meeting treatment objectives (Robichaud et al. 2000, GAO 2003).

Therefore, monitoring postfire treatment effectiveness has been a research focus in recent years. Small catchments (4-20 ac) have been identified as an appropriate size to be able to evaluate the effectiveness of these treatments. These catchments are large enough to generate sufficient runoff and sediment, yet easily measurable. At the same time they are not large enough to include hydrological process such as channel storage and redistribution, which would mask treatment effectiveness.

This paper describes how to install a rapid response measuring system with a paired watershed approach. Study design, installation procedures, equipment and monitoring methods are discussed.

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Methods

The study sites, identified during the postfire assessment, usually have high runoff/erosion potential and are reasonably accessible with pickup trucks or ATVs. Generally, treatments are installed before the first storm event following the fire. Also the sites are protected from other disturbances, such as salvage logging or grazing, for the duration of the monitoring period (3 to 5 years).

Paired watershed approach is used to compare a postfire treated to a non-treated control catchment (MacDonald et al. 1991). Site selection is usually made after the fire is controlled by reviewing topography maps, burn severity maps, discussions with local officials, and aerial reconnaissance. Ideally the catchments will have similar topographic features, size, soils, pre-treatment vegetation characteristics, burn severity and are located adjacent to each other. Treatments have included contour-felled logs or log erosion barriers (LEBs), straw wattles, raking, or mulching. By locating the catchments adjacent to each other, isolated thunderstorms will affect both catchments with similar rainfall characteristics.

Each catchment, treated or untreated, contains a sediment basin/storage area and flume/weir. The measurement apparatus for sediment and runoff is robust enough to handle big events (the design storm plus extra capacity), yet sensitive enough to reliably detect small events. Because it is critical to know when storm events occur relative to water and sediment movement, measurement equipment operates continuously.

Sediment traps

A sheet metal cut-off wall is used to divert all runoff and sediment into a sediment trap area. Sheet metal (4 by 12 ft-16 gauge) is used and buried 8-10 in. beneath the ground surface in the base of the channel and extending up the side slopes (Figure 1). The center sheet (4 by 6 ft) has two wing walls bent at 45 degrees, 1 ft in on each side. Additional metal sheets are added to the top with similar bends to provide for additional storage. Straight sheets are used for each side. The sides extend 5 to 25 ft from the 45-degree bend to define the storage basin.

Wooden posts (4 by 6 in and 4 by 4 in) are used to support the sheet metal. The sheet metal is attached to the wood posts (Figure 2). Lateral supports (4 in by 4 in wooden posts or 2-3/8 in diameter round metal pipe-chain link fence posts) and dead man anchors are used to stabilize the cut-off wall. After the sheet metal and post are in place, concrete is used as a footing along the bottom edge and around each post. Generally about 2 yd^3 of concrete is used per installation. Additionally, silicon caulk is used for all sheet metal overlaps and joints to make the sediment basin watertight.

Figure 1. Sheet metal cut-off wall installation.

Figure 2. Sidewalls of sediment trap.

Figure 3. V-notch weir and trash/debris screen.

A 90-degree V-notch (15 in deep) weir is located from the top center of the cutoff wall as a weir for measuring runoff (Figure 3). A trash/debris screen (chain link fencing) is located about 3 ft upstream of the V-notch. A splash apron is located just downstream of the notch, made with concrete, rocks, logs or half- round 12 in diameter culvert pipe.

Instrumentation

A complete weather station (Campbell Scientific Inc., Logan, UT) is used to measure wind speed, wind direction, solar radiation, humidity, peiziometer measures ground water level, flume/weir level, rainfall intensity (tipping bucket rain gauge), soil moisture and sediment/snow depth in sediment basin.

Depth of runoff flow is measured with a magnetic float along a stainless steel rod (magnetic linear actuator) placed inside a 4 in diameter slotted PVC pipe mounted along the cutoff wall. An ultrasonic sensor is mounted above the floor of the sediment basin to measure the height of the water and sediment in the sediment trap (Figure 3). During winter months, the ultrasonic sensor provides the depth of snow.

All instrumentation is connected to a CR10 data logger (Campbell Scientific Inc., Logan UT) that can store the data, as well as transmit the data via cell phone or radio transmission. The data are downloaded daily from the data logger and uploaded to our web server (http:forest.moscowfsl.wsu.edu/engr/weather). All electronics are powered by 12-volt battery that is charged daily by 32-Watt solar panel with a voltage regulator.

Other measurements

Soil descriptions including physical characteristics of soil horizons, texture, structure, bulk density, and conductivity are determined using standard methods to ensure that the control is similar to the treated catchment. Soil wetability is measured at stratified and randomly selected locations after the fire to characterize the extent and degree of water repellent soils.

Eight to ten channel cross-sectional measurements for assessing sediment loss due to channel scour are completed at the beginning of the study, should the primary measurement systems fail or become overwhelmed. Channel cross-sections are measured at 100 ft intervals or at major slope change above the sediment trap.

Annual measurements of ground cover (e.g., plants, litter, mineral soil, rock) are made following Chambers and Brown (1983).

Since treatments may vary, monitoring/evaluating treatments also vary. For example, contour felled logs are surveyed for size, number, orientation, position and degree of functionality. After each runoff event, the contour-felled logs are evaluated to determine the accumulation of sediment, failure rate and mechanisms. Mulch treatments are monitored/evaluated by ground cover measurements and vegetation response.

Sediment basin cleanout methods

Periodic clean out of the sediment traps is required to obtain reliable measurements of sediment. Cleaning the sediment traps following every storm improves the prediction accuracy of the storms that produce the sediment. Sufficient time is allowed for most sediment to settle before water is drained.

Hand labor is generally used to collect and weigh the small amounts of sediment (under 2000 lbs) (Figure 4). Large amounts of sediment require small equipment (mini-excavator, Bobcat or wheel backhoe) to clean sediment traps. When equipment is used, volumes are first estimated and bulk densities determined, this allows for converting volumes to mass calculations.

Figure 4. Buckets are used for sediment clean out.

The direct measurement of the total weight of the sediment is the most accurate technique for estimating sediment amounts (Robichaud and Brown 2002). The sediment is weighed and recorded in the field using a plastic bucket (5 gal). Place the sediment into the container and weigh in the field with a hanging or platform parcel scale (scale with 0.5 or 1 lb) increments with a maximum capacity of

80-100 lb. Weigh each bucket and place a representative subsample (0.1 lb) into a soil tin or recloseable plastic bag for water content determination in the laboratory. The remaining material can be discarded downhill of the sediment trap.

Data processing and analysis

Telemeter data are reviewed and summarized after downloading. Periodic field collected data are reviewed and summarized. Comparisons are made between the treated and untreated catchments on a storm-by-storm and cumulative basis.

Runoff, expressed as area-depth, rate, or volume flow, is calculated for each storm event. Sediment delivery results can be normalized based on catchment's size (weight/area) to compare treated verse control catchments or based on rainfall intensity and amounts. Additionally, various methods are used to compare treatment efficiency.

Results

We have installed six paired catchments after wildfires in Colorado, Montana, Washington, and California. The first sites were installed in 1998 and additional installations have occurred each successive year. Preliminary results suggest that the postfire emergency rehabilitation treatments are effective with low intensity, long duration rainfall events and less effective with high intensity, short duration rainfall events (Table 1).

Table 1. Annual sediment yields from selected research sites.

Site	Year	Untreated (T/ac)	Treated (T/ac)
North 25-WA	1999	0.3	0.5
	2000	No change	No change
	2001	No change	No change
Mixing-CA	2000	0.01	0.2
	2001	0.8	0.04
	2002	0.8	0.06
Bitterroot-MT	2001	0.2	0.07
	2002	0.4	0.2
West Pine-MT	2002	4.5	4.6

The North 25 and West Pine sites had short duration, high intensity rain events occur during the first year after the wildfire, thus annual sediment yields were similar between the treated and untreated catchments.

Summary

A rapid response catchment installation procedure has been developed and utilized at six locations in the Western States to compare postfire rehabilitation treatment effectiveness. The quick response system can be installed in a few weeks and provides accurate data to be able to make comparison between treated and untreated catchments. These installations can be removed when the studies are complete usually 3 to 5 years after installation. Preliminary analysis suggests that these methods provide reliable results to be able to compare treatment effectiveness. Additionally, these data are at an appropriate scale to valid our erosion prediction models.

Acknowledgments

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Impacts of Wildfires on Hydrologic Processes in Forest Ecosystems: Two Case Studies

Gerald J. Gottfried, Daniel G. Neary, Malchus B. Baker, Jr., Peter F. Ffolliott

Abstract

The Coon Creek Wildfire in 2000 and the Rodeo-Chediski Wildfire in 2002 devastated large areas of forest in north-central Arizona. The presence of historic hydrologic control structures on the three Workman Creek Watersheds within the Coon Creek burn area and the two Stermer Ridge Watersheds within the Rodeo-Chediski area provide opportunities to study the impacts of watershed-scale fire on hydrologic processes in the forest ecosystems of the Southwest. Fire intensities and severities varied among the watersheds within each area. Streamflow regimes, erosion-sedimentation processes and other on-site hydrologic characteristics are being monitored. High intensity summer thunderstorms following the fire at Workman Creek produced three peak flows that exceeded the historic peak flow. One storm in June 2000 produced a peak of approximately 2,000 cubic feet per second, 7 times the previous record peak flow. The severely burned watershed at Stermer Ridge produced a peak flow of 232 cubic feet per second, an increase of 2,350 times the historic high flow, after an intense summer storm. Hydrophobic soils were identified within the severely burned areas. Preliminary measurements at the Main Dam sediment basin at Workman Creek and at erosion pin sites at Stermer Ridge indicate increases in erosion and sedimentation from the severely burned watersheds. Continued monitoring of the watersheds will provide additional information about the impacts of wildfires on hydrologic processes in southwestern forest ecosystems.

Keywords: hydrologic processes, erosion and sedimentation, wildfires, forested ecosystems, watershed studies, Arizona

Introduction

Two historical wildfires in north-central Arizona in 2000 and 2002 provided opportunities to study the impacts of watershed-scale fire on hydrologic processes in the forest ecosystems of the Southwest. Streamflow regimes, erosion-sedimentation processes, precipitation, and other on-site hydrologic characteristics are being monitored and evaluated on the Workman Creek and the Stermer Ridge Watersheds following recent large and devastating wildfires. The Coon Creek Fire of 2000 burned 9,644 acres on the Tonto National Forest, including the three experimental watersheds at Workman Creek on the Sierra Ancha Experimental Forest, while the Rodeo-Chediski Fire in 2002 burned approximately 475,000 acres on the White Mountain Apache Reservation and Apache-Sitgreaves National Forests. The Stermer Ridge Watersheds are located within the burned area on the Apache-Sitgreaves National Forests, south of the towns of Heber and Overgaard. The hydrologic control structures at both locations had been mothballed after previous watershed research studies were completed, but had been left in place. The control-sections were re-instrumented and

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weather stations re-established immediately following the respective wildfire to provide a basis to study the relative impacts of light, moderate, and severe fire severities on selected hydrologic processes. Preliminary findings with respect to peak flows, annual streamflow regimes, on-site changes in soil properties, and sediment transport are reported; however, much of the data still need to be analyzed. Vegetation also is being monitored but these data are not reported here. Information of this kind is needed by land managers to plan and manage for on-site post-fire watershed rehabilitation and to understand the impacts of fire on downstream riparian corridors and anthropological infrastructures.

Coon Creek Fire

The Coon Creek Fire originated on April 26, 2000 at an unattended campfire in the lower reaches of Coon Creek on the eastern side of the Sierra Ancha Mountains. The fire eventually burned approximately 9,644 acres including parts of the Workman Creek Watersheds, within the Sierra Ancha Experimental Forest, and the Sierra Ancha Wilderness. The burned area originally supported a vegetative cover of mixed ponderosa pine (*Pinus ponderosa* var. *scopulorum*) and Gambel oak (*Quercus gambelii*), ponderosa pine, and mixed conifer forests. The fire crossed the three experimental watersheds (South Fork, Middle Fork, and North Fork) at the headwaters of Workman Creek (Gottfried and Neary 2001, Gottfried et al. 2002). These watersheds, which cover a total of 1,087 acres, were established in 1939 to investigate the hydrology of mixed conifer forests and the impacts of forest management treatments on watershed hydrology. The area is between 6,590 and 7,724 feet in elevation and received an average of about 36.2 inches of annual precipitation between 1960 and 1991 (Martinez 1993). Two-thirds of the annual precipitation occurs during the October through May period, mostly as snow, and produces most of the annual streamflow.

The control sections on the three watersheds were "mothballed" in 1983 following more than 40 years of continuous hydrologic monitoring and evaluations. The Middle Fork of Workman Creek, which had been the hydrologic control for the earlier experiments, supported an undisturbed old-growth mixed conifer forest of ponderosa pine, Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), and white fir (*Abies concolor*) prior to the fire. The Middle Fork stand averaged more than 200 square feet of basal area per acre and fuel loadings averaged from 40 to 50 tons per acre (USDA Forest Service 2000) but might have been heavier in some areas. The forests on South Fork and North Fork had been modified by the earlier experimental treatments and contained mosaics of grass, shrub, and forest covers at the time of the fire (Gottfried and Neary 2001, Gottfried et al. 2002). While most of the burn severities were low, approximately 20 percent of the area, including Middle Fork, was burned at high severities. The vegetation and soil surface on two-thirds of this watershed were subjected to high soil heating where litter, duff, and logs were completely consumed exposing the surface soil, which was reddish in color. Burn severities on the other two watersheds were generally low-to-moderate.

Streamflow measurements

The Rocky Mountain Research Station and Tonto National Forest opened the weirs and a flume at Workman Creek in June 2000 to assess the impacts of the Coon Creek Fire on streamflow volumes, peak flows, and soil erosion and sedimentation rates. Main Dam, a combination 90° V-notch weir and 7-foot Cipolletti weir, measures stream flows from the entire three-watershed area. The South Fork and North Fork watersheds are gauged by 90° V-notch weirs, and streamflows from the 411-acre main part of Middle Fork are measured at a trapezoidal flume. A weather station was re-established at Peterson Meadow within Middle Fork. A Natural Resources Conservation Service snow course and/or SNOTEL station has been maintained continuously on Middle Fork since 1951. Statistically significant relationships have been developed for the pre-fire period between peak snow water equivalent data and winter snowmelt runoff and mean daily peak flows for all hydrologic control sections and should be useful in analyzing the impacts of the wildfire on snowmelt runoff parameters (Gottfried et al. 2002).

Peak flows

Several record peak flows have been estimated at the Main Dam site since the wildfire. A 15-minute rainfall at an intensity of 2.6 inches per hour on Middle Fork in June 2000 produced a peak flow that was more than 7 times that of the historic high peak of 289 cubic feet per second that was measured on October 7, 1972 (Neary and Gottfried 2002). The earlier peak flow was related to approximately 11.67 inches of precipitation that was recorded in Middle Fork from October 3 through October 7, 1972. The

June 2000 streamflow overtopped the weir and, therefore, was estimated from high water marks.

Two other peak flow events were also observed in August 2001. The higher peak flow, of about 420 cubic feet per second, was recorded on August 11, 2001, when a thunderstorm produced a rainfall event of approximately 1.3 inches per hour in intensity. The partially cleaned settling basin and associated hydrologic structures at Main Dam were filled with sediment after this event. Observations at South Fork and North Fork showed slight increases in trapped sediments behind the weir walls, indicating that most of the streamflow and sediments originated from the severely burned Middle Fork. The Main Dam was overtopped by both events and the instrument shelters were partially submerged in the second event. These two flows contained large amounts of sediment and several logs, making accurate streamflow calculations difficult.

On-site changes in soil properties

The Burned Area Emergency Rehabilitation (BAER) assessment indicated that 330 acres or 29% of the burned area in Workman Creek would have a high to very high response to precipitation events because of moderate to strong water repellency and minimal vegetative ground cover (USDA Forest Service 2000). The report estimated a 50% reduction in infiltration and that the soil erosion hazards on sites like Middle Fork are high.

Sediment transport

Severe surface soil erosion and sub-channel scouring were observed on Middle Fork following the June 2000 storm but no measurements were made because the control sections had not been instrumented at the time. Currently, erosion and sedimentation information are being measured on a series of channel cross sections that have been established on Middle Fork and on South Fork above the weirs. Sediment also is being measured at the settling basins behind the weirs at Main Dam, South Fork, and North Fork. Measurements are made on series of cross-sections that extend at 1-meter intervals parallel to the cutoff walls. Most of these data have not been fully analyzed; however, the August 2001 storms produced more than 1,456 cubic feet of sediment in the Main Dam basin, which had been cleaned less than two month earlier, compared to 526 cubic feet measured between late June 2000 and December 2000 when no unusual streamflow events occurred. Most of the sediment originated from Middle Fork although some should be attributed to a main forest system road that runs parallel to the channel below the confluence of the three forks and crosses Middle Fork at one point. A pebble count of sediment behind the Main Dam in June 2001 indicated that 48% of the sediments were sand (< 0.08 inch in diameter) and 36% were cobbles (2.5-10.1 inches). Approximately 3% of the sediments were in the boulder category (>10.1 inches).

Rodeo-Chediski Fire

This historic fire actually consisted of two fires that ignited on the Fort Apache Reservation and then merged into one devastating fire. The cause of the Rodeo Fire, which began a few miles from Cibecue on the Reservation on June 18, 2002, was arson. A seemingly lost person set the Chediski Fire as a signal fire a few days later. This second fire spread out of control and eventually merged with the on going and still out of control Rodeo fire. Burning northeastwardly, the re-named Rodeo-Chediski Fire then moved onto the Apache-Sitgreaves National Forest, along the Mogollon Rim in central Arizona, and into many of the White Mountain communities scattered along the Mogollon Rim between Heber and Show Low. Over 30,000 local people were forced to flee the inferno. The fire had burned 276,507 acres of Apache land and 462,606 acres in total by the time that most of the firefighters had left the area on or about July 13. Nearly 500 buildings had been destroyed, with over one-half of the burned structured being the houses of local residents or second-homes of summer visitors.

Streamflow measurements

Two nearly homogeneous, 60-acre watersheds had been established along Stermer Ridge at the headwaters of the Little Colorado River in 1972-73 as a cooperative project of the Rocky Mountain Research Station, USDA Forest Service, and the University of Arizona to obtain baseline hydrologic and ecological information on watersheds located in ponderosa pine forests on sedimentary soils (Ffolliott and Baker 1977). Cretaceous undivided material with mineralogy similar to that of the Coconino sandstone formation lies beneath the watersheds. The soils have been classified as Typic Eutroboralfs with sandy loam textures (Laing et al. 1987). The two watersheds were situated on relatively flat topography, with few slopes exceeding 10 percent. Their elevations range from 6,800 to 7,000 feet. The most recent timber harvest removed approximately 45 percent of the merchantable sawtimber by group selection in the early 1960s. Sixty-five percent of the 20-to-25 inches of annual precipitation falls from October to April, much of it as snow, and the remainder in rainstorms from July to early September. Summer storms, while often intense, rarely produced significant stormflows before the watersheds were burned.

The two watersheds had been "moth-balled" in 1977-78 after completion of the baseline watershed measurements. However, the control sections (3-foot H-flumes) were left in place. Following cessation of the Rodeo-Chediski Fire, these control sections were re-furbished and re-instrumented with water-level recorders and a weather station was re-established on the site to study the impacts of varying fire severities on hydrologic processes. A fire severity classification system that relates fire severity to the soil-resource response to burning (Hungerford 1996) was used to determine the relative portions of the watersheds that were burned at low, moderate, and high severity (Wells et al. 1979). This extrapolation indicated that one of the Stermer Ridge watersheds experienced a high severity stand-replacing fire, while the other watershed had been exposed a low-to-medium severity stand-modifying fire.

Post-fire peak flows

Summer storm flows on the Stermer Ridge watersheds had been uncommon. The highest peak flow measured in a summer storm flow event in the 1972-76 pre-fire period was about 0.10 cubic feet per second. However, high-water marks observed at the control sections in the first visit of researchers to the watersheds following the fire indicated peak flows in orders of magnitude larger than earlier recorded (Ffolliott and Neary 2003). The estimated peak flow on the watershed that experienced the high severity stand-replacing fire was almost 8.9 cubic feet per second or nearly 90 times that measured in 1972-1976 period. Peak flow on the watershed subjected to the low-to-medium stand-modifying fire was about one-half less in magnitude but still far in excess of the previous observations. A subsequent rainfall event of unknown intensity generated even higher peak flows. Precipitation records are not available for Stermer Ridge for these early storm periods. On the severely burned watershed, the peak flow following this second event was estimated to be 232 cubic feet per second or about 2,350 times that measured earlier. This flow represents the highest known post-fire peak flow measured in the ponderosa pine forest ecosystems of Arizona or, more generally, elsewhere in the southwestern United States.

On-site changes in soil properties

The initial effect of the Rodeo-Chediski Fire on soil properties has been manifested largely through changes in infiltration rates and transpiration. The reduction in infiltration because of the widespread formation of water-repellent soil following the fire decreased the amount of water entering the soil as a result of the monsoon storm events in the two-months following the fire. However, at the same time, the extensive loss of vegetation to burning meant that less water was removed from the soil body by the transpiration process. The net effect of these "compensating changes" is expressed by the spatial and temporal measurements of soil moisture storage that began shortly after the fire was controlled.

The fire as a result of the extensive loss of protective vegetative cover exposing mineral soil intensified erosion forces on the watersheds. Soil erodibility also increased because of the volatilization of soil organic material and a destruction of aggregated soil structures. The reduction in infiltration rates on water-repellent soils and consequent increase in overland flows increased the dislodgement and transport of soil particles as witnessed by the large accumulations of sediment in the stream channels.

Sediment transport

Sediment transport immediately following the fire can only be approximated because erosion pins had not been established. The BAER assessment estimated that the average post-fire sediment delivery for the total burned area would be 68 cubic feet per acre per year or an increase of more than 3,200 percent over pre-fire conditions (USDA Forest Service 2002). Based on observations of post-fire soil pedestals and measurements of sediment accumulations in the stream channel, it is estimated that a sediment load in excess of 30-to-35 tons per acre passed through the control section on the watershed that experienced the high intensity stand-replacing burn in response to the summer monsoonal storms. It is likely that sediments in excess of 15-to-20 tons per acre were generated on the watershed subjected to the low-to-medium stand modifying fire. These values exceed the previous estimates for maximum rate of soil erosion for the watershed areas of 3 to 5 tons per acre per year (Laing et al. 1987).

Erosion pins were installed on the watersheds in October to estimate the changing magnitudes of subsequent soil erosion rates and sediment loads in terms of the respective fire intensities observed. Measurements made in April 2003 indicated that the intervening erosion was about 23 tons per acre on the watershed that burned at a high severity and nearly 12 tons per acre on the watershed that burned at the low-to-medium severity. The relative amounts of this eroded soil that moved into the stream channels is unknown at this time.

Hydrologic impacts of historical Arizona forest fires

Other wildfires impacting peak flows have been measured in the forests of Arizona (Neary et al. 2003). Rich (1962) reported on the impacts of a 60-acre wildfire within the upper section of the South Fork of Workman Creek in July 1957. He measured a peak of 78 cubic feet per second after a high intensity summer storm that had produced between 3.50 and 4.05 inches of precipitation with intensities exceeding 2.00 inches per hour. The storm removed more than 41,800 cubic feet of soil from the 60-acre site, most of which was re-deposited on flat areas below the burned area or in the stream channel. The Rattle Burn in May 1972 covered two of three watersheds in ponderosa pine forests southwest of Flagstaff (Campbell et al. 1977). One watershed was severely burned, one was moderately burned, and one was not burned. Average annual runoff for the first three years after the fire was 1.1 inches for the severely burned watershed, 0.8 inch for the moderate area and 0.2 inch for the unburned area. Peaks averaged 10.98 cubic feet per second for the severely burned area for two large post-fire precipitation events while the moderate and unburned areas had similar responses, averaging 0.34 and 0.22 cubic feet per second.

Some post-fire impacts have been difficult to analyze on a watershed-basis because of instrumentation or design limitations. One example is the Dude Fire of 1990, one of the more devastating wildfire in Arizona's history, which burned 27,120 acres of mostly ponderosa pine forests surrounding Dude, Bonita, and Ellison Creeks in central Arizona, destroyed over 60 homes and structures, and, sadly, resulted in the loss of six lives. Because streamflow measurements are not available for the individual watersheds impacted by the Dude Fire, it is not possible to directly assess the fire's impacts on the peak flows from these burned watersheds. However, streamflow records from the U.S. Geological Survey gauging station situated on Tonto Creek above Roosevelt Lake, which monitors a 675 square mile watershed including the Dude Fire, can be used to "qualitatively consider" the Dude Fire in the general context of its effects within a broader and larger river-basin scale (Neary et al. In press). Streamflow measurements at this gauging station for the year of the fire (1990) were compared to mean daily streamflow records for two years preceding (1988-89) and two years following (1991-92) the fire. The results indicated significantly higher peak flows in the initial post-fire period than in the other periods following summer monsoon rainfall events of similar magnitudes and time periods. The initial storm flow following the fire had a peak that was 80 times the base flow at the time. While the proportion of this increase that is attributable to the fire is unknown, there can be little doubt that the fire played a significant hydrologic role in generating this increase.

Summary

The post-fire impacts on hydrologic processes reported in this paper have been documented more-or-less on a controlled watershed-basis; that is, either pre- or post-fire measurements were made on a watershed or an unburned control watershed that were available for comparison purposes. While specific case studies are presented, the increases in peak flows, streamflow regimes, and erosion-sedimentation rates that have been observed are likely to bracket the post-fire peak flows that might be expected in Arizona montane forests. It is further assumed that the post-fire peak flows observed immediately following the Rodeo-Chediski Fire represent the "upper-end" of the range of these values. The results and observations reported here are preliminary; research is continuing on the Workman Creek and Stermer Ridge watersheds and more comprehensive information will be available in the next few years.

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Integrated Management

Managing Interdisciplinary Research: Lessons Learned from the EPA-STAR /NSF/USDA Water and Watersheds Research Program

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Abstract

As environmental problems became more complex, interdisciplinary research teams were recognized as key elements needed to solve environmental problems. With the recognition that interdisciplinary research was essential to laying the scientific foundation for solving complex watershed problems the U.S. EPA's *Science to Achieve Results (STAR)* program in partnership with the NSF initiated the Water and Watersheds research program in 1995. In 1998, the USDA joined the partnership. A major goal of the program was to promote integration across the biological, physical, and social sciences in the area of watershed management. In October 2000, the EPA sponsored a workshop comprised of 10 of the Principal Investigators (PIs) to investigate the issues of managing interdisciplinary research teams. The output of the workshop was a *Lessons Learned* document that reviews the scientific, personnel, administrative issues of managing complex grants.

Keywords: watershed, research management, interdisciplinary

Introduction

Over the last few decades, approaches to environmental management have significantly changed. Traditional approaches to watershed research have focused on individual scientists testing hypotheses within single disciplines (Popper 1959). As environmental problems have become more complex, interdisciplinary research teams, including engineers and social scientists, have become the preferred approach to solving problems at multiple scales (Odum and Pigeon 1970; Reichle 1970).

In recognition of this need for broader interdisciplinary studies to address larger-scale environmental issues, the US Environmental Protection Agency (EPA) *Science To Achieve Results* (STAR) program, in partnership with the National Science Foundation (NSF), initiated the *Water and Watersheds Program* in 1995 to competitively fund large, interdisciplinary teams of scientists from the natural, socioeconomic, and engineering sciences. In 1998, the US Department of Agriculture joined the partnership.

Methods

By 2000, the Water and Watersheds Research Program had funded 87 projects. It was clear that the time had come to assess the progress of these grants, not only from a scientific perspective but also from a managerial perspective. In order to do so, EPA convened a "lessons learned" workshop. Ten Principal Investigators were brought together to discuss the scientific, personnel, administrative, and institutional issues associated with managing large interdisciplinary grants. The focus of this paper is to translate their insight into useful lessons for the future interdisciplinary research manager.

Conclusions

Scientific lessons

Four primary scientific lessons were distilled from the presentations by the Principal Investigators and the ensuing discussion. These four lessons were:

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① *Interdisciplinary research is changing environmental science.*

There was consensus among workshop participants that these interdisciplinary interactions sharpened the research questions and focus, brought a freshness of perspectives to the research, provided a better appreciation of the complexity of the issues, provided new tools and techniques for studying ecological systems, and contributed to integrated conceptual models that better framed critical pathways and watershed processes. Scientists and engineers from different disciplines taught each other new ways and approaches for investigating problems that each thought they had previously understood.

② ***Interdisciplinary science is more applied and management/policy more relevant than in disciplinary research.***

Emphasizing the integration of socioeconomic and political attributes, conditions, and consequences in these projects brings interdisciplinary research into the policy and management realm. While the Water and Watershed research project investigators are not making policy and management decisions, the results and information from these projects are being injected into the decision making process. In addition, as science becomes more accountable to the public, these interdisciplinary projects are involving stakeholders in the process, demonstrating why the information is needed, and how it contributes to addressing stakeholder issues. While interdisciplinary research is more applied, good, basic, disciplinary science is accomplished within the interdisciplinary framework.

③ ***Social sciences contributions are a major strength of the program.***

Watershed management is fundamentally social in nature. Virtually every Principal Investigator (PI) indicated that the social sciences made major contributions to the natural sciences, both in providing analytical techniques and tools, and in providing insight into why various issues arise and how they can be addressed. Initially, differences in study designs (e.g., experimental versus survey), qualitative versus quantitative analytical procedures, and interpretations based on "hard" versus "soft" science required discussion. The discussions lead to overcoming preconceived prejudices and to identifying approaches for integrating results from what appear initially to be disparate data sets.

④ ***Stakeholder input and communication is invaluable to good science.***

Stakeholders provide more than just local perspective on issues and problems. They understand the community fabric that contributes to these problems. They are also the best liaisons for getting information back to the local, state, or regional communities for addressing the problems. The stakeholders can explain why the research is needed and how it will eventually contribute to resolutions of issues. Many stakeholders also bring skills and expertise that can contribute to better research question formulation, experimental design, and implementation of management alternatives.

Stakeholder involvement, however, takes time. A time frame of 3-5 years to develop trust and a good working relationship with stakeholders is not uncommon.

Personnel issues

People and their interactions are key to the success of any research project. They are particularly important in interdisciplinary research where these interactions contribute to creativity and innovation. Six lessons related to personnel emerged from the workshop discussions.

① ***A strong project leader (PI) is needed to oversee the research project.***

Given the different personalities, disciplines, departments, and institutions that are involved in many interdisciplinary projects, a strong leader is essential for overseeing the research project. Two characteristics common to all the effective Project Leaders were that they were respected by the team members and were effective communicators. Senior principal investigators also play a critical role in interdisciplinary research and must be part of a project team. Not only is their leadership experience needed to work through some of the initial communication issues, but the stabilizing influence of senior PIs is critical in moving the research through any start-up difficulties. Senior PIs can also mentor graduate students and younger faculty, encouraging and helping them maintain focus on the overall project goals and objectives.

② *All investigators need to park their egos at the door.*

This is a critical lesson that must be infused in any interdisciplinary project. This is also an area where senior PIs can lead by example. The senior PIs can raise the ego issue without appearing arrogant and ask that all individuals respect the comments, opinions, and ideas of others. Insecurities should be parked in the space next to egos. There must be mutual respect for each others' opinions, information sources, and efforts if interdisciplinary research is to succeed. This means interactive discussions, analyses, and joint authorship on publications.

③ *People must be committed to the project.*

Commitment to the project is essential if it is to succeed. There are several corollaries to this statement. First, the number of PIs on a project can get too large. While more PIs might seem advantageous, the greater the number of individuals, the smaller the role for each. In general, the degree of individual commitment to the project is inversely proportional to the number of PIs. Most individuals, appropriately, will prioritize their time on projects in which they have a major role. Attitude is an integral part of commitment, and it is critical that project morale be maintained. One "bad apple" can demoralize a team, whether it is a scientific or athletic team. Maintaining a positive attitude during those periods when studies must be redesigned or interactions among team members is strained is a challenge.

④ *A Rule of Inclusiveness must be established and sustained.*

Teams function effectively only when everyone feels part of the team. It is important that rules of inclusiveness be established at the onset of the research and that these rules be sustained throughout the project. Everyone must have an opportunity to have their ideas, thoughts, and suggestions aired and objectively discussed. The Senior PIs must also commit to these rules of inclusiveness and be both a part of the team and attend the team meetings.

Inclusivity means that everyone also has access to the same information and that information is shared among all team members via email, teleconferences, meeting minutes, team meetings, and other forms of communication.

⑤ *Younger faculty members need to maintain disciplinary expertise.*

It is important that younger faculty maintain a strong disciplinary focus because their career development and promotion comes from publications in recognized and respected disciplinary journals. As was stated above, sound disciplinary research is conducted within interdisciplinary projects and contributes to the overall project success. Most of the younger faculty will be rewarded and promoted within disciplinary, rather than interdisciplinary, departments. Therefore, it is important that they retain that focus while expanding their knowledge and vocabulary in other disciplines that can contribute to them becoming better scientists and engineers.

⑥ *Interdisciplinary programs are developing the next generation of scientists and engineers.*

One of the major obstacles to interdisciplinary research has been disciplinary jargon and terminology. Different disciplines use the same words, but with different meanings and definitions. Initially, these differences interfere with effective communication. The next generation of scientists and engineers working on interdisciplinary projects will be versed in the language of other disciplines. They will also be knowledgeable about the techniques, procedures, and methods in other disciplines, as well. Information transfer will not only be more efficient, it will be more effective in communicating with stakeholders and other sciences.

Administrative lessons

Interdisciplinary research requires greater attention to administration, coordination, and management than disciplinary research. Four lessons were identified during the workshop.

① *A management plan should be part of the research plan.*

Every research plan submitted to the Water and Watersheds Program should include a management section. Interdisciplinary projects are being conducted by personnel in different departments, colleges, institutions, agencies and organizations. This requires greater attention to how the project will be managed so that it can achieve its goals and objectives. The management plan should also consider contingencies that might arise during the project. In several projects, senior PIs left during the project, which created problems in leadership, disciplinary expertise, and interdisciplinary interactions.

② *Project management takes considerable time.*

Every Water and Watersheds Program PI indicated they grossly underestimated the amount of time required to administer and manage interdisciplinary projects. Managing these projects is a major commitment of time and resources. Learning the administrative procedures and protocols used by different departments, colleges, institutions, and agencies/organizations requires considerable time. Each PI indicated they either had, or strongly recommended, that an administrative assistant or lieutenant help administer the project.

③ *Three years is short for interdisciplinary research projects.*

Interdisciplinary research requires more time because there are communication and language barriers to initially overcome among disciplines and with stakeholders. The first year can be frustrating because of differences in definitions and semantics among project members. This improves through time, but effective lines of communication and understanding can take 3 years, or longer, to fully develop. Many of these projects also have field studies or phased research efforts with precursor research results used to refine the design of subsequent research efforts. For example, hydrodynamic models or modules might be required before water quality or biological modules are developed. A 5-year time frame would be a more realistic duration for the projects.

④ *PIs must interact routinely with team members.*

Establishing trust and communication applies not only to stakeholders, but also to the relationships among PIs and team members. It is important that PIs have regular interactions among themselves and team members. These interactions should not just be professional, but also social through parties, get-togethers, and similar events so that team building can occur. A social gathering is strongly recommended at project initiation so that people begin to feel comfortable with team members from different departments or institutions and in expressing their ideas and thoughts in an open forum. One multi-institutional project was initiated by having a 3-day workshop at the Iowa field site. This contributed to shared, place-based knowledge about the site and established strong lines of communication early in the project.

Institutional lessons

Institutional administrators, in general, are skeptical about the benefits of interdisciplinary research. Four lessons that emerged from the workshop relate both to this skepticism and to the reality of functioning within institutional constraints.

① *It pays to promote projects in your institution.*

Publicity pays. Several Water and Watershed projects received publicity through their interactions with stakeholder groups. This publicity had a positive influence on institutional support for the project. The support of the stakeholder group also contributed to public support of the projects and greater acceptance of the results and proposed management actions. This support was generated because the stakeholders were part of the project team. Team building, including T-shirts, joint field sampling, and similar approaches, was one of the major focuses of the interdisciplinary research project. Stakeholder involvement paid dividends in getting the project results considered and enacted in natural resource management decisions.

② *Departments still prevail, and younger faculty members need departments for advancement.*

Younger faculty members need to focus on publishing and participating in disciplinary journals and professional societies, respectively. As indicated above, however, sound disciplinary research is conducted within interdisciplinary projects. The interdisciplinary projects also contribute to the professional and career development of the faculty by increasing their awareness of other analytical techniques and procedures that are applicable to their research, increasing their vocabulary, perspective and insight into interdisciplinary problems, identifying additional research areas, and establishing the linkages between their research and other environmental or societal problems or issues.

③ *Multi-institution projects can work, but this requires pre-planning.*

The time to consider the intricacies of working among institutions is during proposal preparation, not after award of the contract. Budget administration, transfer of funds, distribution of overhead, administrative procedures and protocols, and personnel procedures can all create significant contractual and administrative problems for the project. Problems arising after the contract is awarded that cannot be satisfactorily resolved can seriously jeopardize project success.

④ ***The quality of graduate students and subsequent positions benefits the institution.***

Institutional benefits accrued both through the quality of the students produced from interdisciplinary programs, employers hiring these students, and the support from stakeholders using the results. Employers have been proponents of interdisciplinary projects because the students from these programs are not only sound in their respective disciplines, but also have greater breadth, being conversant and knowledgeable in other disciplines. Breadth, in addition to depth, is critical in many agency and private sector positions. Institutions producing these students will benefit from continued support and demand for programs providing this interdisciplinary background.

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San Simon Watershed Assessment and Restoration Plan

Bill Brandau, Rod Wittler, Barron Orr

Abstract

This paper describes a partnership of the Bureau of Land Management (BLM), the Bureau of Reclamation (Reclamation), the University of Arizona, and the Arizona Department of Water Resources (DWR). The partnership is assessing and restoring the San Simon watershed, a sub-basin in the Upper Gila River Watershed. EPA selected that watershed as one of 12 Clean Water Action Plan Showcase Watersheds.

The San Simon River incised early in the 20th century. Primary land uses within the San Simon Watershed include recreation, a designated OHV use area, farming, and livestock grazing. BLM installed grade control structures on the San Simon River and tributaries, beginning in the 1930s. The purpose of the grade control was to halt channel incision and degradation of the watershed land resources. In all, there are 19 major detention dams, several dikes, and earth structures. They all require assessment to determine effectiveness and condition for repair/maintenance. The Safford Field Office proposes to complete an assessment of the San Simon River watershed as a prerequisite to a community-based planning effort setting the future direction and management of the watershed.

The assessment will answer three questions:

1. Are the existing grade control structures effective?
    a) Are the structures hydraulically and geomorphically effective?
    b) Is maintenance of the existing structures cost-effective?
2. Would additional grade control structures hasten the restoration of the river and tributary channels, and aid in the recovery of the watershed?
3. How does the Resources Inventory and Assessment of the uplands and the San Simon channel depend upon the grade control structures?

Keywords: watershed, restoration, incision, fluvial geomorphology, resource inventory

Introduction

In 2001, the BLM Safford Field Office, in conjunction with the San Carlos/Safford/Duncan Watershed Group (SSD), began a four year community-based watershed evaluation and plan for the San Simon River and watershed. Figure 1 shows the location of the San Simon watershed in southeastern Arizona. There are some anecdotal signs of resource improvement on the San Simon. However, there is uncertainty surrounding the rate of improvement. The purpose of this planning effort is to determine the overall effectiveness of the restoration work done on the San Simon over the past 70 plus years, and where to focus future restoration efforts.

Preliminary plans are for the Bureau of Reclamation to conduct a fluvial geomorphology study of the San Simon River corridor. This study will determine if the San Simon has achieved a new dynamic equilibrium. The study will also focus on causation in areas that continue to erode. The fluvial geomorphology study is an extension of a larger effort on the Upper Gila Watershed by Reclamation and the SSD. The SSD, in cooperation with the Bureau of Reclamation, is currently conducting a fluvial geomorphology study of the main stem of the Gila River, to which the San Simon is a major tributary.

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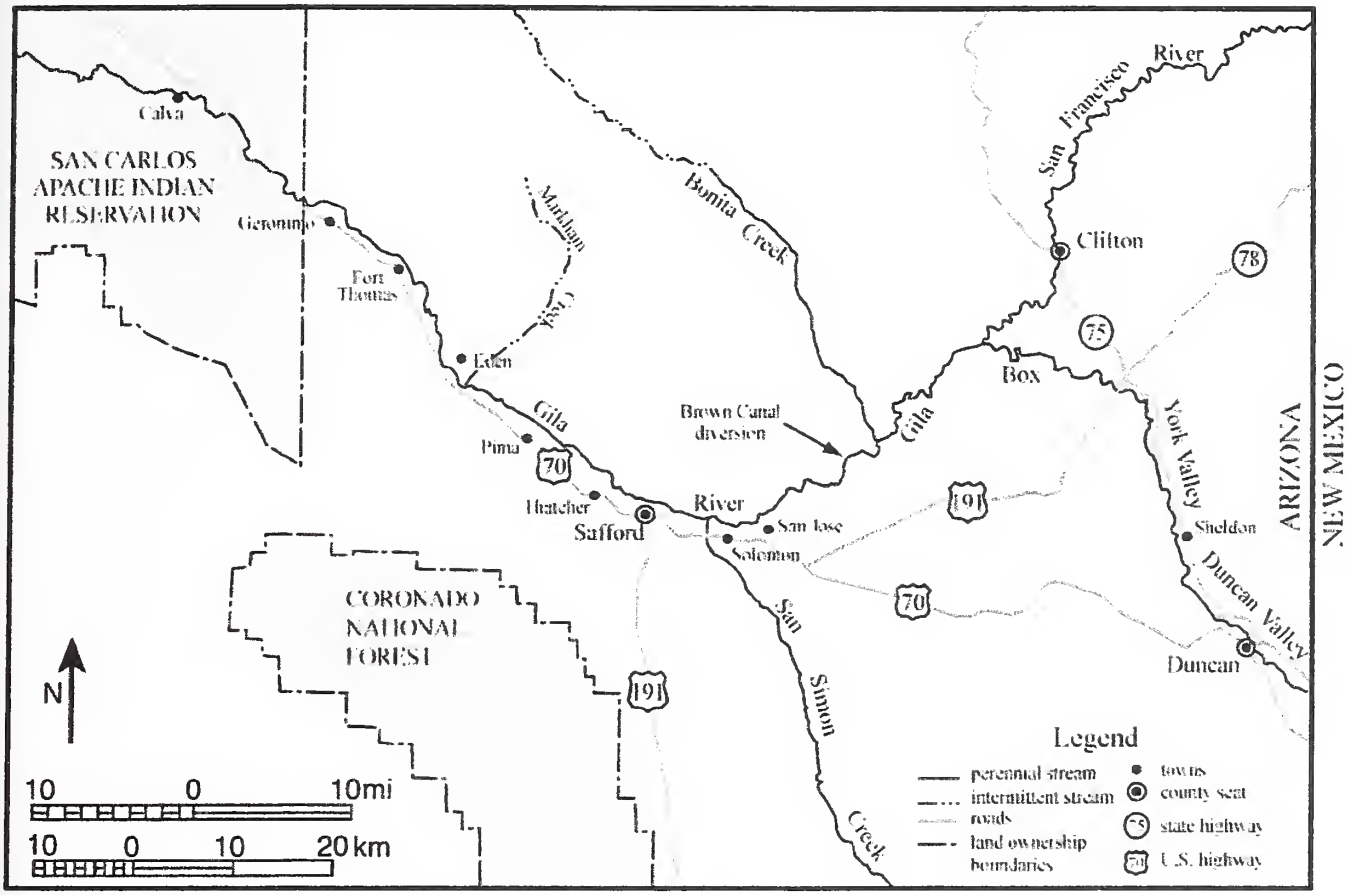

Figure 1. Location map of San Simon Creek (River).

Upon completion of the initial resource evaluation, the partners will develop a community-based plan that will present recommendations for future restoration actions on the San Simon. Actions could be: to do no more work, maintenance of existing structure, abandonment of some existing structures, construction of additional structures(which have been proposed in previous plans, but not constructed), increase effort on small scale erosion control work on the upper watershed.

Project Description

The Upper Gila River Watershed in Arizona is one of the most important watersheds in the nation. The Environmental Protection Agency identified the watershed as one of 12 showcase watersheds under the Clean Water Action Plan. The San Simon is a sub-watershed, part of the Upper Gila River Watershed. The San Carlos/Safford/Duncan Watershed Group (SSD) identifies the San Simon as an important component of the causative factors on the mainstem Gila River below Solomon, Arizona. In FY 2001, the BLM determined that the San Simon watershed is one of the top ten watersheds in need of restoration.

The BLM Safford Field Office (SFO) administers about 750,000 acres in the watershed. Primary land uses within the San Simon Watershed include; Recreation, a Designated Off-Highway-Vehicle use area, farming and livestock grazing. The primary objective of the management is to improve water quality by decreasing San Simon watershed silt load and salt load into the Gila River. There is a present need to evaluate past restoration work, to determine its effectiveness, and to evaluate resource conditions of the San Simon. A detailed resource inventory and assessment will form the basis of future planning efforts.

The San Simon River was incised early in the 20th century. In an attempt to stop this incision, slow sediment delivery, and restore the river channel, BLM installed grade control structures on the San Simon River and its tributaries. This was a multi million-dollar effort. The structures are numerous, including at least 19 major detention dams, several dikes, and several earth structures, totaling miles in length. With age, these now require assessment to determine effectiveness and condition of the structure for repair/maintenance. Determining the effectiveness of the watershed restoration efforts in the past 70 years on the San Simon is critical to any future planning effort. The Safford Field Office proposes to complete an entire assessment of the San Simon River watershed as a prerequisite to a community-based planning effort the would set the direction for management of the watershed in the future. The three phases of the assessment are detailed in the following sections.

Phase I: Geomorphic and engineering analysis of the San Simon River and tributaries

There are four primary tasks necessary to complete Phase I of the assessment. The following list summarizes the major tasks:

- Task 1 – Mapping
- Task 2 – Catalog of Historical Changes
- Task 3 – Geomorphic and Engineering Analysis of Channel
- Task 4 – Channel Assessment Including Structures and Predictive Qualitative Model

The purpose of the four primary tasks is to answer these fundamental questions:

1. Are the existing grade control structures effective?
   a) Are the structures hydraulically and geomorphically effective?
   b) Based upon their hydraulic and geomorphic effectiveness, is maintenance of the existing structures cost effective?
2. Based upon the status and effectiveness of the existing structures, would additional structures hasten the restoration of the river and tributary channels and aid in the recovery of the watershed?
3. What are the critical hydraulic, geomorphic, range, and resource considerations for future management decisions on the San Simon River and major tributaries.

Methods

Orthophotography – Orthophotographs are scheduled to be completed as part of an existing project with existing funds. BLM and ADWR are funding the aerial photography and ground control, as well as the orthophotographs.
Topography – Detailed Digital Terrain Model (DTM) will be constructed in the state plane coordinate system.

Catalog of Historical Changes – Collect Historical Photographs & Historical Information – Reclamation will collect, review, and document historical information about the San Simon River. This information includes historical aerial photographs, maps, photography, and written accounts. Reclamation will then use this information to develop a history of channel changes.

Digitize 1935 Plane Table Survey – BLM will provide to Reclamation either the original or a copy of the 1935 Plane Table Survey now in the possession of BLM. Reclamation will digitize the survey map for comparison of the 1935 topography to the 2001 DTM.

Volumetric Comparison 1935 to 2001 – Reclamation will compare the 1935 plane table survey of the San Simon River to the 2001 topographic map. Using computer based CAD tools, Reclamation will calculate changes in the volumes of material along the river channel compared to 1935. This calculation provides a quantitative measure of channel change over this 66-year period. Areas of volume loss and gain will be depicted on the map along with the corresponding amounts.

Develop History of Channel Changes – Reclamation will use the catalog of historical photographs and historical changes to chronicle the changes in the San Simon River channel. Reclamation will document progression of the historical channel changes of the San Simon River based on aerial photography, other historical information, and comparison of the 1935 and 2001 topography. The history of the channel changes facilitates comparison of historical channel change, watershed changes, and land use.

Compare, Analyze, and Summarize – Reclamation will compare, analyze, and summarize the results of the volumetric analysis and the history of channel changes. Reclamation will develop a model of historical channel change.

Map Channel Characteristics – Reclamation will divide the San Simon River into mapable units based on physical characteristics of the channel.

Reclamation will then map the river using these units, forming the basis for future channel management.

Analysis of Geologic Controls on Channel Erosion – The underlying geology controls the incision of the San Simon River and its key tributaries. The extent of key geologic units may control future incision. The identification of these natural controls and their location will play a vital role in the development of a predictive model of channel change. In particular, these controls and their locations may inform management decisions of where to place or not place additional grade control in the system if additional grade control is necessary and potentially effective.

Quantification of Channel Characteristics – Reclamation will measure the physical (hydraulic, bank geometry, geomorphic) characteristics of key channel and tributary reaches. These measurements are important for the monitoring of future channel changes resulting from management actions, the development of a predictive model of channel change (Task 4), and for use in the stable channel analysis.

Hydraulic Analysis of Grade Control Structures – Reclamation will evaluate individual grade control structures for their hydraulic and geomorphic effectiveness. The evaluation includes determining the hydraulic range of influence upstream and downstream from the grade control structure, and an estimate channel change since installation of the grade control.

Stable Channel Analysis – This cross-section based analysis of channel stability balances sediment transport, channel roughness, and water discharge. It will provide an indication of channel stability relative to slope, width, and depth. This will help determine the effectiveness of existing and potential future grade control. In particular, Reclamation hypothesizes that the stable channel analysis will indicate if there are reaches where the channel is too steep, thus requiring additional grade control, and how the channel has approached stability in areas upstream of existing grade control structures.

Channel Assessment including Structures and Predictive Qualitative Model – Reclamation will develop a coherent qualitative model of potential future San Simon River channel change. The model will include existing grade control and identify locations to effectively place grade control structures in the future. The results will also identify any current grade control that is either ineffective or not serving its intended purpose. This model will form the basis for future management decisions for the San Simon River and major tributaries.

Phase II: Resources inventory and assessment of the uplands and the San Simon channel

In addition to the Geomorphic and Engineering Analysis, the SFO feels there is additional information needed to complete the watershed assessment. They include:

- Inventory of existing grade control structures, dikes, and dams
- Road inventory
- Retaking historic photos
- Ecological site inventory
- Complete cultural resource inventory
- Complete stream and riparian inventory
- Complete a threatened and endangered species habitat inventory
- Avian corridor inventory and other wildlife inventories
- Complete standards and guidelines evaluation and rangeland monitoring
- Complete paleontology inventory
- Develop a GIS database

First year

- **Inventory of Existing Grade Control Structures, Dikes and Dams**. Beginning in 1930 the CCC, Soil Erosion Service, SCS, BLM and private parties began construction a variety of grade control structures in the San Simon Watershed. There has never been a complete inventory of these structures.
- **Road Inventory**. Roads are suspected to be a major contributing factor to sediment loading directly into the San Simon Channel, and indirectly into the Gila River. Together these water bodies support 3 T&E fish species and one T&E bird species. Roads also contribute to surface erosion factors which contribute to the deterioration of numerous cultural resources.
- **Orthophotography**. Orthophotographs are part of an existing project with existing funds. BLM and ADWR are funding the aerial photography and ground control, as well as the orthophotographs.

Second year

- **Repeat Historical Photography**. There are several hundred historical photo of the San Simon Valley dating from 1860-1970. These photographs provide a valuable insight to the change on the San Simon. Selected scenes would be re-photographed.
- **Ecological Site Inventory**. This effort is in direct support of Arizona Standards for Rangeland Health. It provides an inventory of ecological sites based on soil and vegetation. This inventory will serve all resource disciplines by providing ecological status for DPC objectives. This inventory is being developed for GIS format and will fulfill BLM Geospatial Metadata standards. This is part of a coordinated effort with the Coronado National forest and the University of Arizona.
- **Complete Cultural Resource Inventory**. A Class II cultural resource inventory would be completed covering the San Simon Valley. Sample units would be selected along the eroded San Simon channel and major tributaries to inventory at the Class III (intensive) level. An ethnoecology study would be completed by contract for the San Simon watershed. The study would consist of library/archival research and informant interviews to collect information on: the watershed's past environment; historic and present land uses; changes in the environment resulting from these uses; the areas present environment; and recommendations for management.
- **Complete Stream and Riparian Inventory**. Research and develop inventory methods for ephemeral drainages. In coordination with a State-wide effort in Arizona, partnering with USFS and the National Riparian Team.
- **Complete a Threatened and Endangered Species Habitat Inventory**. Black-tailed prairie dog inventory for potential habitat.
- **Avian Corridor Inventory and Other Wildlife Inventories**. Complete inventory of avian species within the San Simon watershed. This area appears to function as a sky island corridor connecting mountain ranges in Mexico with the Gila River.
- **Complete Standards and Guidelines Evaluation and Rangeland Monitoring**. AZ Standards for Rangeland Health Evaluations within the San Simon watershed. This effort will consist of monitoring indicators of rangeland health to determine if allotments within the watershed are meeting the Arizona rangeland health standards. This is part of a coordinated effort with the University of Arizona and the Coronado National Forest.
- **Complete Paleontology Inventory**. Paleontological resources have been found at the 3,000-foot elevation level. Excavations are on going and efforts have been coordinated with the Mesa Museum. These resources have included glyptodons, tortoises, and bears. These resources may be extensive, and there is a need to do a complete inventory.
- **Develop a GIS Database**
- **Project Coordinator**. Oversees administration of all resource inventory contracts throughout the assessment and planning efforts for the San Simon Watershed. The Contract Administrator would assures product review is performed by all BLM and Reclamation specialists, assure quality control, and tracks scope of work, time tables and budget.
- **University of Arizona - Office of Arid Land Studies**. An academic course began in the fall of 2001 that explores aspects of resource use and management in arid land ecosystems. It focuses on the drivers and consequences of management interventions, coupled with natural environmental change. Students are currently organizing old archive files, using remote sensing to map sheet eroded areas, and retaking historical photographs. Dr. Ray Turner, author of *The Changing Mile*, has volunteered his services as part of this project.

Third year

- **Ecological Site Inventory**. This effort is in direct support of Arizona Standards for Rangeland Health. It provides an inventory of ecological sites based on soil and vegetation. This inventory will serve all resource disciplines by providing ecological status for DPC objectives. This inventory is being developed for GIS format and will fulfill BLM Geospatial Metadata standards. This is part of a coordinated effort with the Coronado National forest and the University of Arizona.
- **Complete Cultural Resource Inventory**. A Class II cultural resource inventory would be completed covering the San Simon Valley.
- **Stream and Riparian Inventory**. Research and develop inventory methods for ephemeral drainages. In coordination with a State-wide effort in Arizona, partnering with USFS and the National Riparian Team.

- **Avian Corridor Inventory**. Complete inventory of avian species within the San Simon watershed. This area appears to function as a sky island corridor connecting mountain ranges in Mexico with the Gila River.
- **Paleontological Inventory**. Continue ongoing inventory of paleo resources found within the 3,000 foot elevation zone of the Whitlock Mountains.
- **Continue to Develop a GIS Database**

Fourth year

- **San Simon Watershed Assessment**. Assimilation of resource inventory reports and results of geomorphic studies. Complete all data digitizing into GIS, report preparation and publishing.

Phase III: Community-based partnership

As part of this phase a community-based partnership training would be offered to the community. This would allow for a coordinated and interdisciplinary planning effort. This effort is necessary to allow the community to partake in planning for the management goals and objectives for the watershed and development implementation actions necessary to achieve them.

Project goals are:

1. The determination of effectiveness of 70 years of watershed restoration efforts.
2. Determine the status of the San Simon Watershed, both uplands and channel.
3. Proper inventory, evaluation and assessment of all resources.
4. Management of threatened and endangered and special status species.
5. Identifying long-term uses and management opportunities and goals for the watershed and its natural resources.

At this point, inventory and assessment of the structures will have been completed. Resource inventories will be underway and some will be complete enough to begin forming analyses and interpretation. There will be a big picture to look at with fragmented resource inventories in place. It is now time to form the official community-based partnership and work group to look at the big picture, and discuss the social/political questions provided by the restoration effort.

Fifth year

- **Community-Based Partnership**. Provide training locally to interested public for development of a watershed plan. Identify committed participants for the planning effort.
- **Project Coordinator**. Once the Watershed Assessment contracts are completed, the project coordinator would assist the BLM field office planner with the community-based planning effort. This coordinator would also assist with efforts of the Safford/San Carlo/Duncan Watershed Council.

Sixth year

- **San Simon Watershed Plan**. Cost associated with plan development and printing.
- **Project Coordinator**. Once the Watershed Assessment contracts are completed, the project coordinator would assist the BLM field office planner with the community-based planning effort. This coordinator would also assist with efforts of the Safford/San Carlo/Duncan Watershed Council.

Benefits

The benefits of the assessment would be the determination of the overall effectiveness of the past 70 years of restoration work and identify where we need to focus restoration efforts to improve and restore the watershed condition for our community-based planning effort. Specifically, we would be able to characterize the environmental changes over the last 70 years and assess probable causes and consequences of changes to base our planning effort on.

The affect of the uplands on the water quality in the Gila River has been an issue for many years. There are three threatened and endangered fish species in the Gila River and tributaries, including the spikedace, razorback sucker, and loach minnow. There are two threatened and endangered birds species, the cactus ferruginous pigmy owl and southwestern willow flycatcher are also found along the Gila River and possibly some tributaries. In the uplands of the San Simon there is the lesser long nose bat. Section 7 consultation is complete on all species for the Safford Field Office Resource Management Plan and the grazing program. Upon completion of the community-based watershed plan there may be a need to re-consult.

Water quality and quantity have been a long standing issue on the Upper Gila Watershed, with on going litigation over both subjects. The Clean Water Act placed the responsibility upon the states to implement many portions of the Act including reduction in non-point source pollution. The Clean Water Action Plan has directed agencies to take a holistic management approach to improving the water resources. This effort is being made through the Gila Monster Watershed Council for the whole Upper Gila Watershed and through the San Carlos/Safford/Duncan Watershed Group in Arizona. This effort was initiated in 1993 and is a community-based effort that has wide local and national support.

Public lands in the San Simon Watershed contribute substantially to silt load entering into the Gila River and into the San Carlos Reservoir. During the 1930s the US Geological Survey estimated as much as 30% of the silt load entering into the San Carlos Reservoir came from the San Simon watershed. Over the years the watershed has improved but is still a highly eroding area. Much of the watershed of the Gila River is on public lands; effective management will require a joint effort from both public and private lands.

Budget and personnel constraints have prevented the bureau from doing a proper assessment and planning for the watershed. Another major concern is the inability to do proper maintenance on roads and existing structures that affect watershed function. Without proper maintenance these contribute to soil erosion and deterioration of water quality in the Gila River. Roads need to be inventoried and classified and their contribution to sediment monitored. Future conservation work and management needs to take place within the watershed in order to be in compliance with parts of the Clean Water Act.

Partnerships

The Upper Gila Watershed has many on going projects that focus on public and private lands. Many cooperators participate. The primary funds come through:

1. Arizona Water Protection Fund (~$500,000 grant)
2. Bureau of Reclamation (~$500,000 grant)
3. University of Arizona ($150,000 through in kind service)
4. Arizona Department of Environmental Quality, 319 grants for various projects ($100,000)
5. Arizona Department of Water Resources various grants ($78,000 grant)
6. Watershed Affiliates ($200,000 through in kind service).

The following list of watershed affiliates have been contributing to similar projects on the Gila River through the Arizona Gila Monster Watershed Council. The San Simon discharges into the Gila River and the San Simon project is of great interest to the Gila Monster Watershed Council. The affiliates include: Gila Valley NRCD; Farm Bureau; Phelps Dodge Mining Company; Arizona Cattle Growers; Gila Valley Irrigation District; Franklin Irrigation District; Graham and Greenlee County Boards of Supervisors; City Councils of Safford, Thatcher, and Pima; Chambers of Commerce of Safford, Thatcher and Pima; The Nature Conservancy; San Carlos Apache Tribe; Natural Resource Conservation Districts; Federal agencies including Bureau of Reclamation, Environmental Protection Agency, U.S. Forest Service, and U.S. Fish and Wildlife Service; State agencies including Arizona Game and Fish Department, Arizona Geological Survey, Arizona Department of Environmental Quality, and Eastern Arizona College; and a variety of private citizens.

Conclusions

Together, the partners are prepared to study the past restoration actions, and to propose new actions to accelerate watershed and river restoration on the San Simon River and Watershed. The study is causation based, using the latest in analytical and resource inventory tools.

Acknowledgments

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Research and Management Partnership for Stream Habitat Inventory in the Appalachians

C. Andrew Dolloff, Craig N. Roghair, John D. Moran, Dawn M. Kirk

Abstract

The U.S. Forest Service Center for Aquatic Technology Transfer (CATT) has worked with resource managers from the George Washington-Jefferson National Forest (GWJNF) since 1995 to collect and analyze stream habitat data from watersheds across the Forest. Data summaries are used for project-level analysis and monitoring and in revising or establishing the desired future condition of watershed-related attributes. To date, the CATT has inventoried 298 streams (624 stream kilometers) using basinwide visual estimation technique (BVET) habitat surveys. A typical inventory includes estimates and measurements of several habitat attributes, such as amount of large woody debris, in every habitat unit (e.g. pool, riffle) encountered. BVET surveys provide large amounts of data that can be compared within or between streams at scales ranging from stream reaches to drainage basins. In 1999, the CATT and the GWJNF began to enter all summary data into a geographic information system, making comparison between streams and watersheds much easier than in the past, and providing managers with a powerful tool to aid decision-making. The CATT and the GWJNF are working together to improve the functionality of the existing GIS and to anticipate incorporation of the BVET data into larger corporate databases such as the Forest Service's Natural Resource Information System.

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Keywords: BVET, stream habitat, George-Washington-Jefferson National Forest, large woody debris

Introduction

The George Washington-Jefferson National Forest (GWJNF) manages 1.8 million acres of land within eight major watersheds, primarily within Virginia (Figure 1). Over 2,300 miles of perennial streams, 1,000 of which are classified as 'trout waters,' flow through GWJNF managed lands in the Blue Ridge and Valley and Ridge physiographic provinces. In addition to providing a diverse fishery, Forest waters support over 100 species of freshwater fish and mussels, of which 26 are listed as threatened, endangered, or sensitive.

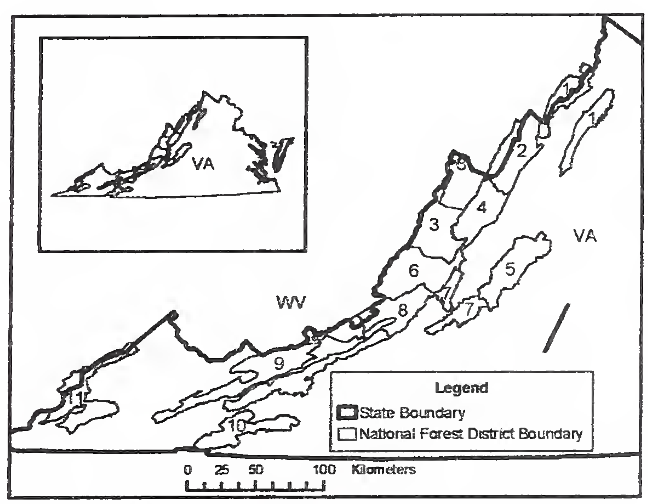

Figure 1. Location of GWJNF and associated ranger districts. 1 = Lee district; 2 = Dry River district; 3 = Warm Springs district; 4 = Deerfield district; 5 = Pedlar district; 6 = James River district; 7 = Glenwood district; 8 = New Castle district; 9 New River Valley district; 10 = Mt. Rogers district; 11 = Clinch district.

Table 1. Total number of streams and stream kilometers surveyed on the GWJNF using BVET habitat surveys between 1995 and 2002, and number of streams with less than 125 total pieces of LWD per kilometer.

Ranger District	Year	Streams Surveyed		Less than 125 pieces per km[1]	
		(count)	(km)	(count)	(%)
Pedlar	1995	60	191	45	75
New Castle	1996	31	81	15	48
Glenwood	1997	39	102	22	56
Mt. Rogers	1998	61	150	19	31
New River Valley	1999-2000	24	100	9	38
Lee	2001	47	140	27	57
Dry River	2002	36	78	26	72
Total		298	624	163	55

[1] Streams with less than 125 pieces of LWD per km do not meet the GWJNF DFC for LWD.

GWJNF resource managers recognize the importance of monitoring and evaluation in the management of aquatic resources. Stream habitat data are needed to ensure that effective standards and guidelines are established to provide habitat for the persistence of species as directed by the National Forest Management Act of 1976, and to meet the requirements of Federal environmental laws, including the National Environmental Policy Act, Clean Water Act, and Endangered Species Act. To this end, the GWJNF has engaged in a collaborative partnership with the U.S. Forest Service, Southern Research Station, Center for Aquatic Technology Transfer (CATT) to collect, analyze, and report stream habitat data.

Since 1995, we have collected over 600 kilometers of habitat data on 298 wadeable streams within 7 GWJNF ranger districts (Table 1). The underlying methodology used to collect habitat data has remained consistent, allowing us to compare results both within and between watersheds. However, the types of data collected and the tools used to collect and report them have changed in response to changing Forest needs and technological developments. Our purpose here is to describe the continuing evolution of BVET habitat survey data collection, analysis, and reporting processes within the context of the GWJNF stream monitoring program.

Methods

In 1995, we collaborated with resource managers from the GWJNF to develop a standard stream habitat survey based on visual estimation methods (Hankin and Reeves 1988). The basinwide visual estimation technique (BVET) habitat survey allowed us to collect data on a pre-selected set of habitat attributes in an entire watershed. The original habitat survey included attributes to quantify the total surface area in pools and riffles, water depths, quantity of large woody debris (LWD), and bankfull channel and riparian widths. In 1998, the GWJNF requested that we add attributes to describe substrate composition, distribution of channel types (Rosgen 1996), channel gradient, and water temperature.

Here, for the sake of simplicity, we will limit our methods, results, and discussion to the inventory of LWD using BVET habitat surveys in GWJNF streams.

Field work

The BVET habitat survey is performed by a two-person crew using visual estimation techniques (Dolloff et al. 1993). The crew enters the stream at a recognizable location, such as the Forest boundary, and proceeds upstream recording habitat attributes. The crew divides the stream into individual habitat unit types (e.g. riffle, pool), records the location (distance from survey start point) of each habitat unit, and records visual observations of habitat attributes such as the amount of LWD within the bankfull stream channel in each individual habitat unit. LWD are recorded in one of four size categories during the survey (Table 2). Data are recorded electronically on field data loggers.

Table 2. LWD size classes used during BVET habitat surveys in GWJNF streams.

Size Class	Length (m)	Diameter (cm)
1	>1 and < 5	10-55
2	>1 and <5	>55
3	>5	10-55
4	>5	>55

Data analysis and reporting

Since a typical survey includes hundreds of individual habitat units (pools and riffles), basinwide habitat surveys quickly generate large amounts of data. Although it is possible to summarize BVET data by hand, it is not practical. Between 1995 and 1998 we used commercial statistical software packages to analyze BVET data but as the surveys evolved we needed an analysis tool that provided more flexibility. In 1999, we began analyzing BVET data with spreadsheet based software that is more easily manipulated in response to changing data collection methods and reporting needs.

Since 1995, we have prepared and delivered annual hard-copy and electronic media reports that provide basic data summaries; for example, see Duty et al. (2002). The reports include both tabular and graphical presentations of various analyses for all stream habitat attributes surveyed. For example, in the case of LWD, we calculate the number of pieces of LWD per kilometer for each stream and present the data in a table with results from all other streams for data comparison. We also include a figure showing the distribution of LWD along the entire length of the stream.

Geographic information systems (GIS) are now widely used in data analysis and interpretation in fisheries data. GIS combines spatial data, such as stream location, with descriptive attribute information, such the amount of LWD, to provide a powerful analysis tool. Several pre-requisites must be met to develop an effective GIS, including: 1) consistent data collection methods over time, and 2) collaboration with qualified professionals to develop the GIS tool. Once entered into the GIS, data become dynamic, making it relatively easy to compare data within or between watersheds over time.

The GWJNF recognized the potential for interpreting their BVET habitat survey data with a GIS. Their stream habitat dataset had become so large that comparisons between streams and districts could not be easily compared using hard-copy reports. A GIS could provide relatively quick answers to questions such as: 'How many streams within the GWJNF do not meet our desired future condition (DFC) for amount of LWD?' and 'Are streams that do not meet the DFC clustered into small areas or are they scattered across the Forest?' In 2001, the GWJNF contracted with the Conservation Management Institute (CMI) at Virginia Polytechnic Institute and State University to develop a GIS tool that could answer such questions. The resulting GIS incorporates several spatial data layers such as ranger district boundaries and streams locations, which are linked to attribute tables containing summaries of BVET habitat survey results. We have begun to use this tool to analyze and interpret BVET habitat survey results in a spatial context.

Wilderness watershed: An example

Hogback Creek is a second order stream located within the St. Mary's River Wilderness in the Pedlar Ranger District. The U.S. Forest Service Southern Research Station performed a BVET habitat surveys on the stream in 1989 and we performed a second survey in 2002 (Moran et al. 2003). We surveyed an additional 60 streams on the Pedlar Ranger District in 1995. The results of the Hogback Creek and additional Pedlar Ranger District surveys allow us to demonstrate some of the many ways in which the GWJNF uses BVET data to examine stream conditions.

We will use the Hogback Creek and Pedlar Ranger District surveys to show: 1) variation in LWD distribution within a single stream during a single survey, 2) variation in LWD amount within a single stream during multiple surveys, 3) variation in LWD amount in multiple streams within a single ranger district, 4) variation in LWD amount across watersheds and ranger districts.

Results

Single stream, single survey

In 1989, we found a total of 87 pieces of LWD per km in Hogback Creek. The LWD consisted mostly of pieces less than 5 m in length and 10 to 55 cm in diameter (size 1) (Figure 2). The total amount of LWD was below the GWJNF DFC. There was very little LWD in the lower 350 meters of the stream (Figure 3).

Single stream, multiple surveys

We found over twice as much total LWD per km in 2002 as in 1989 (Figure 2). In 2002, the LWD once again consisted mostly of pieces less than 5 m in length and 10 to 55 cm in diameter. The total amount of LWD was greater than the minimum GWJNF DFC and LWD was more evenly distributed throughout the stream in 2002 (Figure 3).

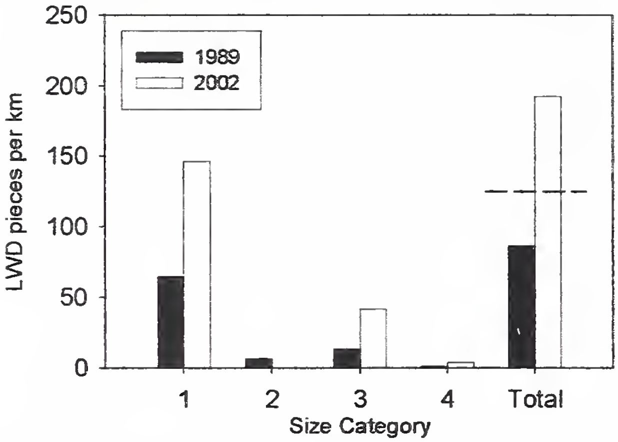

Figure 2. Amount of LWD per km in Hogback Creek in 1989 and 2002. See Table 2 for a description of LWD size categories. Total is the sum of all four size categories. Dashed line indicates GWJNF DFC for total LWD (125 pieces per km).

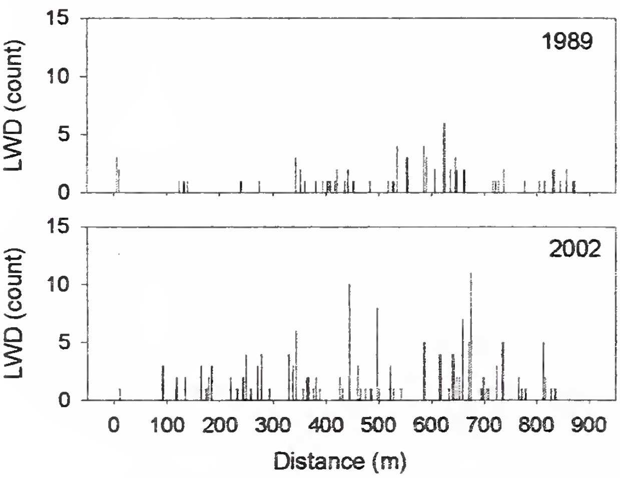

Figure 3. Distribution of LWD in Hogback Creek in 1989 and 2002. Distance is number of meters from downstream end of survey. Count is sum of all four LWD size categories.

Multiple streams, single district

In 1995, streams on the Pedlar Ranger District ranged from 3 to 369 total pieces per km, with an average of 104 pieces per km, below the GWJNF DFC (Figure 4). In 1989, Hogback Creek had approximately the average amount of LWD found in other Pedlar Ranger District streams, but by 2002 it had increased to near the 90th percentile (Figure 4).

Multiple streams, multiple districts

Overall, 55% of streams surveyed were below the DFC for LWD in GWJNF streams (Table 1). The Mt. Rogers Ranger District had the lowest percentage (31%) of streams below 125 pieces per kilometer and the Pedlar Ranger District had the highest percentage (75%).

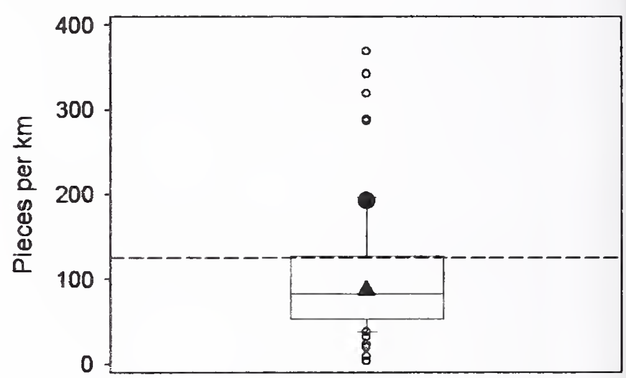

Figure 4. Amount of LWD per km in all Pedlar Ranger District streams surveyed in 1995 (boxplot), Hogback Creek in 1989 (closed circle), and Hogback Creek in 2002 (closed triangle). Dashed line indicates GWJNF DFC for total LWD pieces per km. Bottom and top of box represent 25th and 75th percentiles, solid line represents median value, whiskers represent 10th and 90th percentiles, open circles show complete range of data.

Discussion

The LWD results demonstrate the myriad ways that BVET habitat surveys can be analyzed to provide stream habitat information. For example, we can use a single survey on a single stream to locate areas in need of LWD addition. Multiple surveys on the same stream show changes in stream conditions over time. These changes may then be traced to specific causes. In the present example we suspect that the dramatic increase in the amount of LWD in Hogback Creek is related to increased tree damage and deaths related to a 1990s infestation of the watershed by gypsy moths (*Lymantria dispar*) and a present day infestation by hemlock wooly adelgid (*Adelges tsugae*). Surveys can also be compared across ranger districts or entire Forests to help allocate resources to the areas in greatest need. In addition, such data can be linked to a GIS to analyze stream habitat conditions in the context of past or proposed future land uses.

The numbers of ways in which BVET habitat data may be analyzed and presented are limited only by the amount of time available and the reporting tools available to the analysis team. Hard copy reports are important tools for presenting summary information to biologists, administrators, and the public on a annual basis. A GIS tool centralizes results contained within hard-copy reports, provides for relatively easy comparisons over space and time, and can also incorporate other spatial data layers such as timber cuts, road crossings, etc. to provide a better planning tool and clearer interpretation of results.

The GWJNF has already made use of its Forest-wide analysis capabilities. The Forest used BVET data to assess status of streams when preparing their new Land Resource Management Plan that provides direction for Forest management over the next decade.

In the near future the majority of the BVET habitat data we have collected will be incorporated into the natural resources information system (NRIS), a corporate database intended to house all Forest Service natural resource related data. This will allow even greater potential for spatial and temporal data analysis as we will be able to compare data collected on the GWJNF to BVET habitat survey data collected from watersheds in other National Forest across the country.

Acknowledgments

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Integrated Watershed Management: A New Paradigm for Natural Resource Management

Vicente Lopes, J.E. de Steiguer

Abstract

Watershed management has been developed primarily as a technical field. Major intellectual and fiscal investments have gone into exploring the biophysical dimensions of processes in the "natural environment" and the quest for technical "solutions" to perceived "problems" by experts. Traditionally, the profession and the disciplines of watershed management have dealt only with the consequences of human activity, ignoring processes that govern human activity itself. With sustainable development becoming a widely recognized management goal, there is an urgent need to integrate the human dimension into the management strategy. The emerging field of integrated watershed management (IWM) recognizes the importance of the human dimension and the need to integrate technological tools with broad-ranging social, political and economic change. Instead of focusing exclusively on biophysical processes and human impacts, IWM includes stakeholder participation, adaptive management and experimentation at scales compatible with the scales of critical ecosystem functions and services. In this paper we explore the integrated paradigm to watershed management and present a case study to illustrate the importance of community learning and collaborative planning in this type of approach. We conclude that for integrated approaches to watershed management to succeed, stakeholders must be willing to engage in all aspects of the management strategy, experts must be willing to learn skills outside their areas of expertise, and institutions must change traditional modes of operation.

Keywords: watersheds, integrated resource management, collaborative planning, community learning

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Pulses: The Importance of Pulsed Physical Events for Louisiana Floodplains and Watershed Management

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Abstract

In the PULSES project, we studied the impacts of restored flood inputs from the Mississippi River into coastal marshes of the Breton Sound watershed, Louisiana. This diversion has multiple objectives including maintaining a desirable salinity gradient, restoring deteriorating wetlands and enhancing fisheries. Pulses ranged from 185 m^3 sec^{-1} for high flow, and 15 m^3 sec^{-1} for low flow. The high river pulse resulted in nearly 30 % of the discharge flowing over the marsh, while during the low pulse, most river water was confined to channels. Sedimentation on the marsh surface during the pulse was locally high in areas within 10 km of the diversion. In the upper estuary, estimated maximum removal rates of total nitrogen and nitrate during a two week pulse in March 2001 were 44 % and 57 % respectively, while phosphate and silicate were reduced by maximum values of 23 % and 38 % respectively. There was strong nitrate uptake by sediments and a major pathway of nitrogen removal was denitrification. Stable isotope analysis showed that nitrogen and carbon in river water were incorporated into estuarine organisms such as shrimp. Spatial models predicted that species composition in emergent marsh plant communities should shift in response to salinity gradients imposed by the diversion. Socio-economic surveys documented a wide diversity of opinions regarding the costs and benefits of the reintroduced river water. Overall, results showed several strong impacts in the low-salinity (<1 psu) region near the diversion structure, with some impacts such as salinity reduction extending further down-estuary.

Keywords: river diversion, Gulf of Mexico, habitat restoration, pulses

Introduction

Over the past century, there have been many changes in the Mississippi delta. These include massive loss of coastal wetlands, salt water intrusion, and deteriorating water quality. Partially in an effort to enhance fisheries production and to address the land loss problem, diversions of Mississippi River into coastal watersheds are being carried out. Over the last century, there have been a number of changes to the Mississippi River including increasing nutrient levels, especially nitrate, decreasing suspended sediment concentrations, and the introduction of exotic species such as zebra mussels and grass carp. These changes must be taken into consideration when designing diversions. Is the reintroduction of river water restoring a natural floodplain system, a result expected if the volume and timing of freshwater inputs are most important, or is this watershed management practice of reintroducing of river water driving the deltaic system in a new direction? This latter result may occur if the combination of much-lower-than-historical riverine sediment loads, and much-higher-than-historical nutrients, exotic species

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and anthropogenic chemicals proves more dominant than simple hydrologic restoration of a river-floodplain system. This article reports on initial results from the a multi-investigator 3 year project that set out to investigate these questions.

Study Area

In the PULSES project funded by EPA, USDA and NSF in the Water and Watershed program, we studied the multiple effects of different scales of river inputs into the Caernarvon watershed, just south of New Orleans (Figure 1). River inputs have been ongoing since the 1991 opening of a gated river diversion structure. Discharge levels ranged from 185 m^3 sec^{-1} for high flow, and 15 m^3 sec^{-1} for low flow (Figure 2).

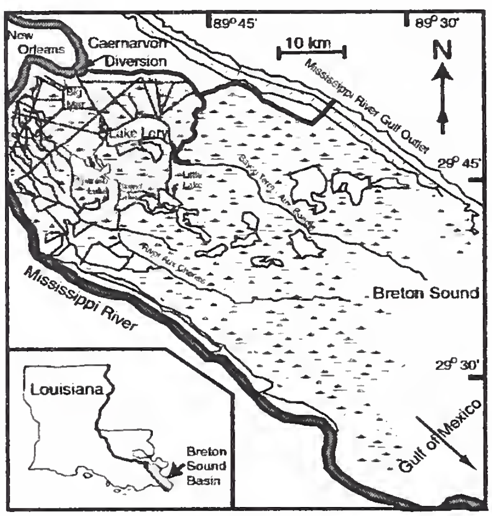

Figure 1. Breton Sound Basin with main region of estuary influenced by diversion highlighted in gray.

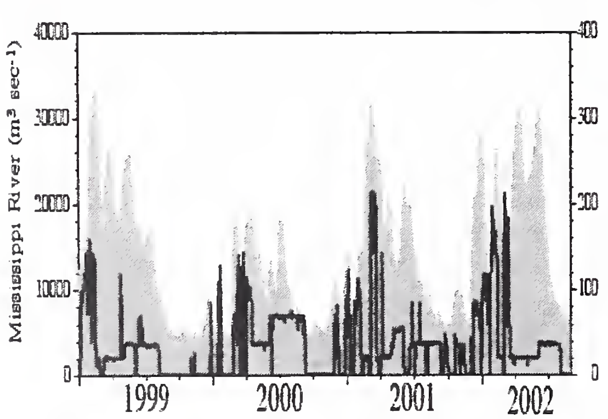

Figure 2. Mean daily Mississippi River (shaded) and Caernarvon structure (line) discharge from 1999 to 2002. Note the large pulses in 2001 and 2002 associated with this project.

Methods

Short-term sediment accumulation was measured by collecting sediment on 9 cm diameter pre-ashed, pre-weighed glass fiber filters (GF/F) with an underlying petri dish base (also 9 cm) (Reed 1989). Traps were pinned to the sediment at the level of the marsh surface, and a 0.5 inch mesh wire cage (0.3 m x 0.3 m) was anchored to the marsh around the filter pads as location markers and for protection from interference by fauna and large detritus. Filter pads were collected from the field sites at 1-4 week intervals and the filter number, location, and the date were recorded. This technique provides the net short-term deposition for individual locations, but cannot quantify deposition versus resuspension processes over the intervening sampling periods (Reed 1989).

In the laboratory, each pad was dried in an oven at 60 °C to obtain dry weight and then ashed in a muffle furnace at 350 °C (without preheating) for at least 16 hours to obtain estimates of ash weight and organic matter. Deposition of inorganic and organic sediments are reported as the mean (± 1 standard error) of 4 sediment traps at each location within a transect.

A 40-cm long x 10-cm diameter ^{210}Pb ($t_{1/2}$ = 22.3 y) core was collected on September 8, 2001 three km southwest of diversion in the marsh. The core was sectioned into 1-cm intervals immediately after collection, individually bagged and labeled, and returned to the LSU sediment laboratory. Each interval was dried overnight at 60 °C, homogenized with a mortar and pestle, packed into small volume vials, and sealed with epoxy. All core samples were set-aside for at least 30 days for the ^{210}Pb to reach equilibrium with its parent, ^{226}Ra ($t_{1/2}$ = 1620 y), prior to processing in an intrinsic germanium detector. Accumulation rates were calculated using the CIC model.

A total of 32 water quality transects were carried out in the Breton Sound estuary since September 2000 for water quality analysis. We used a flow-through system to continuously measure chlorophyll a, total suspended sediments (TSS), salinity, and temperature in major bayous and channels leading from the Caernarvon diversion to Breton Sound (Madden and Day 1992). Discrete water samples were taken at 20 locations in the estuary and later analyzed for nitrite+nitrate, ammonium, total nitrogen, total phosphorus, phosphate and silica (Standard Methods 1992).

Benthic nutrient exchange rates were estimated using a continuous flow system modified after Miller-Way and Twilley (1996) for the marsh metabolism studies. Sediment cores were taken seasonally at Big Mar (BM), Lake Leary East (LLE), Lake Leary West (LLW) and Grand Lake (GL). These stations are located with increasing distance from the diversion site. Cores were incubated with filtered water from their respective field site. Exchange rate calculations are based on concentration differences between the influent and effluent lines in steady state. Nutrient samples were measured using standard colorimetric techniques (Strickland and Parsons 1972) on a Lachat autoanalyzer.

Grass shrimp were collected 11 times between December 2000 and July 2002 at 12 stations in upper Breton Sound for stable isotope analysis. Using dip nets, three replicate samples of at least 10 individuals were obtained at each station. For each replicate, composites of the tail muscles of ten average sized specimens were dried (60 °C, overnight) and ground to a fine powder. For stable isotope analyses with a continuous flow system, we used a Carlo Erba 1500 elemental analyzer, coupled to a Finnigan Delta Plus mass spectrometer (Barrie and Prosser 1996). Our primary and secondary laboratory standards were glycine and bovine liver, respectively. The $\delta^{15}N$ values are reported relative to air N_2 that has a value of 0.0‰. The precision of our analyses was better than 0.2‰.

We have developed a general framework for the implementation of a 2-D finite-element hydrodynamic model in the Caernarvon diversion area. In a parallel effort, we have developed and calibrated an estuarine eutrophication model that includes multiple N, Si, and P uptake of the Monod type, and multiple algal assemblages, whose productivity is simultaneously dependent on nutrient concentrations, nutrient ratios, and ambient light intensity. The eutrophication model is designed to run either as a stand-alone module or as a component of a larger 2-dimensional hydrodynamic model. We also developed a landscape model to show impacts of pulsed freshwater input into the habitat (See Reyes et al. in this volume for detailed information).

We used a questionnaire-based survey for over 100 local stakeholders and face-to-face interviews with members of the Caernarvon Interagency Advisory Committee (CIAC), a legally constituted stakeholder groups of decision makers, responsible for providing inputs to the operation of the diversion facility. The CIAC includes federal, state, and local agency and government representatives, oyster, shrimp, and recreational fishers, and land-owners.

Results

The Caernarvon freshwater diversion structure discharges Mississippi River water into the Breton Sound estuary located east of the river and just SE of New Orleans. The estuary stretches SE for about 70-80 km from the diversion structure to the Gulf of Mexico (Figure 1). The upper 40 km of the estuary, encompassing an area of about 1,100 km^2, is composed of extensive marshes, small to medium size water bodies, and channels, while the lower estuary is open water in Breton Sound. Thus, the upper estuary is weakly to moderately coupled to the lower estuary due to shallow, sinuous channels and extensive marshlands. Tidal amplitudes at the Gulf of Mexico end are about 35 cm but are much less in the upper basin due to dampening effects of the marsh dominated area. In this upper estuary, winds and diversions cause much higher water level variations than do tides. Salinity is generally fresh (<1 psu) in the upper basin except during prevailing south winds or very low diversion flow.

Sediment deposition on marshes resulted from a complex set of conditions in which prevailing winds, water velocity, water levels or tides, river flow, and suspended solid loads all contribute to marsh surface delivery. Wind direction was a major controlling factor in providing both TSS and water levels high enough for marsh delivery. The Caernarvon diversion delivered sediment into the northernmost reach of the Breton Sound estuary, but strong or sustained south winds dampened diversion flow and sequestered diverted sediment in the northern estuary, thus preventing deposition in the lower reaches. Statistical analysis revealed that deposition in Breton Sound estuary varied by season, with distance from the diversion ("new" sediment source; see Figure 3), and with proximity to a major waterway.

Calculations based on results from the sediment pads indicated marsh vertical accretion could reach 2.25 cm yr^{-1}, while excess ^{210}Pb measurements recorded at the same site showed a much slower rate of 0.11 cm yr^{-1}. The ^{137}Cs sediment activities showed an annual accretion rate of 0.10 cm yr^{-1}, which agreed very well with ^{210}Pb measurements. These data illustrate an important point about marsh deposition. Short-term sediment trap measurements do not capture the effects of compaction and decomposition, and thus, represent a more ephemeral mode of deposition.

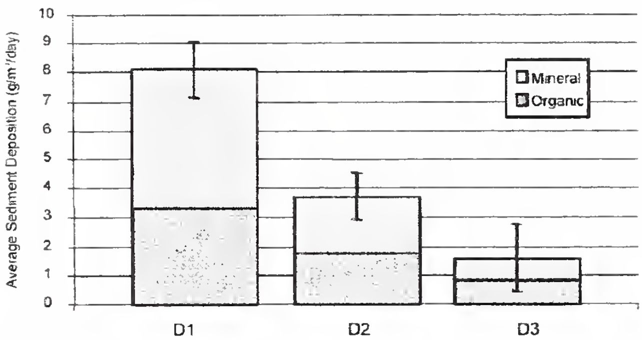

Figure 3. Average sediment deposition by sampling site distance, where D1 = < 6km (n = 5), D2 = 6 to 10 km (n = 6), and D3 = >10 km (n = 3). Background conditions at a reference station gave similar results to the D3 data. Overall deposition was highest within 10 km of the diversion.

Discharge from the diversion structure controls salinity through much of the estuary, especially during large 'pulses' when almost the entire estuary freshens. Temperature of incoming Mississippi River water ranged from 6-32 °C and generally equilibrated to the rest of the estuary within 10 km, but there were several times when cooler water from the diversion propagated through the entire estuary. During most transects, there were substantial reductions in most nutrient forms, especially nitrate, as water flowed through the estuary. Incoming Mississippi River nitrate concentrations ranged from 41-285 µmol L^{-1}, and concentrations at mid-estuary ranged from 1-75 µmol L^{-1}. Possible mechanisms for this reduction are dilution with Gulf water, rain, or ground water, and uptake by phytoplankton, bacteria, and marsh plants, denitrification, or burial. In the upper estuary, maximum estimated removal rates of total nitrogen and nitrate during a two-week pulse in May 2001 were 44% and 57 % respectively, and phosphate and silicate were reduced by 23 % and 38 % respectively. During this period, the upper estuary was almost entirely fresh, rainfall was low and we assumed that ground water was negligible. On the other hand, overland flow across marshes mixed river water with low nutrient marsh water, diluting river nutrient concentrations, and actual removal rates may be lower than the calculated maximum rates to the extent that such dilution occurred. N burial and denitrification are other sinks for N in these coastal watersheds.

These changes in nutrient concentrations during the May 2001 river pulse also led to downestuary changes in stoichiometric nutrient ratios, with an overall increase in the dissolved DSi:DIN ratio and a decrease in the DIN:DIP ratio (Figure 4, Table 1, Lane et al. in review). At this time and during other spring discharge pulses, chlorophyll concentrations near the diversion were low, but increased after suspended sediments decreased below 80 mg L^{-1} several km from the diversion structure. Overall, chlorophyll levels generally peaked at mid-estuary, and gradually decreased to low levels in Breton Sound (Lane et al. in preparation).

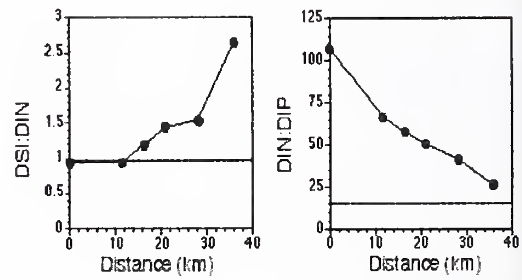

Figure 4. Molar ratios of DSi:DIN, and DIN:DIP with distance from the Caernarvon structure during the spring pulse of 2001. Horizontal dashed lines indicate the Redfield ratio. Distance was determined as a straight-line from the structure to the respective sampling stations.

Table 1. Concentrations of dissolved inorganic silicon (DSi), total nitrogen (TN), dissolved inorganic nitrogen (DIN), total phosphorus (TP), dissolved inorganic phosphorus (DIP) and salinity with distance from the Caernarvon structure during the spring pulse of 2001 (most data are from March 22, 2001).

Distance (km)	DSi (µM)	TN (µM)	DIN (µM)	TP (µM)	DIP (µM)	Salinity (PSU)
0	117.8	138.9	128.3	5.1	1.2	0
11.8	108.3	108.8	112.9	3.4	1.7	0
16.4	96.6	99.3	86.8	2.6	1.4	0
21.2	76.8	72.9	50.6	2.0	1.0	0
28.3	62.1	60.3	35.5	2.2	0.9	1.5
36.0	47.9	49.9	16.9	2.0	0.8	4.5

Nitrate fluxes in winter/spring 2002 indicated very low uptake or even efflux of nitrate from the sediment. Denitrification rates (see Kana et al. 1998) increased with increasing distance to the diversion and increasing water temperatures in the field, with maximum rates of up to 325 µmol N_2-N flux m^{-2} h^{-1} at Grand Lake (GL). In summer 2002 high nitrate fluxes into the sediments of more than 350 µmol NO_3 m^{-2} h^{-1} were estimated for cores from Big Mar (BM). Nitrate uptake rates decreased regionally from that maximum down to zero with increasing distance from the diversion. This drop is consistent with a reduction of ambient dissolved inorganic nitrate in the water to non-detectable levels (Figure 5). Denitrification rates at Big Mar were higher than 300 µmol N_2-N flux m^{-2} h^{-1} in summer and even though no nitrate uptake was detected for Grand Lake with our method, denitrification rates of up to 100 N_2-N flux m^{-2} h^{-1} were measured.

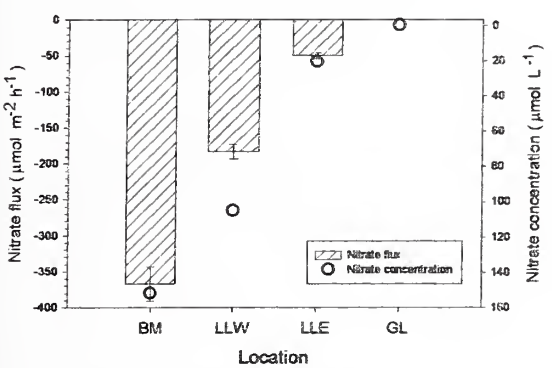

Figure 5. Nitrate flux averaged from 3 replicate cores per location (summer 2002). Bars show nitrate flux, with flux into the sediment as negative value. Circles give the ambient nitrate concentration in the water column.

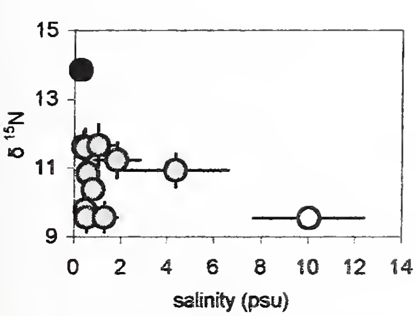

Figure 6. Average $\delta^{15}N$ (‰) values of grass shrimp muscle tissue vs. average salinity for the 12 sampling stations. Samples were collected 11 times between Dec. 2000 and July 2002. Error bars represent 95% confidence levels. The black and white symbols stand for the stations closest to the diversion and marine station, respectively.

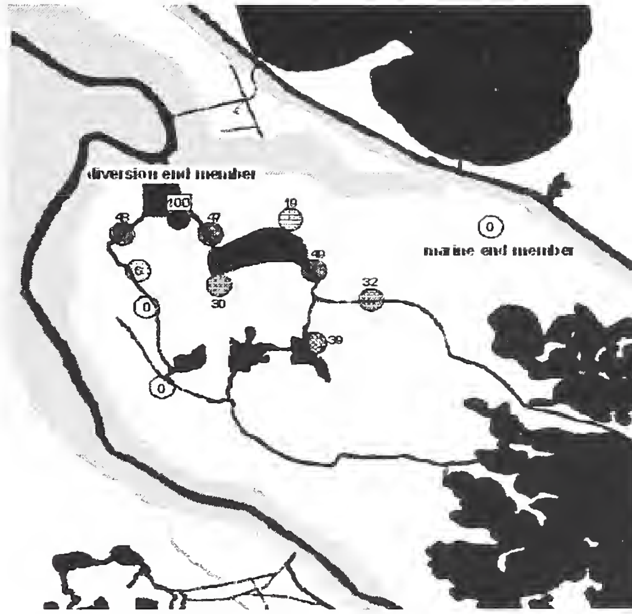

Figure 7. Average contribution (%, black = 100%, and white = 0%) of Mississippi River nitrogen to shrimp muscle tissue for the 12 sampling stations. Calculations area based on figure 3 and the assumption that the $\delta^{15}N$ values at the station closest to the diversion completely derive from Mississippi River nitrogen and that the $\delta^{15}N$ values at the most marine station are not influenced by the Mississippi River nitrogen.

The $\delta^{15}N$ values for grass shrimp showed a strong gradient throughout the sampling area, with highest values close to the diversion and decreasing values further away from it (Figures 6 and 7). River-derived nitrogen was strongly incorporated into the food web leading to grass shrimp with elevated $\delta^{15}N$ values through much of the estuary. However, at marsh-influenced sites, this effect was much diminished. The simplest interpretation of these results is that there is another nitrogen source present in marsh-influenced sites, with the source having low $\delta^{15}N$ values. Nitrogen fixation that is active in Louisiana marshes (Nixon 1980, DeLaune and Patrick 1990) could provide this source, with low $\delta^{15}N$ values near 0‰ typically associated with N from nitrogen fixation (Shearer and Kohl 1989). Use of marsh nitrogen derived from this fixation, or also possible use of dissolved organic nitrogen that may have low $\delta^{15}N$ values in river water (Fry and Allen 2003) possibly could lead to the lower $\delta^{15}N$ values in food webs at these stations.

The habitat model was used to test the effects of different management scenarios. Modeled riverine inputs had strong effects on watershed dynamics, as detailed by Reyes et al. in a separate paper in this volume.

We incorporated information from Caernarvon in an analysis of the relations among natural capital, pollution, and social welfare. Results showed that sustainable functioning of natural systems should contribute substantially to development of societal wealth in Louisiana. Preliminary results of the stakeholder analysis revealed that although there is high level agreement that coastal land loss is a significant problem, diversions are not always viewed as an appropriate solution to this problem. This is due to a combination of factors, especially that 1) local people tend to make judgments based on heuristics (e.g., personal experience for generalization), 2) there are significantly different responses among decision makers, experts, and local people, 3) channels to accommodate public opinions are available, but not actively used, therefore, 4) diverse opinions and conflicts exist and persist.

Conclusions

The ecological studies revealed very dynamic behavior of salinity, suspended sediments, chlorophyll a, and nutrients. In this system that is relatively well-flushed by riverine pulses and other weather events, year-round algal blooms were not observed. However, nutrients from the Caernarvon diversion supported high chlorophyll a concentrations in the mid estuary, especially during summer. During high discharge, chlorophyll a concentrations close to the diversion structure were low, likely due to high light attenuation caused by high suspended sediment concentrations, but chlorophyll often increased in mid estuary after suspended sediment concentrations decreased below 80 mg L^{-1}. During periods of low discharge, this mid estuary production was sustained, and was most likely supported by the regeneration of nutrients supplied in the year during high discharge. These results indicated moderate eutrophication at some sites, but added nutrients may also have beneficial effects, especially stimulating marsh productivity.

There are also active sinks for added nutrients, especially denitrification. The availability of dissolved inorganic nitrate in the water column has a major impact on observed denitrification fluxes, and especially in summer competition between plankton and sediment for potentially limiting nitrogen supplies, must be taken into account for estimations of nitrogen removal from the system.

The importance for river-derived nitrogen for the aquatic food web can be relatively high, especially close to the diversion. The overall geographic pattern of $\delta^{15}N$ values reflects the general hydrology of the Breton sound watershed, with some distinction between open water and marsh-influenced stations.

Landscape modeling and socio-economic analyses showed positive effects of the diversion, with little indications of negative impacts. This is consistent with previous analyses that emphasize the value of natural systems for a sustainable biosphere (Templet 1999, 2000). The stakeholder analysis revealed significant perception gaps among diverse stakeholders on the diversion, partially due to heuristics-based judgments, resulting in the increasing need of systematic information processing.

The results of the study suggest that there are benefits as well as potential detrimental impacts of diversions. The diversion results in nutrient uptake, some marsh accretion, lower salinities and incorporation of riverine materials into local food webs. There is concern, however, over the potential of high nutrient levels leading to eutrophication, and that long-duration pulses of one month or more can depress fisheries yields of shrimp and oysters. These costs and benefits need to be carefully considered in the management of diversions.

Acknowledgments

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Experimental Watersheds, Nutrient Dynamics and Collaborative Research

Sherri L. Johnson, Frederick J. Swanson, Kevin McGuire, Julia A. Jones, Mark Harmon

Abstract

Experimental watersheds can be a focal point for collaborative research across multiple disciplines. For over 50 years, research at the H.J. Andrews Experimental Forest, Oregon, has examined hydrologic, nutrient and sediment responses to forest harvest. Core measurements of hydrologic and climatic parameters from eight small basins, two large basins and four complete climatic stations continue to provide a foundation of high resolution, environmental data that attract multiple interdisciplinary research projects. Today, research in our small watersheds is exploring complex physical and biological pathways that control the availability, movement, and export of nutrients from basins. Due to the complexity of C and N cycles through time and space, it is difficult to predict nutrient transformations and fluxes through the terrestrial and aquatic environments. Part of the difficulty is due to the historical tendency of scientists to focus on separate aspects of these cycles. Hydrologists have traditionally considered nutrients to be transport limited, where physical processes control the availability of dissolved N and C from soils to streamwater, while ecologists generally consider nutrient concentrations to be production and uptake limited, where biological processes respond to and modify nutrient availability. In N-limited systems of the interior Pacific NW, instream uptake can sequester 30-50% of available ammonium over short distances, raising questions to what extent nutrient fluxes from basins may be representing local instream processes rather than whole basin dynamics. Results from our collaborative research show that interdisciplinary teams can stimulate learning in each discipline and are necessary to tease apart these questions.

Keywords: interdisciplinary research, hydrology, nutrients, temporal variability

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First Interagency Conference on Research in the Watersheds

October 27–30, 2003

Ecology II

Watershed Analysis of Pulsing Freshwater Events Using Landscape Modeling in Coastal Louisiana

Enrique Reyes, Robert Lane, John W. Day

Abstract

Holistic performance measures for coastal restoration are needed as management alternatives are implemented in the Gulf of Mexico. Regional questions can be addressed with watershed models using large-scale spatial dynamics to examine environmental impacts. A watershed simulation model investigated habitat shifts as consequence of different event-pulsing scenarios, and evaluated watershed health in the Mississippi delta, an area with restricted freshwater inputs. Wetland conversion to open water and yearly shifts of marsh habitats were assessed. The watershed model forecasted effects of river diversion management plans for 50 years. Results indicated that healthy functioning of this delta area depended largely on river-borne contributions. Watershed models could provide natural resource managers and decision makers with a scientific instrument for environmental policy.

Keywords: river diversions, Gulf of Mexico, wetlands, habitat restoration

Introduction

Management of diminishing natural resources and economically important ones requires a combination of insight and scientific understanding (Boesch et al. 1994). Holistic perspectives on coastal restoration efforts are needed to provide accurate assessments of different management alternatives as they are developed and implemented along the Gulf of Mexico (Templet and Meyer-Ardent 1988). To date, restoration projects have developed extensive data sets under strict scientific protocols that allow managers to rate the likelihood of success at local scales. Ecological footprints of projects can be determined and local beneficiaries identified. However, watershed level evaluations are sorely lacking (Clark et al. 2001). Compound consequences of several local projects and their future development are largely unknown.

The Caernarvon marshes in coastal Louisiana, (Figure 1), were selected to evaluate restoration efforts at the watershed level. The Caernarvon marshes lie along the eastern side of the Mississippi River, an area with tidal influences and periodic flooding due to river water. It is in this watershed that river contributions are highly regulated by a man-made diversion structure (Villarrubia 1998). Regional management questions can be addressed using watershed models that couple large-scale hydrodynamics and ecological processes (Costanza et al. 1989, Costanza et al. 1990). As river diversions become a favored tool for restoration, it is important to evaluate this approach by using a watershed model capable of forecasting such management approach to long-term regional ecosystem planning.

This paper presents the results of an environmental assessment of river diversions and anthropogenic impacts on a large watershed. The goal was to utilize a spatially-articulated watershed model to forecast planned management alternatives. Several steps were taken to achieve this goal: (1) a watershed process model was developed for the Caernarvon marshes, capable of predicting regional habitat change; (2) this model included the effects of fresh water and sediment additions from river inputs on habitat preservation, changes in forest and marsh vegetation succession, primary production rates and, soil aggradation; (3) overall, the model analyzed hydrological and ecological dynamics at the watershed level. The model tested ideas about system functioning to form the basis for sustainable

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Figure 1. Location of the Caernarvon Marshes ecosystem simulation.

management of these coastal ecosystems (Day et al. 1997).

The Caernarvon freshwater diversion outlet is among seven diversions currently in operation on the lower Mississippi River. The diversion structure dominates the freshwater inputs into the watershed. The structure was completed in 1991 and freshwater discharge began in August of that year averaging 21 $m^3 s^{-1}$.

The Caernarvon Marshes and Breton Sound Estuary are located between the Mississippi River and the Mississippi River-Gulf Outlet (MRGO), with wetlands encompassing about 63 % of this watershed (Figure 1).

Methods

The watershed model used the previously described approach for spatially-articulated watersheds (Costanza et al. 1990, Reyes et al. 2000, Sklar et al. 2001, Martin et al. 2002). This approach required the development of three simulation modules: hydrodynamic, soil dynamics and ecological productivity. These modules were combined according to a geographical grid and calibration was done using previously collected environmental data. (Table 1).

The hydrodynamic module computed the water movement and material transport with a two-dimensional, vertically averaged, finite difference solution (Singh and Aravamuthan 1995). Water inputs into the system were only the freshwater discharge and tidal exchanges. Seasonal changes in tidal height were included as boundary conditions in the Gulf of Mexico boundary.

Table 1. Environmental and model characteristics for the Caernarvon watershed.

Environmental Characteristics	**Caernarvon Marshes**
Area *	1748 km^2
Discharge volume	0 - 266 $m^3 s^{-1}$
Model Specifications	
Hydrological resolution	0.25 km^2
Length of simulation	50 years
Num. of cells	14806
Num. of modeled habitats	6

* Total area includes only natural habitats.

Processes that determined the movement of sediments in deltaic and estuarine environments occurred across a range of temporal and spatial scales. The watershed model simulated long-term geologic processes like subsidence and river shifts were included as boundary conditions. Changes in river regime were accounted for by manipulating the daily sediment concentrations and river flow.

At smaller scales, the sediment module computed the processes related to wetland formation, including resuspension, deposition, and transport of sediments, while ecological processes of soil formation and habitat change were derived from averaged results of the mechanistic processes simulated in the hydrodynamic and sediment modules.

Marsh and swamp productivity were calculated as net primary productivity for each plant community in the model. The ecological module computed above and below-ground biomass as a function of biomass, maximum growth rate and a limiting function. The limiting function accounted for daily temperature, water level, salinity, and their synergy (Hopkinson et al. 1988, Mitsch 1988, Nyman et al. 1990).

Results

The Caernarvon Watershed Model was calibrated and validated using a multiple resolution fit analysis (Costanza 1989) that compared actual habitat maps to simulated maps (Costanza et al. 1990, Reyes et al. 2000, Martin et al. 2002). A 1988 simulated map, resulting from a 10-year simulation, was compared to the 1988 USFWS habitat map (Figure 2) for calibration purposes. These 10-year calibration runs were repeated, varying the initial spatial parameters (elevation) and forcing functions (boundary salinity and tidal elevations), until a fit of 90 or greater (maximum 100) was reached. The model was accepted as calibrated after reaching a fit of 94.10. This value was well above acceptable validation standards for this type of model (Reyes et al. 2000, Martin et al. 2002).

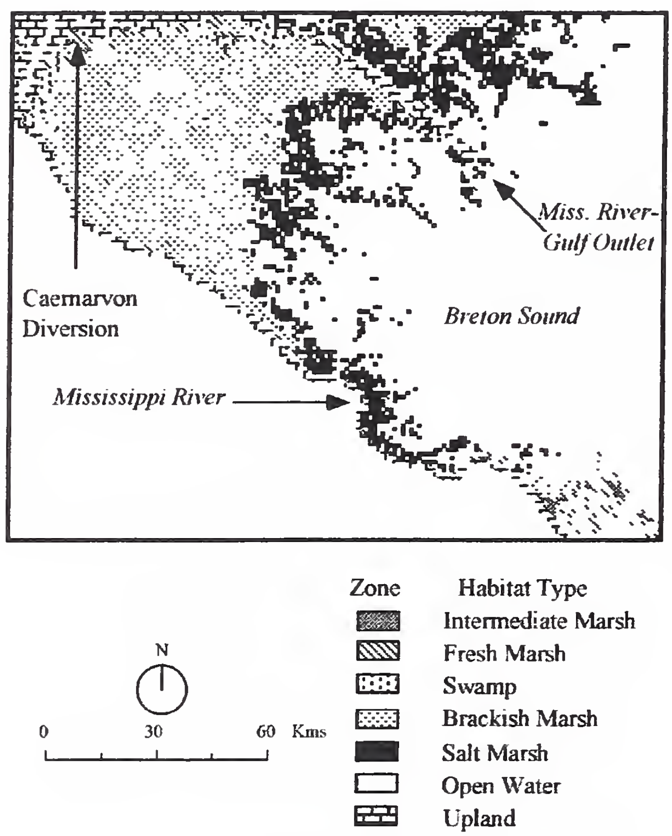

Figure 2. The 1988 U.S. Fish & Wildlife Service habitat map for the Caernarvon watershed.

The Caernarvon diversion controls the largest freshwater input to the watershed. In the past, the diversion discharge regime was established mainly to control salinity in the middle part of the basin (Villarrubia 1998). Starting in the year 2001, an experimental discharge was implemented as part of a research project (Day et al. 2000). This discharge regime involved a high flow discharge followed by a no discharge period, and thus creating a pulse in the freshwater input signal that serve to isolate response of the ecosystem. The scenarios implemented using the watershed model tested the long-term consequences of varying the discharge regime. We considered that river diversions could result in large modifications of the existing habitats. Specific scenarios tested included:

(1) No freshwater discharge (pre-1991 conditions),
(2) 1991-2000 freshwater discharge (scheduled discharge),
(3) Freshwater discharge as proposed for the Caernarvon Diversion experimental pulses regime (2001-2002 actual discharge).

These scenarios examined the main forcing function by which the marshes and open water communities experienced in these hypothetical futures. Modifying the present conditions in a step-wise manner, the model results can be used as an indication of potential cumulative impacts on these environments.

No diversion discharge scenario

This scenario was considered to be representative of the pre-Caernarvon discharge (before 1991) or when the diversion closes for management reasons. The consequences of introducing freshwater into the ecosystems were assessed by comparison with the "Scheduled Discharge" scenario.

The habitat distribution under No Discharge showed extensive salt marsh growth in the southeastern part of the watershed. It was noticeable that the establishment of salt marsh on the northeastern part of the basin continued due to the enhanced saltwater intrusion through the Mississippi River Gulf Outlet. The total extension of salt marsh under this scenario surpassed the "Scheduled Discharge" scenario by 39 km^2 (Table 2), however total land loss was only 10 km^2.

We used the resulting map from the No Discharge scenario as base for comparison with the other two scenarios. Difference maps were prepared to spatially analyze where habitat changes occurred. A difference map was computed as the result of overlaying two scenarios on each other.

Scheduled discharge scenario

This scenario tested the overall effect of using the scheduled discharge for the 1991-2000 period for 50 years. The scheduled discharge for this period included several extreme events at the beginning of the discharge schedule. These included a long period of high discharge (Oct. 1993 to Apr. 1994) and another with no discharge (Jul. to Dec. 1997). The present discharge schedule was established in 1997 (Villarrubia 1998). For simulation purposes, the period of record was repeated as necessary to cover 50 years, instead of using an annual average of the actual discharge.

Table 2. Comparison of habitat extension for future conditions (2037) under three different discharge regimes.

	Habitat Extension (km^2)		
Habitat Type	No Discharge	Scheduled Discharge	PULSES Discharge
Intermediate Marsh	2	2	2
Fresh Marsh	34	34	34
Swamp	32	32	32
Brackish Marsh	562	601	562
Salt Marsh	362	333	361
Open water	2413	2403	2413

The distribution of habitats and salinity across the watershed with the Scheduled Discharge scenario *versus* the No Discharge was substantially different (Figure 3). Freshwater inputs to the basin resulted in a reduction of salt marsh of 29 km^2. When compared to the other two alternatives (Table 2), this scenario resulted in the most favorable conditions to preserve the spatial distribution of habitats and prevent further land loss.

PULSES discharge scenario

This scenario increased the frequency of discharge without modifying the total volume delivered through the diversion. This simulated discharge regime was applied in reality to the Caernarvon watershed under the PULSES project during January 2001 and January 2002 (Day et al. 2000).

The most interesting results from the PULSES scenario were the similar values for habitat distributions when compared to the 2037 No Discharge scenario (Figure 4). There were small wetland gains in using a pulsed regime, if the total amount of water delivered to the basin is less than the total from the Scheduled Discharge scenario (Table 2).

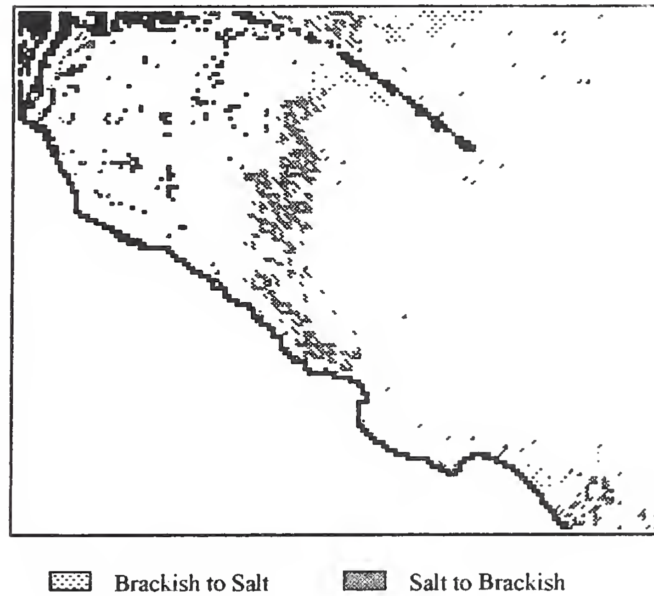

Figure 3. Habitat distribution difference for the Caernarvon watershed in 2037 under No discharge and Scheduled Regime scenarios.

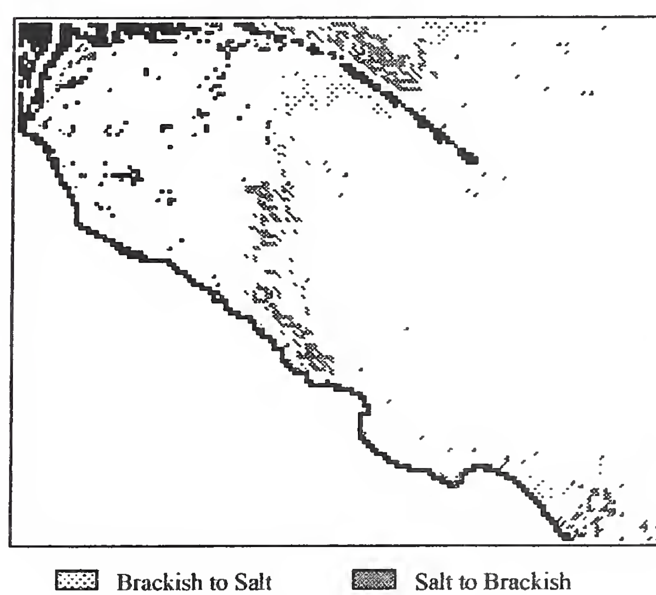

Figure 4. Habitat distribution difference for the Caernarvon watershed in 2037 under No Discharge and Pulses regime scenarios.

Discussion

Traditionally management and implementation of restoration alternatives have used a piece-meal approach, with permits and environmental impact assessments issued for small land parcels. The cumulative effects of these actions made difficult to evaluate their regional consequences without a holistic perspective and an ecosystem-level analysis (Boesch et al. 1994, Sklar et al. 2001). Watershed simulation models have been used to evaluate the potential for success of diverse management strategies and the response of wetland habitats to increased effects of global warming, cumulative impacts and future human development (Reyes et al. 1994, Rybczyk 1998, Reyes et al. 2000). The present watershed model considered large-scale effects of large and punctual impacts, providing a regional view and evaluation, and allowing the exploration of future trends of ecosystem response.

Faced with continued land loss, as result of increased sea-level rise and subsidence, the State of Louisiana response has been the implementation of freshwater diversions. The Caernarvon watershed model represented an area where massive restoration efforts are in place. The objective for freshwater diversions was to compensate land loss by restoring the sediment contribution to the watersheds. Generally, these diversion impacts were projected to be beneficial to wetland health and sustainability, but it remains to be determined if there are threshold increases in freshwater input due to anthropomorphic alterations (diversion management scenarios) that might impact the system negatively (Day et al. 1995). Possible negative impacts included flood stress to the wetlands and potential algal blooms in water bodies. River diversions represent a viable mechanism for decreasing the nutrient load of river water prior to its reaching offshore waters and salinity intrusions (Weisner et al. 1994, Adler et al. 1996).

As with any large-scale tool, data availability and long-term time series limit the precision of both the watershed models. This was particularly evident in the Caernarvon watershed model as it treated the degree of separation between brackish and intermediate marshes by a unit of salinity. This sharp delineation has to be explored experimentally to a greater depth. The hydrodynamic module used a two-dimensional model on a fixed grid, thus limited in the vertical dimension and averaging values through large areas (0.25 km^2). However, the model predictions (Table 2) demonstrated the unique competence of the Caernarvon hydrodynamic module to replicate observed conditions continuously for long periods, well beyond the practical run length for any other two-dimensional coastal model.

Conclusions

A fully spatially articulated ecological model was developed for the Caernarvon watershed. Historical environmental conditions were used for the calibration of the Caernarvon Watershed Model. A comparison of 1988 habitat spatial patterns with the results of the model showed a fit index of 94.1.

Three different future scenarios were evaluated. A No Discharge, Schedule Discharge and Pulses Discharge water regimes were used to forecast habitat conditions in 50 years. Resulting yearly average salinity maps indicated similar overall conditions for all three scenarios. However, habitat distribution patterns varied substantially. As the different scenarios were computed it became apparent that freshwater has a beneficial effect on the marsh communities. This increased freshwater discharge translated in less land loss and a preservation of the original 1988 habitat diversity.

The present study serves as an example of the scientific tools available to resource managers for the evaluation of long-term cumulative impacts. As demonstrated in other areas of Louisiana, the use of spatially articulated models allows to identify gaps and problematic areas for the decision maker (Smith-Korfmacher 2001).

Acknowledgments

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Changes in Land Uses, Hydrology and Fish Habitats in an Urban Drainage, Cedar River, Washington

Robert C. Wissmar, Raymond K. Timm II

Abstract

This synthesis presents multi-scale approaches for evaluating influences of changing land cover characteristics (e.g., forest and impervious surfaces) on the composition of watershed landscapes, hydrological regimes, habitat restoration and habitats preferred by salmon. Spatially explicit modeling of changes in land covers and flood discharge regimes for urban and rural watersheds are compared for two periods, historical to 1991 and from 1991 to 1998. For the historical to 1991 period, impervious surfaces increase in both urban (range +43% to +71%) and rural (range +8% to +15%) watersheds while forest covers decline (range from –63% to -83% in urban and from -28% to -34% in rural areas). For the 1991 to 1998 period, impervious areas also show increases, ranging from +4% to +27% in urban and from +38% to +60% in rural watersheds. Land cover changes in urban areas are caused by infilling and continued development in an already urban matrix. In contrast, rural areas lost forests through rapid land conversions characterized by scattered low-density residential, clustered dense commercial, residential developments and increases in transportation facilities near newly incorporated areas. Hydrologic simulations indicate annual flood frequencies increase in all watersheds in response to increases in impervious surfaces and declines in forests. For the historical to 1991 period, flood discharges range from +68% to +169% in urban and from +7% to +21% in rural areas. During 1991 to 1998, smaller percent changes in discharge occur for all watersheds (range from -5% to +17%). Comparison of water yields (discharge per unit area, m yr^{-1}) for watersheds as functions of different impervious and forest land covers (percent of watershed area) indicate two phases of abrupt changes for water yields. The first shows sharp increases in yields when impervious surfaces are between 10% and 23% and forest covers are between 59% and 81%. The second phase shows higher yields that coincide with larger areas of impervious surfaces (between 46% to 74%) and lower forest covers (between 17% and 37%). These relationships indicate that our characterizations of impervious surfaces and forested covers and their use in a spatially explicit hydrology model provides a potent approach for revealing how variations in spatial distributions of different land covers affect stream discharge rates and "thresholds" of water yields. Subsequent land cover evaluations using a multi-scale habitat model identify priority river reaches and floodplain habitats for restoration and conservation. Large patches of positive indices indicate the most favorable habitats are characterized by low fragmentation, greater connectivity and availability to salmon. Factors commonly preferred by spawning salmon (*Oncorhynchus nerka*) are upwellings of subsurface waters, moderate water depths, and gravel/cobble sized substrates.

Keywords: multi-scale, spatial modeling, watershed, hydrology, land cover, habitat, salmon

Introduction

This paper presents a synthesis of multi-scale watershed and ecological modeling studies that evaluate influences of changing land cover characteristics (e.g., forest and impervious surfaces) on the composition of watershed landscapes, hydrological regimes, habitat restoration (e.g., riparian and stream) and habitats preferred by salmon. The study areas include channels, floodplains and tributary watersheds of the lower Cedar River drainage near Seattle, WA (Figure 1). The objectives include: a) evaluating changes in land covers between historical (full forest cover, pre-20th century), 1991 and 1998 conditions; b) determining effects of changing land

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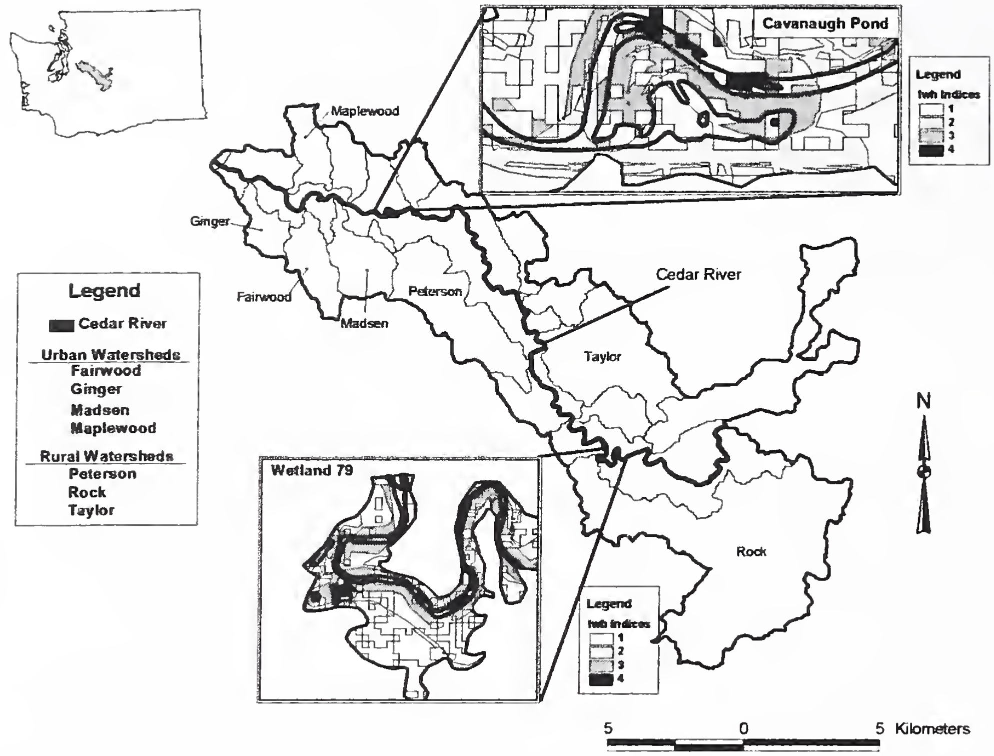

Figure 1. Select urban and rural tributary watersheds of the lower Cedar River, and river reaches and habitats prioritized for restoration and conservation. The urban (Ginger, Maplewood, Fairwood and Madsen Creeks) and rural watersheds (Peterson, Taylor and Rock Creeks) were analyzed to determine changes in land covers between historical ("full forest cover"), 1991 and 1998 conditions and to assess effects of land-uses on hydrological regimes. Spatial distributions and characteristics of positive patch indices (I_{wh}) indicate that Wetland 79 and Cavanaugh Pond are priority floodplain habitats. The Cedar River drains into Lake Washington near Seattle, WA.

uses on hydrological regimes of tributary watersheds; c) using landscape indicators of spatial compositions of co-occurring "natural systems" and human developments to identify opportunities for habitat restoration and conservation; and d) identifying habitat preferences of salmon in order to improve habitat restoration initiatives that facilitate fish recovery. The goal of this research is to couple multi-scale watershed and ecological approaches that can be used by watershed managers. In many coastal areas of North America changes in land-uses and hydrology, and ESA listings of salmon and other species, requires improvements in watershed management efforts designed to conserve and restore environments of declining species.

Methods

A summary for the modeling approaches includes status (newly developed or modified), authors, model type, grid cell size (resolution), spatial extent (km^2, ha, m^2) and simulation time (Table 1). Key data sources and files (forested areas, impervious surfaces and other land covers), that facilitate cross-scale integration and modeling of changes in land covers and hydrology, are developed using classifications of Landsat TM scenes and extensive empirical measurements of landscape patches (Burgan et al. 1993, Logsdon et al. in review). These extensively ground-truthed data sets are inputs for the spatial assessment (Wissmar et al. 2000) of changes in land covers and the application of a spatially explicit hydrology model (Wigmosta et al. 1994). This model evaluates impacts of changing land covers on hydrological regimes of urban and rural tributary watersheds (Wissmar et al. in review). Spatially explicit modeling of changes in land covers and flood discharge regimes for urban and rural watersheds are compared for two periods, historical to 1991 and from 1991 to 1998.

Land cover evaluations using the multi-scale habitat model (Timm et al. in press) that applies different land cover files is used to prioritize habitats for restoration and conservation. The land cover files include four

Table 1. Model characteristics for watershed research in the Cedar River drainage near Seattle, WA. Model descriptions include status N (new) or M (modified), citations, type, grid size, spatial extent and simulation time.

Model	Citations	Model type	Grid size (resolution)	Spatial extent	Simulation time
Changes in land-uses (N)	Wissmar et al. (2000)	Spatial Landscape	30 m	Watershed (km^2)	Decadal
Classifying land covers (M)	Logsdon et al. (in review)	NDVI[a]	30 m	Watershed (km^2)	Decadal
Affects of land-uses on hydrology (M)	Wissmar et al. (in review)	DHSVM[b]	30 m	Watershed (km^2)	Decadal
Prioritizing restoration areas (N)	Timm et al. (in press)	Spatial Landscape	5 m	Watershed (km^2), reach/ habitat (ha, m^2)	Annual, decadal
Affects of Habitat Factors on fish (N)	Hall et al. (2000) Hall (2002)	Multiple logistic Regression (statistical)	1 m	Habitat (ha, m^2)	Daily, monthly

[a] NDVI classification: Normalized Difference Vegetation Index (Burgan 1993)
[b] DHSVM: Distributed Hydrology Soil Vegetation Model (Wigmosta et al. 1994)

habitat factors (i.e., forest canopy, wetlands, channel configurations and gravel sources) and four anthropogenic factors (i.e., impervious surfaces, real estate values, land zoning and physical channel constraints). Model outputs include habitat indices (I_{wh}) and composition and configuration metrics of habitat indices at the valley floor and reach scales. Homogeneous patches as indices (I_{wh}) of potential sites are measured in terms of their locations, sizes, and relative degree of fragmentation. These patch indices are further analyzed to characterize the intra-patch heterogeneity for combinations of habitat and anthropogenic factors for each grid cell. Model application requires spatial weighting of indices to prioritize riparian zones along salmon bearing streams.

Two priority restoration areas (Wetland 79 and Cavanaugh Pond) are further analyzed to determine habitat preferences of spawning sockeye salmon (*Oncorhynchus nerka*) (Hall et al. 2000, Hall 2002). Logistic regression and electivity models are used to determine preferred habitats of spawning fish. Ground-truthed GIS data sets of aquatic-riparian landscapes and empirical measures of habitats factors (water depth, substrate, detrital depth, subsurface water flow, water temperature, and cover) provided inputs for these models.

Results

Spatial modeling of land cover changes between historical and 1991 and from 1991 to 1998 indicate losses of forest covers and increases in impervious surfaces. For the historical to 1991 period, impervious surfaces increase in both urban (range +43% to +71%) and rural (range +8% to +15%) watersheds while forest covers decline (range from –63% to -83% in urban and from -28% to -34% in rural areas). For the 1991 to 1998 period, impervious areas also increased, ranging from +4% to +27% in urban and from +38% to +60% in rural watersheds (Table 2).

Table 2. Percent change in forest covers, impervious surfaces and annual flood discharges from historical to 1991 and 1991 to 1998 within urban* and rural watersheds of the lower Cedar River. Percent changes for periods are summarized as plus or minus Δ%. Historical conditions (Hist.) assume fully forested cover. See Figure 1 for locations of watersheds. Flood discharges are based on the 10-year recurrence interval.

	Land covers				Annual flood discharge	
	Hist. to 1991		1991 to 1998		Hist. to 1991	1991 to 1998
	Forest	Impervious	Forest	Impervious	Discharge	
Watershed	--- (Δ%) ---		--- (Δ%) ---		--- (Δ%) ---	
Ginger*	-83	+71	+18	+4	+169	+4
Maplewood*	-63	+48	-24	+27	+96	+17
Fairwood*	-63	+46	-3	+17	+84	+3
Madsen*	-63	+43	+19	+9	+68	-2
Peterson	-34	+15	-11	+53	+16	+2
Taylor	-28	+10	-4	+60	+21	+1
Rock	-29	+ 8	+14	+38	+7	-5

Simulations using the spatially explicit hydrology model, where model functions are sensitive to changes in land covers, show increases in flood discharges in urban and rural watersheds. Annual flood frequencies increase in all watersheds in response to increases in impervious surfaces and declines in forests. For the historical to 1991 period, flood discharges ranged from +68% to +169% in urban and from +7% to +21% in rural areas. From 1991 to 1998, smaller percent changes in discharge occur for all watersheds (range from -5% to +17%) (Table 2).

Comparisons of water yields among watersheds are made as a function of different amounts of impervious and forest land covers (percent of watershed area) in the respective watersheds. Water yields, discharge per unit area (m yr^{-1}) for 10 and 25-year recurrences, indicated two distinct phases for abrupt changes in discharge levels during 1991 and 1998. The first phase showed sharp increases in discharge (range from 3.7 to 6.3 m yr^{-1}) when impervious surfaces are between 10% and 23% and forest covers are between 59% and 81%. The second phase occurred at higher discharges (4.1 to 8.7 m yr^{-1}) and coincided with larger areas of impervious surfaces (between 46% to 74%) and lower forest covers (between 17% and 37%).

Land cover evaluations using the multi-scale habitat model demonstrates that patterns and sizes of patch indices can quantify the spatial complexity of habitats within riparian areas and can prioritize habitats for restoration and conservation. Simulations show reaches characterized by high positive indices (I_{wh}) and large patch sizes represent intact-high quality habitats. These habitats are the most favorable for restoration and conservation. Lower positive and negative scores that are influenced by anthropogenic factors are coincident with higher degrees of habitat fragmentation. These spatial configurations show less favorable conditions for restoration and conservation.

Spatial distributions of positive indices and their patch characteristics indicate that specific river reaches and floodplain areas contain high positive indices and relatively low fragmentation. The larger mean patch sizes pointed to less fragmentation and greater connectivity between habitats. Two floodplain sites containing prime examples of priority habitats (high positive indices) include Wetland 79 and Cavanaugh Pond (Figure 1). These floodplain sites and habitats are connected to the main channel of the lower Cedar River by outlet channels.

The two floodplain sites, Wetland 79 and Cavanaugh Pond, are further analyzed to determine habitat preferences of spawning sockeye salmon (Figure 1). The objectives include: a) identifying habitat factors most important to fish selection of redd sites ("egg deposition sites"); and b) using this information to improve habitat restoration initiatives required to facilitate fish reproduction and survival. The necessity for this information relates to the diverse types of habitats required by various fish species and their life history stages. The most frequent habitat factors associated with

Table 3. Summary of habitat preferences of spawning sockeye salmon (*Oncorhynchus nerka*) in two floodplain areas. Floodplain sites include Wetland 79 and Cavanaugh Pond that connect with the main channel of the lower Cedar River near Seattle, Washington. Preferences for habitat types are defined by electivity indices (Hall et al. 2000). The total number of redds at each site are indicated in parenthesis (n). Habitats types indicate major bottom substrate, fluvial and shoreline characteristics (e.g., vegetative cover). Electivity indices (D) calculated as: D = r-p/(r+p)-2rp, where p is the proportion of the habitat available and r is the proportion of the habitat used for redd placement. Electivity values range from +1 (strong selection) to –1 (strong avoidance). Spawning sockeye showed the strongest selection where subsurface waters were upwelling.

Floodplain site	Habitat type	Proportion available (p)	Proportion of redds (r)	Electivity (D)
Wetland 70 (20)	Upwelling	0.13	0.86	+0.95
	Shrub riparian	0.10	0.00	-1.00
	Steep forested	0.22	0.05	-0.71
	Open water	0.55	0.09	-0.85
Cavanaugh Pond (240)	Upwelling	0.11	0.96	+0.99
	Marsh	0.07	0.00	-1.00
	Gravel	0.02	0.00	-1.00
	Island	0.05	0.00	-1.00
	Outlet	0.05	0.04	-0.10
	Open water	0.70	0.00	-1.00

the placement of redds include upwelling of subsurface waters, moderate water depths (10-80 cm), and gravel/cobble substrates (Table 3). Upwelling is the most important factor (electivity indices, +0.95 and +0.99). Relationships between different habitat factors (e.g. upwelling and water temperature, water depth and fine sediments) also influence habitat choices. Favorable intra-gravel flow in redds supplied by upwellings appear to compensate for effects of other sub-optimal habitat attributes. Fish avoid silt and areas with substantial detrital substrates. Fish spawning densities varies between years (1999 and 2000) and appears to affect preferred water depth ranges in both ponds. The identification of habitat factors most important to salmon spawning success in off-channel areas provides a strong ecological basis for improving the design, implementation and evaluation of restoration activities within off-channel and floodplain areas of large river systems.

Conclusions

Our multi-scale approaches demonstrate how analyses using watershed and ecological models can facilitate assessments of influences of land covers on hydrological regimes of watersheds, habitat conditions of riparian and stream ecosystems and habitats selected by fish. Spatial modeling of changes in land covers show losses of forest covers and increases in impervious surfaces. Land cover changes in urban areas are caused by infilling and continued development in an already urban matrix. In contrast, rural areas lost forests through rapid land conversions characterized by scattered low-density residential, clustered dense commercial, residential developments and increases in transportation facilities near newly incorporated areas (Wissmar et al. 2000). Hydrologic simulations indicate annual flood frequencies increase in all watersheds in response to declining forests and increases in impervious surfaces. Flood frequencies within urban watersheds are several times greater than in rural watersheds. Furthermore, comparisons of water yields (m yr^{-1}) for watersheds as functions of different amounts of impervious and forest covers show two distinct phases for abruptly increasing water yields. These relationships indicate that our characterizations of impervious surfaces and forested covers (Wissmar et al. 2000, Logsdon et al. in review), and their use in a spatially explicit hydrology model (Wigmosta et al. 1994, Wissmar et al. in review), provide robust approaches for revealing how variations in spatial distributions of different land covers affect stream discharge rates and "thresholds" of water yields. In summary, both spatial patterns and extent of different land covers influence model dynamics.

Subsequent land cover evaluations using the habitat model show patterns of patch indices that can be used to prioritize habitats for restoration and conservation. Large patch sizes and positive indices indicate the most favorable habitats are those with relatively low degrees of fragmentation, greater habitat connectivity and availability to salmon (Timm et al. 2003). Some important areas include floodplain channels and ponds that contain habitats preferred by salmon. Further analysis using logistic regression and electivity indices (Hall et al. 2000, Hall 2002) indicate that spawning salmon select specific habitat factors (e.g., upwelling waters and gravel substrates). Our ongoing studies are evaluating other models (Timm et al. in preparation, Wissmar et al. in review) that can be used in multi-scale approaches for improving watershed and habitat management and protecting human interests.

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Probabilistic Assessment of Wadeable Streams in the Southeastern U.S.

Peter Kalla, James Maudsley

Abstract

The Southeastern Wadeable Streams R-EMAP project applied probability sampling and analysis to a regional study of wadeable streams. The goal of the project was to determine, at +10% precision with 90% accuracy, the percent of U.S. EPA Region 4 stream miles that were subnominal in terms of habitat, ecological integrity, and trophic state. More than 200 random and 30 reference stations were sampled over four years. Interim results for several parameters are presented, including water column physico-chemistry and nutrients, RBP habitat score, benthic macroinvertebrate indices, and whole-body total mercury in forage fish. After three years the median concentration of mercury was 0.088 mg/kg (wet weight), with 43 + 13 % of Regional stream miles having a concentration that poses an ecological risk to predators (>0.100 mg/kg). Higher values occurred in the Southeastern Plains compared to other ecoregions. Methylation rates are probably higher in the more anaerobic sediments of warm, slowly flowing coastal plain streams. Multi-scale pilot studies are underway to explore the feasibility of using remote sensing and atmospheric measurements to relate watershed, riparian, and channel conditions to in-stream metrics.

Keywords: mercury, fish, regional, bio-assessment

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Small Stream Ecosystem Variability in the Sierra Nevada of California

Carolyn T. Hunsaker, Sean M. Eagan

Abstract

The quality of aquatic and riparian ecosystems is a function of their condition and the integrity of adjacent uplands in their watersheds. While small streams make up a large proportion of the overall stream network, our knowledge of how they function is still limited. The Kings River Experimental Watershed (KREW) was initiated in 2000 to quantify the variability in characteristics of small stream ecosystems and their associated watersheds in the Sierra Nevada of California. The primary management questions to be answered are the effects of prescribed fire and mechanical thinning on the riparian and stream physical, chemical, and biological conditions.

Two mixed conifer sites are being developed. Data will be gathered for at least a 3-year reference period. After fire and harvest treatments are applied, data will be gathered for at least seven years. Each site has a control watershed that receives no treatments, a watershed that is burned, a watershed that is harvested, and a watershed that is both burned and harvested. We are interested in assessing the integrated condition of the streams and their associated riparian and watershed areas (i.e., physical, chemical, and biological characteristics). The watersheds range in size from 49 to 228 ha (120 to 562 acres); a size that can be consistently treated.

Keywords: stream ecosystem, watershed experiment, prescribed fire, mechanical thinning, sustainable forests

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Introduction

Sixty percent of California's water originates from small streams in the Sierra Nevada, yet very little information is known about how these streams are affected at the source. This water is considered some of the highest quality water in the state. The quality of aquatic and riparian (near-stream) ecosystems associated with streams is directly related to the condition of adjacent uplands within their watersheds. The degradation of forest streams and their associated watersheds is often the result of nonpoint sources such as past timber harvesting, roads, fire suppression, and catastrophic wildfires. Restoration of the Sierra Nevada's forest watersheds to historic or desired conditions requires active management such as reintroduction of frequent, cool fires and removal of accumulated fuel loads.

The Kings River Experimental Watershed (KREW) is a long-term watershed research study being designed and implemented on the Sierra National Forest to provide much needed information for forest management plans regarding water quantity and quality. This experimental watershed research is designed to: (1) quantify the variability in characteristics of headwater stream ecosystems and their associated watersheds, and (2) evaluate the effect of fire and fuel-reduction treatments on the riparian and stream physical, chemical, and biological conditions. This is an integrated ecosystem project at the watershed scale and is part of a larger adaptive management study that began in 1994 as a collaborative effort between the Sierra National Forest, Southern California Edison, and the Pacific Southwest Research Station of the Forest Service to evaluate the effects of approaches for creating an uneven-aged forest similar to that present before European settlement, circa 1850. The experiment will implement mechanical thinning, prescribed fire, and thinning with fire combination treatments on headwater areas.

"Aquatic/riparian systems are the most altered and impaired habitats of the Sierra" (University of California, Davis 1996). However, what is considered appropriate management for such ecosystems is currently a point of debate and quantitative information is needed (USDA Forest Service 2001). While watershed research and stream monitoring have been ongoing, much of the work addresses larger streams. Much of the information on effects from forest management activities likely is not applicable for Sierran forests because it comes from wetter forests and severe treatments (e.g., clearcuts and wildfire). Also, few integrated ecosystem studies exist; these are essential for understanding stream/watershed ecosystem processes and functions for adaptive management, thus critical information is lacking (Naiman and Bilby 1998).

Methods

The KREW recognizes that measurements of physical, chemical, and biological variables are necessary to accomplish a holistic watershed study and has carefully tried to include those attributes critical to detect change in both patterns and processes in atmospheric inputs, watershed uplands and riparian areas, and stream channels. Human actions jeopardize the biological integrity of water resources by altering one or more of five principal factors: physical habitat, seasonal flow of water, the food base of the system, interactions within the stream biota, and chemical contamination (Karr 1998). We are taking measurements for each of these factors.

The experimental watershed is planned as a 15-year study and has two sites, the Providence Creek Site and the Bull Creek Site, located in mixed-conifer forest between 1,500 (5,000 ft) and 2,134 m (7,000 ft) elevation on granitic-based soils. As such, these sites are very typical of the central Sierra Nevada and forested headwaters that provide a substantial amount of source water to the San Joaquin River basin of California. The Providence Site typically experiences rain and snow events while the higher elevation Bull Site is a snow-dominated location. Each site consists of four watersheds, one for each of the three management treatments and one control. Core field measurements on each watershed comprise the following components: stream discharge, water chemistry, sediment loading, stream invertebrates, soil characterization, meteorology, vegetation, and fuel loading. Instrumentation began in 2000, baseline data collection (3 years minimum) began in 2002, the first treatments should occur in the fall of 2005, and post-treatment data collection is planned to continue for 5-7 years. A site consists of four adjacent headwater watersheds; the Providence Site is made up of P301, P303, P304 and D102 (Figure 1). A grid with 150-m (492-ft) spacing has been placed within each watershed; for the small size of P304 the grid was densified in the north-south directions with a 75-m (246-ft) spacing. Sampling for physical soil characteristics, upland vegetation, and fuel loads are all located using this grid. All measurements are made in the same manner at both sites, but the Providence Site has evolved to have more types of measurements (i.e., sediment ponds in the streams, sediment fences to quantify upland erosion, vacuum lysimeter collectors for shallow soil water, and riparian microclimate).

Stream discharge is measured using two fiberglass Parshall-Montana flumes in each stream, a large and a small, because the streams have approximately a 500-fold difference between lowest and highest flow for a 20-year time span. We can measure precisely flows from 0.75 l/s (0.03 cfs) to 900 l/s (32 cfs) and with less precision flows from 0.3 l/s (0.01 cfs) to 1,400l/s (49 cfs). These flumes are good at passing debris and do not require large upstream ponds to accurately measure flow. Stage is measured with the Isco 730 bubbler; Sequoia Scientific Aquarods provide backup data for stage measurements.

Measuring the seasonal variation in air temperature, solar radiation, and precipitation is considered basic to all hydrologic and natural resource studies (Hanson et al. 2001). Each site has a high elevation and a low elevation meteorological station with a 6 m (20 ft) tower, a precipitation gage, and a telemetry antenna. The high elevation stations also have a snow pillow to measure snow water equivalence. Each tower has instruments to measure relative humidity, temperature, wind speed, wind direction, wind run, solar radiation, and snow depth.

Universal physical laws govern streams, yet every stream exists in a unique way within its watershed. Differences in watershed size, climate, location, geology, and past management activities are only a few of the factors that create the range of fluvial forms we see. Each stream must balance erosion, transport and deposition in the context of these factors. To understand the effects of a given management practice, a baseline of existing physical conditions must be established for the stream channel. With this foundation of technically correct and comparable data, it is possible to track changes in the character of the stream (Harrelson et al. 1994). A 100-m (328-ft) reach, often just upstream of the flumes, is being

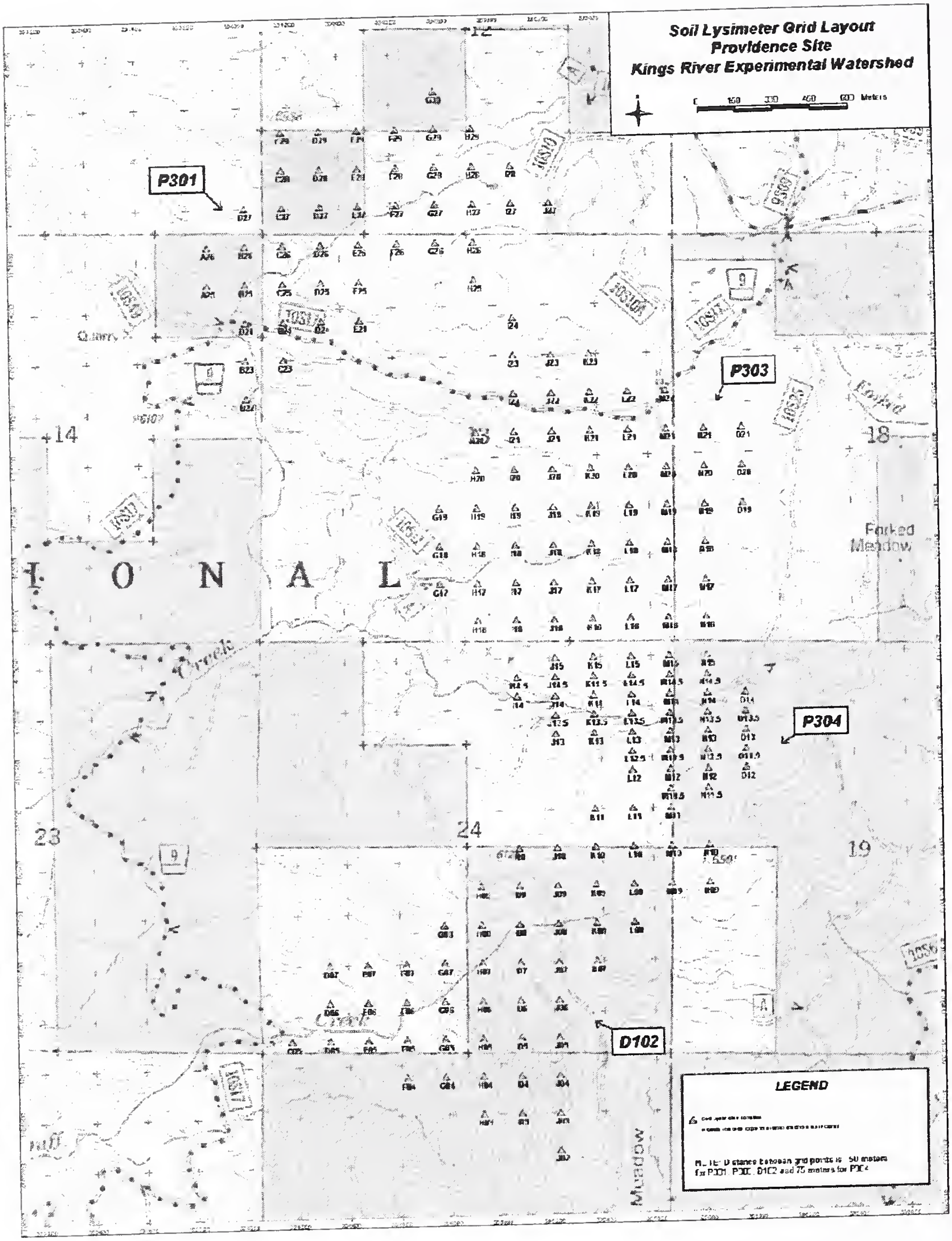

Figure 1. The Providence Site has four adjacent headwater watersheds (P301, P303, P304, and D102). The sampling grid is displayed against the landscape's topography and is used to locate resin lysimeters, vegetation, fuel loading, and physical soil characteristic measurements.

used as a representative reach for stream invertebrate and geomorphology measurements. These measurements are taken annually. Permanent cross sections are planned.

Natural deposition areas that existed directly downstream of the flumes were transformed into sediment ponds with 12 to 40 m^3 (423 to 1,412 ft^3) of storage space. We will be estimating the annual load for each sediment basin, the relative percentage of organic versus inorganic material in that load, and the class size distribution within the inorganic fraction. Upland sediment fences will provide information on the proportion of sediment coming from different upland sources such as roads and other erosional areas. Some existing headcuts are also being characterized and monitored.

Soil physical properties profoundly influence the growth and distribution of vegetation through their effects on soil moisture regimes, aeration, temperature profiles, soil chemistry, and even the accumulation of organic matter. These properties also influence erosion potential and the chemical composition of water that reaches streams. In general, the dominant soil type by watershed accounts for 50% or more of the area: Shaver and Gerle-Cagwin soils dominate the Providence Site, and Cagwin soil dominates the Bull Site. Soil sampling begins in 2003 and will be colocated with vegetation and fuels loading measurements.

Within a watershed the chemical composition of stream water serves as an integrator or expression of the condition of the watershed both in the uplands and in the stream. Our goal is to measure water chemistry in several parts of the hydrologic cycle—incoming precipitation, in shallow soil, and in the stream. Currently KREW is measuring several anions and cations; however, after one water year we will evaluate the need to continue this entire set of measurements. In general, water chemistry measurements are taken every two weeks although during or after a storm event samples may be more frequent. Stream water samples are either collected as a grab sample by a person or by an Isco 6712 automated sampler that draws water from the stream and stores up to 24 un-refrigerated samples before retrieval.

We are using two types of lysimeters to characterize the nutrient fluxes and chemistry of shallow soil water. The soil resin lysimeters are placed on a uniform grid spaced at one depth, 13 cm (5 in), to measure the annual flux of nutrients through the forest litter and soil layers above them. These lysimeters provide information about soil nutrient flux. The Prenart vacuum lysimeter provides a continuous measurement of soil water chemistry at one location from depths of 13 cm (5 in) and 26 cm (10 in). The vacuum lysimeter data provides information on the variability of chemistry fluxes during the wet season.

Similar to soil water, we have two types of collection devices for atmospheric wet chemistry. The snowmelt collectors give an estimate of the variability in the precipitation chemistry during the wet season. The precipitation resin collectors give an annual measure of the total input from rain and snow during the period they are in the field. Soil lysimeters and precipitation collectors are placed in the field at the same time; they are built according to a design published by Susfolk and Johnson (2002).

The major objective of our vegetation research is to examine the treatment effects on the vegetation within the watersheds. A secondary objective is to determine effects on riparian vegetation in particular, and characterize the transition from riparian to upland vegetation. Riparian transects are placed perpendicular to the stream channel, starting at the bankfull edge, and extending 20 m (66 ft) into the upland. Upland transects are placed at a subset of the grid points extending 20 m at a randomly chosen azimuth. The number of riparian and upland transects varies depending on the length of the channel and size of the watershed, respectively. At each transect herbaceous vegetation will be sampled in 1x1 m quadrats, shrubby vegetation by line intercept, and trees by a 10x20 m belt transect. Riparian and upland transects follow an identical protocol.

Ground, surface, understory, and overstory fuels will be measured at the same subset of grid points as vegetation and soils. The protocol for these measurements has to be defined; however, methods for the fuel components will closely match those of the Sequoia National Park site of the Fire/Fire Surrogate study.

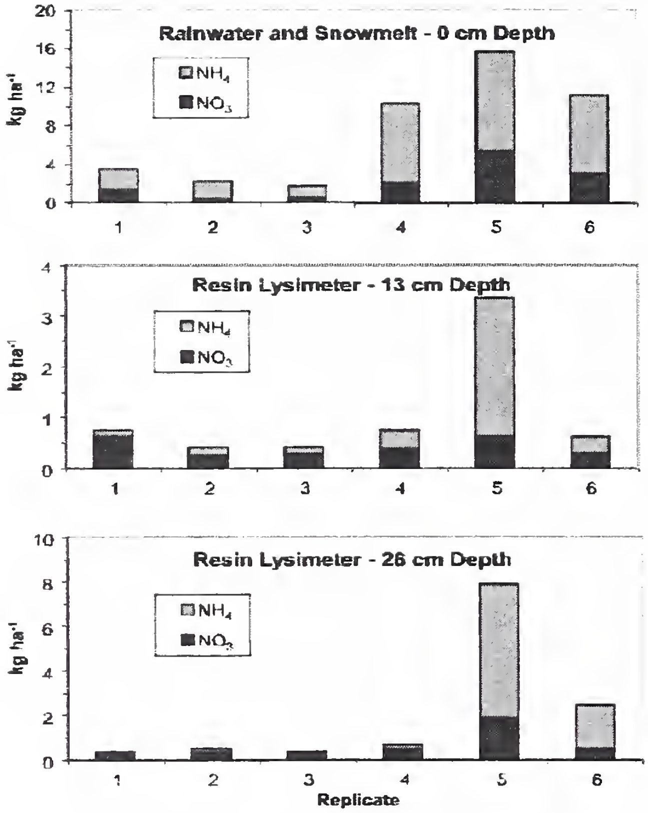

Figure 2. Ammonium and nitrate fluxes are measured with resin lysimeters; replicate 5 is high in both snowmelt and in both measured soil horizons. Preliminary sampling at Providence 303 revealed evidence that there are "hotspots" of high nitrogen availability within a fine spatial scale (meters).

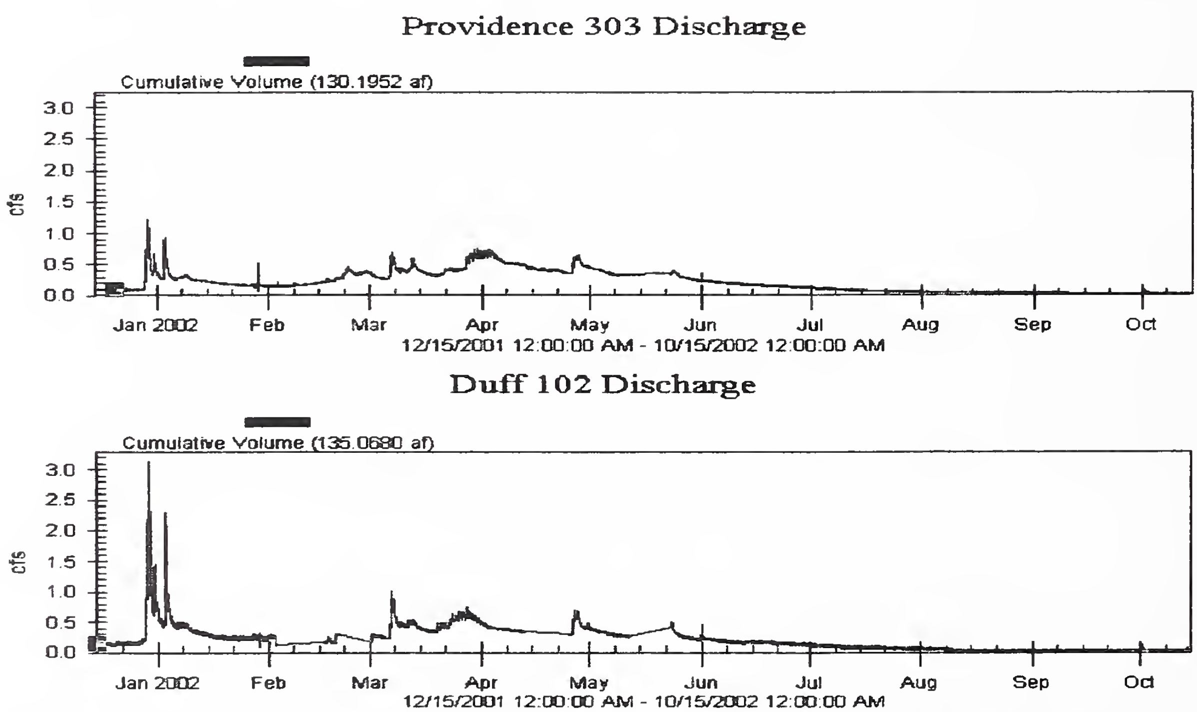

Figure 3. Stream discharge hydrographs for Providence 303 and Duff 102 streams. The base flow of these streams is between 0.05 and 0.1 cubic feet per second (cfs) and is controlled by groundwater sources. Duff has the lowest average elevation, and the January storms came mostly as rain that caused brief, large discharges.

Results and Conclusions

Preliminary sampling at Providence 303 revealed evidence that there are "hotspots" of high nitrogen availability within a fine spatial scale (Figure 2); replicate 5 is high for both incoming precipitation and both soil depths. The causes of this hotspot are not known at present, but the knowledge that such hotspots occur is important in guiding sampling plans and in assessing potential sources of nitrogen for the streams.

One of the challenges of a landscape-scale experiment is the similarity between study units, in this case, headwater watersheds. Stream discharge hydrographs for P303 and D102 during 2002 are shown in Figure 3. While the discharges differ for the two streams, the timing of peaks follows a similar pattern. For January, these hydrographs also illustrate the difference between discharge patterns when one stream receives snow (P303) and another receives rain (D102).

The intention of KREW is to be as holistic and integrated as possible with a focus on physical, chemical, and biological variables of headwater stream ecosystems and their associated watersheds. Much needed information for both basic and applied science questions will be developed for the southern Sierra Nevada and sustainable forestry in general. Attention has been given to developing a suite of measurements that will facilitate modeling in several disciplines: hydrology, meteorology and climate change, fire behavior and effects, soil erosion, and biogeochemistry. An exciting opportunity will be to calibrate and run models in the pretreatment phase and then verify their predictive capabilities after the management treatments.

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An Interregional Comparison of Channel Structure and Transient Storage in Streams Draining Harvested

Brian H. Hill, Frank H. McCormick

Abstract

We compared measures of channel structure and riparian canopy with estimates of transient storage in 32 streams draining old-growth and harvested watersheds in the Southern Appalachian Mountains of North Carolina (n = 4), the Ouachita Mountains of Arkansas (n = 5), the Cascade Mountains of Oregon (n = 8) and the redwood forests of northwestern California (n = 15). Channel cross-sections and riparian canopy were measured at 10 equally spaced intervals along each 50-m to 100-m study reach. Stream depth was measured at 1-m intervals along the study reaches, and stream substrate composition was visually estimated at 50 points within each reach. While there were distinct stream differences between the geographic regions, there were consistent trends among these regions when comparing measurements of channel structure and riparian canopy in streams draining old growth and harvested watersheds. No significant differences were found in stream width, depth, or streambed area, but streams draining harvested watersheds had more open riparian canopies and smaller median substrate sizes. Transient storage (As) was calculated as differences in area under the curves for predicted and actual Cl- transport through the study reaches. Total reach volume was estimated as the sum of transient storage and surface water volume estimated from channel cross-sections. The percentage of reach volume attributable to transient storage was significantly smaller in streams draining harvested watersheds, and was negatively correlated with median particle size, and positively correlated with riparian canopy cover and stream depth.

Keywords: transient storage, physical habitat

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Interregional Comparison of Nutrient Uptake Rates in Managed and Old-Growth Watersheds

Frank H. McCormick, Brian H. Hill

Abstract

We compared nutrient uptake rates to examine the effect of timber harvest on streams. From 1999-2002, nutrient additions were conducted in 50 stream reaches in 4 ecoregions (southern Blue Ridge, NC, Ouachita Mountains, AR, Cascade Mountains, OR, and the redwood forests of the Coast Range, CA). Nutrient uptake (NH4+1, PO4-3) was measured, along with the Cl- tracer, by depletion over stream distance. Streams draining logged watersheds had smaller dominant substrate size, more open canopies, and more sand and fine sediments in the channel. Phosphate (P) uptake lengths were not significantly different when comparing streams draining old-growth or harvested watersheds or ecoregions. Ammonium (N) uptake lengths were significantly longer in old-growth compared to harvested watersheds but were not different among ecoregions.

Keywords: nutrient uptake rates, interregional comparisons, harvest regimes

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First Interagency Conference on Research in the Watersheds

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Integrating Science with Watershed Decision Making II

An Internet-based Spatial Decision Support System for Rangeland Watershed Management

Ryan Miller, D. Phillip Guertin, Philip Heilman

Abstract

The impact of livestock grazing on water quality, especially erosion and sedimentation, is an important concern in the southwestern United States. In response to Federal and State regulations, Best Management Practices (BMPs) for rangeland management are being developed and implemented in many western states, although the efficacy and economic impact of many practices have not been examined. To assess the potential effectiveness of BMPs, a Spatial Decision Support System (SDSS) has been designed to integrate water quality, livestock management and economic concerns. The SDSS was developed through the integration of hydrologic, erosion, livestock management and economic simulation models linked with a geographic information system and database management system. The SDSS can help managers select the type and location of BMPs based on site-specific data and is deployed via the Internet providing access through a web browser. The SDSS provides land managers with a means to identify critical areas causing water quality degradation and design and implement watershed management practices to improve water quality. The poster will describe the SDSS and provide case study examples of its application.

Keywords: GIS, hydrologic modeling, non-point source pollution

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Introduction

Traditional ranching communities in the western United States are under increasing stress. Global competition, development, and changing environmental perceptions are altering the face of rural America. This is especially true in the rural Southwest where ranching has historically been a primary economic activity. The long-term viability of ranching in the southwestern United States is questionable given decreasing beef consumption, increasing urbanization pressures, and negative public perceptions of livestock grazing.

An important issue that impacts rangeland management and livestock grazing is land and water quality, especially erosion and sedimentation. In Arizona, sediment is a principal non-point source pollutant with almost 960 miles of stream channels polluted with sediment, which is over three times greater than impairment caused by the next leading constituent (U.S. Environmental Protection Agency 1998). The advent of Total Maximum Daily Loads (TMDLs) by the U.S. Environmental Protection Agency establishes criteria to evaluate pollutant contribution from land-use impacts and other non-point sources. Under section 303(d) of the Clean Water Act, TMDLs are designated as a tool for watershed-based water quality management decisions; however, problems with implementation of these watershed management programs are due to the inadequacy of procedures, models, and methodologies caused by insufficient focus on watershed-wide diffuse source properties (Novotny 1999). Since management of non-point source pollutants and TMDLs is inherently spatial, distributed hydrologic and water quality simulation models coupled with geographic information systems can be used to simulate the impact of various land-use conditions on water quality.

To complicate the issue, many livestock growers manage ranches composed of a mixture of federal, state, and private land, all with different goals and regulations. A rancher operator may face a multitude of

BMPs on a single livestock allotment that overlaps multiple land ownerships. A single livestock allotment may cross several watershed boundaries; conversely, several livestock allotments managed by different ranchers may lie within a single watershed. Since TMDLs are assigned at the watershed level, integrated impact of management actions across different ranch allotments and land ownership must be assessed (U.S. Environmental Protection Agency 1999).

Ranch planning has traditionally been done at the allotment or pasture level. For example, ranch management plans developed by the USDA Natural Resources Conservation Service (NRCS) primarily address erosion and erosion control within a ranch in concert with other watershed plans, but seldom explicitly assess the potential impacts beyond the allotment boundary. Although numerous hydrologic/erosion/water quality models have been developed (National Resource Council Committee on Watershed Management (NRC) 1999), none are currently capable of supporting allotment planning and the implementation of different BMPs across a watershed.

Effective watershed management and planning requires the integration of knowledge, data, simulation models, and expert judgment to solve practical problems and provide a scientific basis for decision making at the watershed scale (NRC 1999). A user-friendly decision support system (DSS) that would help different stakeholder groups to develop, understand and evaluate alternative watershed management strategies is needed. The DSS would consist of a suite of computer programs with components consisting of database management systems (DBMS), geographic information systems (GIS), simulation models, decision models, and easy to understand user interfaces.

The difficulty in developing the DSS is not a lack of available simulation models but rather making these models available to decision makers, a key observation made by the National Resource Council's Committee on Watershed Management (1999). Over the last forty years the federal government has spent millions of dollars on model development. The USDA-Agricultural Research Service (ARS) currently supports many simulation models addressing various environmental concerns from erosion to exotic species. While these simulation models are used extensively in research settings, they are infrequently incorporated into the decision making process. Reasons for this exclusion include: data requirements are usually only attained in a research setting; models are complex and underlying assumptions are poorly understood by resource managers; and deriving model input parameters is extremely time consuming and difficult. These models represent a valuable library of knowledge that should be utilized. To use these models, expertise in database management system, geographic information systems, computer operating systems, remote sensing and Internet searching for data gathering, graphics, as well as watershed domain knowledge is required. Few seasoned professionals have all these skills, much less the typical watershed stakeholder.

The poster will review the development of an operational GIS-based, integrated watershed-planning tool deployed via the Internet that provides land managers with the information necessary to evaluate the effects of livestock grazing impacts on water quality. The tool will be capable of evaluating available BMPs that can be implemented to mitigate detrimental impacts and assessing economic ramifications of management decisions.

Internet-based SDSS

The Automated Geospatial Watershed Assessment (AGWA) (Miller et al. 2002a) application was developed by the University of Arizona and USDA-ARS Southwest Watershed Research Center in a collaborative effort with the EPA's National Exposure Research Laboratory to evaluate landscape change. AGWA, which served as the starting point for the Internet-based SDSS, is based on ESRI's ArcView (ESRI 2000) GIS application and performs hydrologic model parameterization and results visualization for KINEROS (Smith et al. 1995) and the widely used SWAT (Arnold et al. 1994) watershed scale hydrologic simulation models. The application derives hydrologic model parameters from readily available digital elevation models, soils, and land cover data sets and allows users to spatially visualize changes in hydrologic response through the use of remotely sensed land cover scenes from different time periods. The primary purpose of AGWA is to evaluate the hydrologic response of land cover change (Hernandez et al. 2000, Miller et al. 2001, Kepner et al. 2002, Miller et al. 2002b) and the impact of geometric complexity on watershed scale simulation models (Semmens et al. 2001).

While the AGWA application simplifies hydrologic modeling and reduces the time needed to parameterize a simulation model and improves results visualization, it contains a number of shortcomings. AGWA is based on ESRI's ArcView 3.2, requiring users to have proprietary and expensive software installed, limiting the number of users who can utilize this application. Customizing ArcView 3.2 is conducted through ESRI's proprietary object oriented programming language, Avenue, which prevents the integration with more powerful programming environments. The application also requires users to have an understanding of GIS principles, further limiting its user base.

With advances in Internet technologies and specifically Internet GIS, the current effort developed and deployed a version of the AGWA application through the web targeting rangeland watershed managers. This project, funded by the USDA Cooperative State Research, Education, and Extension Service, provides land managers with access to GIS and hydrologic modeling technology without requiring users to manage complex spatial data sets and model parameter sets. The application allows land managers to compare environmental and economic effects of different land management systems.

The Internet architecture used in development determines the complexity and efficiency provided by an application. Currently, there are two types of Internet applications: client-side and server-side. Client side strategies require the majority of the processing to be conducted by the client, requiring the web browser to load a program (such as an applet or plug-in) the first time users request to view spatial data. This "thicker client" architecture provides the advantage of more functionality for users and requires fewer interactions with the server. However, applets are not persistent and must be downloaded at the inception of the application, and plug-ins are required to be downloaded and installed like traditional applications (Plew 1997). This type of architecture is typically best for applications with literate users (Plew 1997) because users are required to have knowledge of handling and manipulating the data. Server-side strategies perform all processing on the server, relying on the spatial server to conduct the analysis and generate output (Peng 1997). These thin-client applications require a high-performance server due to the computation intensity and have higher network congestion since each operation performed by users must communicate with the server. However, users have access to large and complex data sets and since client machines perform little processing, users are not required to have sophisticated computers (Foote and Kirvan 1997). Since tradeoffs exist between functionality, efficiency and required knowledge, integrated decision support systems should support multiple weight clients providing access to users with different backgrounds, experiences, and network connection speeds.

The Internet-based SDSS uses both architectures. Thicker client programs are used to allow users to enter their own data, such as the location of their own fences and water sources. However, Internet-based AGWA is primarily a thin client application. The databases and models are housed on the host server plus all simulations are preformed on the host server. The simulation results can then be viewed on the web browser. This provides the user with some basic functionality but still provides the advantages of a thin client application.

SDSS Functionality

The Internet-based SDSS provides core functionality required for rangeland watershed management planning and decision making. Users have the capability to dynamically delineate watersheds by clicking on a map to locate a watershed outlet (Figure 1). Using this boundary, users can perform simulations using hydrologic models with parameter sets derived from soils and land cover GIS data layers and spatially visualize results. In essence, the Internet application performs the same operations as the current standalone AGWA application using ESRI ArcView 3.2.

The application provides a "thicker" client to delineate rangeland management systems that consist of pasture boundaries, water points, and sediment detention structures. Each management practice contains user-defined attributes that are incorporated into the modeling process. Hydrologic and economic simulations are performed on user delineated management systems and results are presented in a spatial, graphical, and tabular format. Users can create "what if" scenarios such as locating water sources at different locations within a pasture or change the location of pasture boundaries and compare the runoff, sediment yield, and cost of different scenarios.

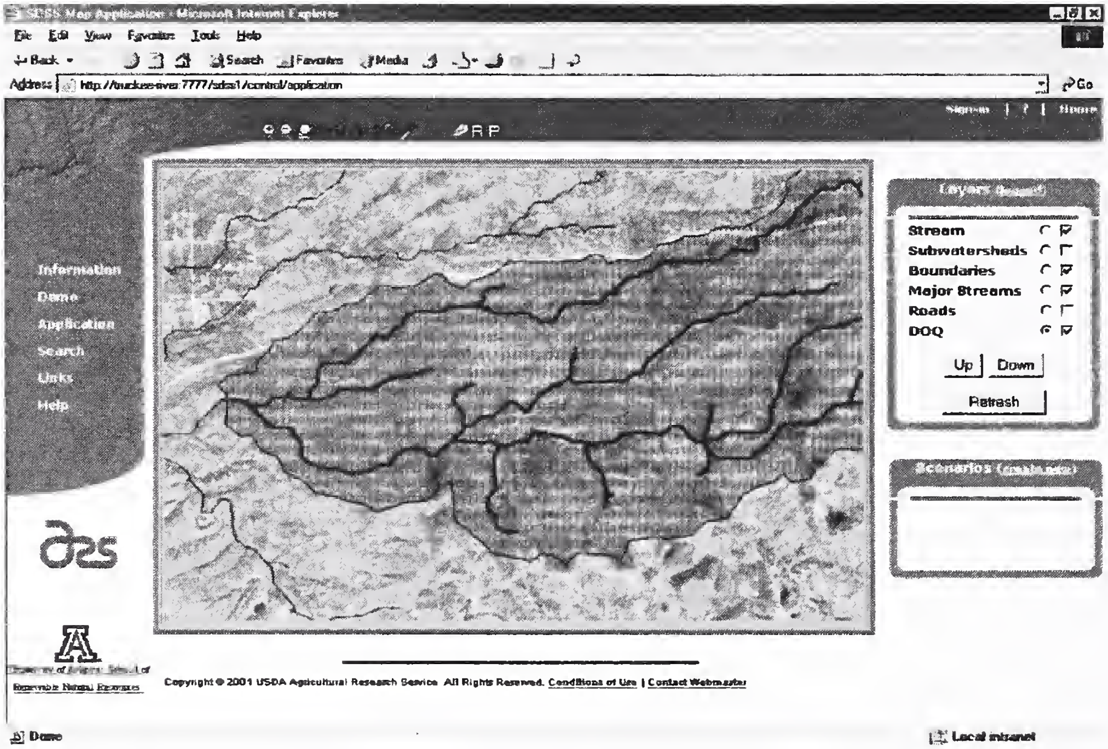

Figure 1. The Internet-based spatial decision support system (SDSS) provides users with the functionality to delineate watersheds by "clicking" on the map locating the watershed outlet.

The application can assess the effects of many of the common best management practices related to livestock management. Currently, the Internet-based SDSS can assess the impacts of fence locations, water source locations, stock ponds and changing vegetation cover and type. Vegetation management is modeled using the NRCS ecological sites guides. Using the vegetation information (herbaceous canopy cover, herbaceous basal cover, shrub basal cover, rock cover, etc.) on an ecological site's states, the Internet-based SDSS can change the hydrologic parameters for KINEROS. The user can delineate an area for improvement, such as shrub removal, and indicate the future transition state. The user can use either the average vegetation condition or simulation the response for low or high precipitation years. The location of fences and water is used to model the level of forage utilization across a pasture.

Areas near water within a pasture will be more heavily grazed (Guertin et al. 1998). Based on current stocking
rates and rotation systems the vegetation condition within a pasture will be assessed and its hydrologic impact simulated.

The user can use the Internet-based SDSS to change the management system and evaluate the effects of a set of different scenarios. The Internet-based SDSS can then perform a change analysis showing the hydrologic and economic effects of the different scenarios. The user can either save the results for later review or publish a report on the results.

Conclusions

The Internet-based SDSS for rangeland watershed management is currently being validated. As with other applications deployed via the web, the Internet based SDSS provides advantages over traditional desktop applications. First, the application is centrally located, simplifying distribution and maintenance. The application also uses predefined spatial data layers allowing the uncertainty in data inputs to be tested and quantified *a priori* using a Monte Carlo simulation approach (Malczewski 1999). In addition, the Internet based approach increases the user base by reducing costs of access to users. The current version of the Internet-based SDSS was targeted for rangeland watershed management. In future versions the Internet-based SDSS the functionality will be expanded to address

Integrated Watershed Management and Planning for semiarid watersheds. This will include the ability to address water supply and flooding issues as well as water quality and address the affect of other land uses including urbanization. Additional information can be found at http://www.tucson.ars.ag.gov/sdss

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Transfer and Application of Simulation Modeling in Important Environmental Problems

L. J. Lane

Abstract

Watershed simulation models developed by the Agricultural Research Service (ARS) and its collaborators have made significant contributions to understanding watershed processes and conservation and protection of natural resources. However, these simulation models have broader applications in important regional and national environmental problems (i.e. Superfund sites). General guidelines and specific suggestions are made to improve ARS watershed simulation modeling research and technology transfer to address these important environmental problems. An example application of simulation models at a Superfund site is used to illustrate properties of ARS watershed simulation models that would make them more useful for these applications. Benefits to ARS from cooperation with agencies and organizations responsible for remediation of Superfund sites include improved model evaluation, verification, and validation. ARS cooperation on these problems and the watershed research appropriate to help solve them would provide renewed vigor, emphasis, and recognition of watershed research.

Keywords: watershed modeling, continuous simulation, Superfund sites, technology transfer

Introduction

Watershed simulation models developed by cooperative research between the U.S. Department of Agriculture – Agricultural Research Service (USDA-ARS or simply ARS), universities, and other cooperators (model developers hereafter) have been extensively applied within the agricultural – natural resources research and technology community.

The main purposes of these models in the research community include formulating and testing hypotheses, developing predictive capabilities, and transfer of the models, and the related technology, to users and cooperators in the agriculture – natural resources conservation community. These models have significantly improved our understanding of natural resource systems, and, their use has significantly contributed to conservation and protection of these resources.

However, there are broader societal concerns in which ARS and its collaborators can make significant contributions through development and transfer of their watershed simulation models.

Superfund sites listed by the U.S. Environmental Protection Agency (EPA) are important regionally and nationally. The major environmental laws governing these sites include the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA or Superfund), the Superfund Amendments and Reauthorization Act (SARA), the National Environmental Policy Act (NEPA), and the Resource Conservation and Recovery Act (RCRA). Summary and complete text versions of these, and other applicable environmental laws, are given in EPA (2003).

Protection of human health and the environment at Superfund sites can be significantly enhanced by using natural resource simulation models such as those for watershed hydrology (water quantity and quality), soil erosion, sediment transport, sediment yield, contaminant transport, and contaminant yield.

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The ARS and its collaborators could significantly benefit from transferring their models and associated technology (e.g. databases, knowledge, and documentation) to agencies and organizations responsible for remediation of Superfund sites. Application of their models to problems at Superfund sites would significantly enhance the testing and evaluation phase of model development (i.e. model verification, validation, extension to extremes, and robustness) and thereby significantly reduce expenditures and time required for ARS scientists to accomplish this phase. Successful technology transfer requires collaboration between model developers and model users. This collaboration can provide additional resources and insight during all phases of model development and transfer. Finally, successful use of ARS models and technology to help solve important national problems enhances the scientific standing of ARS and its scientists.

The main purposes of watershed simulation models used by agencies and organizations performing environmental remediation at Superfund sites include predicting source, transport, fate and impact of contaminants. Their purposes also include analysis and interpretation of monitoring data, and evaluating performance of alternative remediation alternatives. Finally, these models and their results are used to communicate with regulatory organizations, state and local governments, NGOs, other stakeholders, and the general public. Therefore, the models and the results of applying them are subject to peer review, regulatory review, and public review.

Purpose, scope and limitations

This paper reviews selected properties of models that enhance their transferability, discusses some specific examples, and makes recommendations for improving the models and how they are developed and transferred. These recommendations span the entire process from basic research to technology transfer and application and to user feedback necessary to maintain a strong simulation modeling research and development effort. The example used to support broader ARS involvement in environmental problems is for the Rocky Flats Environmental Technology Site (RFETS) in the semiarid western United States. Finally, conclusions and recommendations given herein are based mostly upon the author's experiences as a consultant to agencies and their contractors (model users hereafter) at DOE Superfund sites using watershed simulation models. These simulation models have emphasized hydrology, soil erosion and sediment yield, and contaminant transport due to runoff and erosion and thus were limited to a subset of modeling needs at Superfund sites.

Models Used in Environmental Remediation

As stated above, models used in Superfund site environmental remediation function as predictive tools, data and uncertainty analysis tools, alternative evaluation tools, and, communication tools. In these uses the models are subject to extensive peer, regulatory, and public review. The following sections describe some characteristics or properties these models should have to be successful for these uses.

Scientific credibility

Scientific credibility is crucial to model users, regulators, and the public accepting models and their results. In this user-regulator-public arena (simply user arena hereafter), scientific credibility is established by peer-reviewed publications, documented peer review of the site-specific applications, reasonableness of the results, and the ability to communicate them. A big part of this acceptance by the users is previous acceptance of the models in similar applications. Model developers should document the models and their applications in the scientific literature.

Presentation of results

Modeling results and output must be understandable in the user arena. A critical part of most model applications at Superfund sites is spatially distributed results. The problems usually involve contaminants spatially distributed in the environment and the model results should directly address spatially distributed processes and results. Model developers attacking spatially distributed processes should adopt geospatial referencing early in model development activities. Practically, this means models should be developed and implemented in a geographic information systems (GIS) environment.

Continuous simulation

Continuous simulation means that the models simulate processes during and between precipitation-runoff events. This is necessary to calculate a water balance. For example, evapotranspiration processes continue between events and can deplete soil moisture affecting the amount of infiltration, runoff, soil erosion, etc. when an event does occur. In addition, hydrologic processes such as evapotranspiration, deep percolation and groundwater recharge/discharge occur continuously, sometimes at rates far slower than those that occur during a storm event. Therefore, continuous simulation models are required to compute a water balance, calculate low flows, calculate watershed yields, and thus contaminant loadings.

Evaluation of alternatives and uncertainty

The goal of environmental remediation is to change the system. This means that models must have the ability to predict site performance into the future. Models are used to quantify and predict contaminant transport pathways (surface water migration, ground water migration, air migration, biological transport) and contaminant inventories (especially in soil and water). These evaluations and predictions are made for the initial existing conditions and then for alternative remediation scenarios designed to change the pathways and inventories to reduce contaminant impacts on human health and the environment. Therefore, models most useful in the user arena should be designed to evaluate the impacts of land use and management practices (including landscape reconfiguration) alternatives, to quantify their differences and to quantify prediction uncertainty. Understanding and specifying uncertainty is critical to evaluation of alternatives. The desired changes must exceed the uncertainty bounds in contaminant transport and inventories if the proposed alternatives are to be judged different than the existing conditions.

Reasonable results

Modeling results must be reasonable. And, the criteria with which the results are judged reasonable vary between the model users, the regulators, other stakeholders, and the general public. Model results must "match" empirical data to the extent possible. Matching, in the context of users and peer reviewers, is statistically preserving the means, uncertainty, and trends in space and time. Models must also give reasonable results at the extremes – for small events and for large events. Matching, in the context of the general public, means meeting their expectations as well as those of the users and peer reviewers with regard to how well the model reproduces measured data and trends. Model developers should thus test and document model performance across a broad range of inputs and conditions to make sure the models are robust as they "match" empirical data. Many of the users require the ability to predict contaminant concentrations under extreme conditions (low flows, floods, high winds, etc.) as well as long-term yields of water, sediment, and contaminants. These applications require continuous simulation.

Example of Models Used at a Superfund Site

The Rocky Flats Environmental Technology Site (RFETS) is located in Golden, CO. It is a former site of production of nuclear weapons components as part of the DOE weapons complex. Contaminant spills and subsequent transport by wind and water erosion have resulted in environmentally dispersed radioactive contaminants. The site is being remediated to meet federal, state, and local regulations by reducing offsite transport of contaminants to below regulatory limits. The following sections list the simulation models used and some of their characteristics.

Site-wide water balance

The model chosen to calculate a water balance was MIKE-SHE, a model developed by the Danish Hydraulic Institute (DHI) to simulate the land phase of the hydrologic cycle (e.g. see Storm and Refsgaard 1996, DHI 2003). This model is continuous, couples surface water and groundwater, and operates on a spatial grid. Key features leading to its selection (selection features) were its dynamic operation, continuous simulation, coupled surface and subsurface flow calculations, spatial operation, and its reputation as a "state-of-the-art" model. It also had good graphic presentation capabilities.

Wind erosion and contaminant transport

The wind erosion model was developed by contractors at RFETS because existing models were judged as inadequate. Key development features included its dynamic operation, continuous simulation, coupled sediment and contaminant simulation, and its spatial adaptation to the RFETS.

Water erosion

The Water Erosion Prediction Project (WEPP) model (Laflen et al. 1991, USDA-ARS-NSERL 2003) was chosen to compute upland soil erosion. The WEPP model has continuous and single storm options, simulates at the hillslope scale, and has options for cropland and rangeland applications. Key features of the WEPP model leading to its selection include its ability to operate in a continuous simulation mode, its ability to simulate erosion on complex hillslopes (multiple overland flow elements), its calculation of sediment yield from hillslopes by particle sizes (and thus the ability to compute enrichment ratio for subsequent contaminant transport calculations), and its reputation as a "state-of-the-art" model.

Sediment transport

The model chosen to calculate sediment transport was the HEC6T Model (Thomas 2002, MBH 2003), a proprietary modification of the HEC6 model. The HEC6T model is a one dimensional sediment transport model. Key features leading to its selection included its ability to compute non-uniform flow (backwater at constrictions, drawdown at over falls, etc.), sediment transport by particle size distribution, sub-critical, critical, and super critical flow options, calculation of channel degradation or aggradation (mobile bed hydraulic calculations), and its reputation as a "state-of-the-art" model.

Contaminant transport

The "Actinide Mobility Calculations" model was developed by contractors at RFETS because existing models were judged as inadequate. Key features developed in this model included its ability to use WEPP output (sediment yield by particle size distributions and enrichment ratios), its ability to provide contaminated sediment input to the HEC6T model, and its spatial adaptation to the RFETS.

Discussion

Simulation modeling at RFETS was required to meet several objectives. The RFETS model users required the ability to predict water balance, soil erosion, sediment transport, sediment yield, and contaminant concentrations under extreme conditions (low flows, floods, high winds, etc.). The users also needed to predict long-term yields of water, sediment, and contaminants under alternative Site management scenarios. These requirements could be met using models that used continuous simulation. The models described above were integrated at the watershed scale to meet the user requirements.

The RFETS model users also needed models that met the criteria listed earlier, i.e. scientific credibility, spatially distributed computations, robustness, and reasonableness of results.

Developing Models to Meet User Needs at Superfund Sites

We have already discussed many things the model developers should do to meet users' needs, but these, and additional recommendations need to be formalized and described with sufficient specificity to provide guidelines. The following partial list of model requirements is designed as a starting point for model developers:

- Obtain peer review at all steps in model development,
- Document the models in peer-reviewed scientific publications,
- Develop, operate, and present the models in a GIS environment,
- Use spatially distributed data, starting with digital elevation models (DEMs) and including topography and drainage channels, soils, vegetation, and land use and management should be included as part of the model parameterization process and it should be automated in the GIS environment,
- Formulate the models to be robust; they should operate properly for the means as well as the extremes,
- Produce predictions of means, extremes, annual yields, and their uncertainties,
- Base the models on continuous simulation, rather than individual events only, to enable calculations of watershed yields and contaminant loadings,
- Automate uncertainty analysis within the models, and
- Develop models that include procedures to manage metadata, input/output databases, and to generate reports containing the metadata and documenting input/output for archiving.

Recommendations

The ARS Strategic Plan states that a major goal of ARS research is to "increase the long-term productivity of the United States agriculture and food industry while maintaining and enhancing the natural resource base on which rural America and the United States agricultural economy depend." A major strategy to help accomplish this goal is to "develop new concepts, technologies, and management practices that will enhance the quality, productivity, and sustainability of the Nation's soil, water, and air resources." Finally, a performance goal to help accomplish the above goals is to "experimentally demonstrate the appropriateness of watershed-scale technologies and practices that protect the environment and natural resources (ARS 2003)."

The following recommendations are specific to the Agricultural Research Service and its collaborators responsible for developing and technology transfer of natural resource (watershed based) simulation models as stated in the above goals. My experience has been with hydrologic (including quantity and quality), soil erosion, sediment transport, sediment yield, and contaminant transport and yield models. However, development and transfer of other models such as landscape evolution, interaction of biotic and abiotic processes, and ecosystem sustainability might also benefit from the following recommendations.

These are high level recommendation for ARS. Lower level detailed steps and recommendations are contained earlier in the body of this text.

The ARS should develop a conceptual model of watershed simulation modeling from initial concepts to technology transfer that includes model users and stakeholder inputs at all stages in the process. This conceptual model could be used to communicate goals and strategies from the ARS Strategic Plan to its scientists, cooperators, model users, and stakeholders and thereby enhance technology development and transfer.
The ARS watershed research centers and experimental watersheds are unique and should provide a basis to expand their cooperators and stakeholders to include those involved in important regional and national environmental problems. Cooperation on these problems and the watershed research appropriate to help solve them would provide renewed vigor, emphasis, and recognition of watershed research. A key objective should be to cooperatively develop and transfer the simulation models so vital in protecting our environment and thereby conserving our natural resources.

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The Analytic Hierarchy Process as a Means for Integrated Watershed Management

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Abstract

Integrated Watershed Management (IWM) has emerged worldwide as the preferred model for watershed planning. IWM uses the watershed as the basic geographic planning unit while integrating social, economic, ecological and policy concerns with science to develop the best plan. Stakeholder input is key to successful IWM. However stakeholder participation can present problems when the public is uncertain or unclear about the IWM planning criteria. The Analytic Hierarchy Process is a decision method for assisting IWM because it treats planning criteria and criteria weighting in an open and explicit manner.

Keywords: integrated watershed management, analytic hierarchy process, watershed planning, multi-criteria decision methods, multi-criteria decision models

Introduction

Today, the emerging field of integrated watershed management envisions the watershed as a holistic planning unit. However, while the integrated watershed management approach offers a process for solving watershed management problems, it also presents formidable challenges in terms of implementation. Thus, the purpose of this presentation is to explore the Analytic Hierarchy Process (AHP) as a means of assisting the implementation of integrated watershed management. The focus of this article is upon the AHP as a means for assisting in the plan selection process.

Watersheds and Integrated Watershed Management

The watershed is defined as, "the region draining into a river system, river or body of water" (Morris 1976). Watersheds are a highly desirable unit for planning because they are physical features ubiquitous across the landscape serving as the geographic foundation for political states. As planning units, watersheds transcend political boundaries. However, prior to the 1970's, most watershed management focused on solving localized problems without taking into account the interrelationship between those problems and the biophysical, economic and social elements of the larger watershed system (Heathcote 1998). Furthermore, during most of the mid- to late- 20th century, watershed management was, politically, a top-down planning process with national concerns pre-empting local (National Research Council 1999).

Today, however, countries everywhere are exploring bottom-up watershed planning for water, natural resource and environmental management through "integrated watershed management." Integrated watershed management (IWM) is a holistic problem-solving strategy used to protect and restore the physical, chemical and biological integrity of aquatic ecosystems, human health, and provide for sustainable economic growth (National Research Council 1999). IWM, in its most basic form, considers the interdependencies between science, policy and public participation (National Research Council 1999).

Over the past two decades, there have been numerous applications of IWM worldwide. For example, integrated watershed management approaches have been recently used for combating

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drought in the Jhabua watershed in India (Singh et al. 2002), assessing and managing water resources in the upper Chao Phraya in Thailand (Padma et al. 2001), assessing and managing agricultural phosphorus pollution on the Chesapeake Bay (Sharpley 2000), tackling the problem of land degradation in Australia (Ewing 1999), and managing the Truckee River in Nevada (Cobourn 1999). Also, in the United States, the USEPA has been quite instrumental in promoting the integrated watershed approach to management (National Research Council 1999):

> The lessons learned from these and other initiatives indicate that in order to succeed, integrated watershed management must be participatory, adaptive and experimental, integrating all the relevant scientific knowledge/data and user-supplied information regarding the social, economic and environmental processes affecting natural resources at the watershed level.

Plan Selection Using Stakeholder Values

The IWM process involves several distinct steps as follows (adapted from Heathcote 1998): 1) problem scoping and definition with decision-makers and professionals, 2) assessment of legal and institutional concerns, 3) consultation with stakeholders, 4) inventory of the geology, soil, streamflow, groundwater, water quality, plant and animal communities, land use, and the social and economic systems, 5) development of management options, with associated costs, to solve the problem(s), 6) assessment of management options, 7) environmental and social impact assessment as required by law, 8) selection of the best plan, 9) obtaining financial support, and 10) implementation and monitoring of the plan.

Due to page limitations, we will not attempt to discuss each of these steps in detail. For that detail, the reader is referred to Heathcote (1998). Taken together, all of these steps provide a comprehensive approach to IWM. This paper focuses on one especially critical step, that of "selecting the best plan." The selection of the best watershed plan is an especially important step because it represents the culmination of the IWM process and, thus, sets the course for the future of the watershed.

Observers and practitioners agree that it is crucial for stakeholders to be involved in developing and selecting the best plan (National Research Council 1999). Otherwise, it is said, the entire process may be ineffective. This is true because a strict top-down planning and plan selection process (*sans* stakeholders) can create implementation barriers due to the lack of public support of, or even opposition to, the final plan.

However, as important as stakeholder participation is to the plan selection, this same stakeholder involvement renders this step the "most difficult and controversial" in the IWM process (Heathcote 1998). Difficulty and controversy in "selecting the best plan" arise when stakeholders do not fully understand the criteria used for the IWM process. A group of stakeholders may well agree upon the overall watershed problem. They may also understand the goal of the IWM process. Furthermore, they may very well accept the information and data brought to the process. However, they may not understand the criteria, nor the criteria weights, used to determine the best plan. As a result, they may not always agree upon the choice of the best watershed plan.

As Heathcote (1998) states: "In some decision processes, these...criteria are not made explicit, with the result that participants disagree about the acceptability of an option without a clear understanding of the reasons for their dissatisfaction. Explicit discussion of...evaluation criteria encourages better citizen understanding and more focused decision making and can strengthen...support." Indeed, better understanding of the decision criteria can manage stakeholder conflict.

To illustrate the role of criteria in plan selection, consider a hypothetical watershed planning problem adapted from Heathcote (1998). The situation concerns the implementation of best management practices (BMPs) on farmland to prevent non-point source pollution of a river. Thus, the IWM goal is to select BMPs to prevent non-point source pollution.

There are eight alternative management options being considered: 1) "do nothing", 2) construct buffer strips, 3) construct fencing, 4) use conservation tillage, 5) construct buffer strip and fencing, 6) construct fencing, use conservation tillage, 7) construct buffer strips, use conservation tillage, and 8) construct fencing, buffer strips, use conservation tillage. Each represents a mutually exclusive, potentially independent project for managing non-point source pollution. Each alternative would have a unique budget, schedule and associated considerations.

Finally, there are criteria used to help select the best alternative. Criteria are measures of the effectiveness or suitability of the possible management actions. They must be: 1) measurable by mutually agreed upon methods, and 2) they must vary between, and thus differentiate, the alternatives. In this case, the criteria were: 1) cost, 2) time to implement the BMPs, 3) meets legal requirements.

Thus, in order to choose the best plan, each alternative would be evaluated according to the criteria and the alternative that scored highest would be selected and implemented.

The Analytic Hierarchy Process and IWM

A tool that permits explicit presentation of evaluation criteria and, thus, possibly improves IWM plan selection is the Analytic Hierarchy Process (AHP). The AHP is a Multi-Attribute Decision Method (MADM). MADM refers to a host of quantitative techniques used to facilitate decisions that involve multiple competing criteria. MADM methods use multiple criteria rather than relying on a single criterion to make a decision as in, say, cost-benefit analysis (max net present value). Thus, MADM methods are ideally suited to address decision situations such as our hypothetical BPM problem that featured multiple criteria for selecting the best alternative.

MADM examples include Multi-Attribute Utility Theory, the Novel Approach to Imprecise Assessment and Decision Environments, the Outranking Method, and the Analytic Hierarchy Process (DeMontis et al. 2000).

The AHP is perhaps the most widely-used of the MADM methods. We choose it for study here because it has a number of desirable attributes, such as: 1) the AHP is a structured decision process quantitative process which can be documented and replicated, 2) it is applicable to decision situations involving multi-criteria, 3) the AHP is applicable to decision situations involving subjective judgment, 4) it uses both qualitative and quantitative data, 5) it provides measures of consistency of preference, 6) there is ample documentation of AHP applications in the academic literature, 7) commercial AHP software is available with technical and educational support, and 8) the AHP is suitable for group decision-making.

The steps in the AHP method are as follows:

Step 1: The AHP begins with the development of a decision hierarchy with an *objective*, *alternatives* and *criteria*. Decision hierarchies are most effective if all stakeholders are involved in the development process. An AHP hierarchy can have as many levels as needed to fully characterize a particular decision situation. For example, consider the following hypothetical two-level (i.e., one set of choice criteria and one set of choice alternatives) IWM decision situation (Figure 1). Assume that a watershed councils' objective is to select the best possible watershed plan from three alternatives: Plans A, B and C. Assume also that there are four choice criteria that enter into this decision: 1) water quality, 2) timber production, 3) riparian protection, and 4) cost. The alternatives, although not explicit in this example, could provide the decision-makers with information of either a quantitative nature or a qualitative nature.

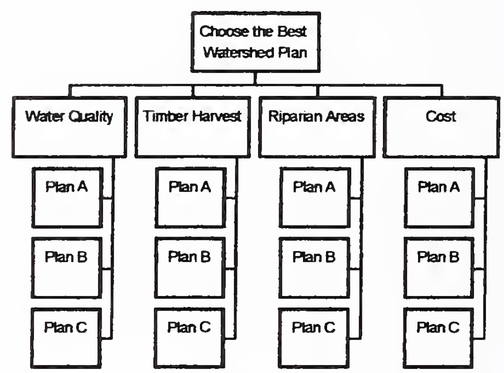

Figure 1. Decision hierarchy for a hypothetical watershed.

Step 2: Next, the decision-makers individually express their opinions regarding the relative importance of the criteria and preferences among the alternatives using *pairwise* comparisons and a 9-point system ranging from 1 (the two choice options are equally preferred) to 9 (one choice option is extremely preferred over the other). If, however, one criterion is preferred *less* than the comparison criterion, the reciprocal of the preference score is assigned. The 9-point scale has been the standard rating system used for the AHP (Saaty 2000). Its use is based upon research by psychologist George Miller (1956) which indicated that decision makers were unable to consistently repeat their expressed gradations of preference finer than "seven plus or minus two."

Step 3: These preference scores next undergo a synthesis process in order to calculate a priority weight vector for the criteria. There are different possible methods for synthesizing preference scores (Anderson et al. 1994, Saaty 2000). Whichever method is used, the final result, illustrated by this example, is a 1 x 4 vector (designated as **X**) of normalized, i.e., summing to 1, criteria preference scores. Once the scoring and synthesis process has been completed for the criteria, it is conducted for the alternatives. In this example, there are three alternatives, hence three vectors of weights, which are arranged to form a 4 x 3 (i.e., four criteria by three alternatives) matrix (designated as **Y**) of normalized preference scores.

Step 4. The final step in the AHP process is to complete the synthesis by multiplying the 1 x 4 "criteria vector" by the 4 x 3 "alternatives matrix" in order to obtain a 1 x 3 vector (designated as **XY**) of normalized unit-less weighted preference scores for each of the three plan options. For example, this hypothetical AHP exercise might have yielded final weighted preference scores for the three plans as follows: Plan A = 0.35 + Plan B = 0.25 + Plan C = 0.40 = 1.0. Plan C then is the decision-makers' preferred plan based upon their subjective judgment. The score (i.e., 0.40 out a possible 1.0) indicates the relative strength of that preference. Hämäläinen and Salo (1997) state that the final weights that result from the AHP represent the priority ordering of the alternatives and, thus, permit determination of the most preferred alternative. Another interpretation of possible "meanings" of the AHP weights is a more complex issue (Hämäläinen and Salo 1997).

In addition to final preference weights, the AHP permits calculation of a value called the consistency index (Anderson et al. 1994, Saaty 2000). This index measures transitivity of preference for the person doing the pairwise comparisons. To illustrate the meaning of transitivity of preference, if a person prefers choice A over B, and B over C, then do they in consistent fashion prefer A over C? Furthermore individual scores can be aggregated to obtain a composite group score (Saaty 2000).

AHP Examples, Drawbacks

Applications of the AHP to complex decision situations number in the thousands (Zahedi 1986). However, the application of the AHP to natural resource problems has been "surprisingly limited" according to Schmoldt, Kangas and Mendoza (2001). Unfortunately, page limitations do not permit a review of this literature.

Despite its widespread use as a decision method, the AHP has received some criticism (Hill and Zammit 2000): 1) because no theoretical basis exists for the formation of hierarchies, decision makers, when faced with identical decision situations, can derive different hierarchies, thus different solutions, 2) the rankings produced by the AHP are arbitrary because they are produced by a subjective opinion using a ratio scale and these arbitrary rankings can lead to "rank reversal," 3) flaws exist in the methods for aggregating individual weights into composite weights, and 4) an absence of a sound underlying statistical theory. Despite these concerns, the AHP remains immensely popular among private and public sector decision-makers.

Conclusions

Integrated watershed management situations consist of multiple criteria and alternatives that must be evaluated by a decision-maker in order to achieve an objective. The AHP provides a systematic method for comparison and weighting of these multiple criteria and alternatives by decision-makers. AHP is thought to be a method and planning framework with potential for implementation of IWM. An advantage of the AHP is that it is capable of providing numerical weights to options where subjective judgments of either quantitative or qualitative alternatives constitute an important part of the decision process. Such is often the case with IWM. A disadvantage is that the AHP method can be time-consuming and tedious if there are many levels in the decision hierarchy. Commercial software is available to simplify the AHP rating process, consistency indices and to perform matrix calculations. Also, the context of the decision and the sophistication of the decision-makers is crucial to the use and success of AHP.

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Urban Watershed/Water Body Restoration – The Driving Forces

Vladimir Novotny, David Clark, Robert J. Griffin

Abstract

Urban streams are used for several purposes. Some uses are conflicting and some are complementary. The use of urban water bodies and the resolution of conflicts is driven by anthropogenic and biocentric/ecocentric interests that must be optimized and the conflicts resolved.

This article examines and analyzes land ethics (biocentric) and socio-economic (anthropocentric) drives for stream restoration of urban watersheds located in the Milwaukee (WI) metropolitan area. The basins experienced increased flooding, significant degradation of sediment and water quality, and loss of aquatic species, all due to urbanization. It was found that the primary drivers for restoration of urban streams are the ethical attitudes of population towards the ecocentric benefits of restoration in combination with a desire for flood control. A Contingent Valuation Survey of citizens residing in two Milwaukee watersheds revealed that those who see the watershed in ecocentric terms appear to have a greater Willingness to Pay for watershed/water body improvements than those who see the benefits solely in anthropogenic terms of reduction of flood damages.

Keywords: stream restoration, flood control, benefits evaluation, contingent valuation, urbanization

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Introduction

Urbanization in most cases results in downgrading the integrity of water bodies and watershed in urban and urbanizing areas. The root causes of degradation are well known and include hydrological changes such as increased peak flows and flooding at one end and loss of base flow on the other. Water quality is degraded by contaminants in urban runoff and, in some cases, by overflows from combined and even sanitary sewers (CSOs and SSOs). An ultimate degradation of an urban stream is to line it with concrete or riprap, straightening the channel, and sometimes covering it, essentially converting the stream to an underground sewer. These modifications could be categorized, based on a definition in Section 5 of the Clean Water Act, as pollution. However, the only tool available currently to agencies to initiate stream restoration, the Total Maximum Daily Load process (Section 303(d) of the Act), is ineffective to bring about compliance of the goals of the Clean Water Act of urban streams affected by "pollution" that does not involve discharges of pollutants.

Urban streams are used for a variety of purposes, including : (1) flood conveyance; (2) disposal of urban runoff and overflows from sewer systems; (3) aesthetic enjoyment by the urban population; (4) aquatic life propagation; (5) contact and noncontact recreation (sailing and fishing); (6) potable and nonpotable water supply (e.g., golf course irrigation); (7) other uses that may include cooling, navigation, and groundwater recharge. Some uses are conflicting with one another. For example, fast conveyance of floods interferes with aquatic life propagation and recreation, potable water use typically restricts contact recreation. Some uses complement each other; e.g., the use of the urban water body for water supply or for contact recreation necessitates a healthy ecology of the water body. The uses of the urban water bodies are driven by anthropogenic and biocentric interests that may conflict; therefore, the conflicting uses must be optimized.

Rationale

The research described in this paper was part of a large interdisciplinary research sponsored by the USEPA/NSF/ USDA STAR (Science to Achieve Results) watershed program. The relatively small urban watersheds analyzed in the research (Novotny et al. 2001) and the paper are

- the Menomonee River and Oak Creek in Milwaukee County (WI)
- Lincoln Creek in Milwaukee County (WI)

Other notable examples of restoration of a stressed urban water body are the Rouge River in the Wayne County (MI) and the Muddy River in Boston/Brookline (MA).

The partially urbanized Menomonee River and fully urbanized Lincoln Creek in the Milwaukee metropolitan area have been undergoing substantial restoration efforts, with approximately the same cost (more than $70 million each). The restoration of the urban Muddy River in Boston/Brookline is in the final planning stages. Oak Creek in Milwaukee County is undergoing rapid urbanization/transition from rural to urban. None of the analyzed water bodies receives significant point inputs from wastewater effluents.

The three urban watersheds (Menomonee River, Lincoln Creek and Muddy River) are experiencing increased and more frequent flooding, degradation of sediment and water quality, and loss of aquatic species, all due to the impact of urbanization. In Milwaukee's Menomonee River and Lincoln Creek watersheds, the annualized tangible benefits of flood control amounted to only a fraction of the cost. Because most ecological benefits are intangible, the ecocentric uses are in a distinct disadvantage. However, societies and agencies, today, may not accept nor finance flood control and stream restoration projects that would have negative net benefits.

Because restoration of the water bodies and riparian floodplains and development of storage oriented best management practices for storage and treatment of runoff (e.g., ponds and wetlands) also have significant flood control benefits, accomplishing both goals is possible in the investigated watersheds.

Implementing abatement of urban diffuse pollution, stream habitat restoration, and remediation of contaminated sediments is a problem because the solution cannot be mandated and only minimum federal government funding is available for water body restoration (with the exception of Rouge River that is a pilot project with significant federal funding). Most funding must come from local sources and from citizen's initiatives. Thus the ecocentric attitudes of the citizens of the watersheds, originally defined fifty years ago by Leopold (2001) play an increasingly important role. Watershed/water body restoration will not happen if citizens do not exercise their land (environmental and biocentric) ethic attitudes. However, until recently, the biocentric and environ-mental attitudes were demonstrated only by citizens' organizing into "friends of the river" committees, public pressure on developers and legislators, and court action. However, quantitative measures of attitudes (willingness to pay) were sparse.

Problems of urban water bodies

Multiple and conflicting uses of urban water bodies without reconciling conflicts, leads to short term resolution of the most publicized problem (e.g., flooding), often with long term adverse consequences. In the past, urban engineers tried to resolve the problem of increased floods by increasing the velocity and flow capacity of urban streams. Such conveyance oriented flood control approaches did not improve water quality, were detrimental to habitat and dangerous during flooding to citizens. Moreover, they passed flood control problems downstream. At the same time, development continued to encroach on floodplains, exacerbating flooding problems. Traditional cost-benefits evaluations often revealed negative net benefits as cost far exceeded the flood control damage reduction and the tangible benefits were frequently limited to citizens residing in floodplains.

In contrast, storage oriented approaches enhance flood storage by including infiltration, storage ponds and wetlands both throughout the watershed and in existing and reclaimed floodplains provide numerous ecological benefits and are the necessary prerequisite of revitalization of urban streams. Such best management practices are also an integral part of solving the diffuse pollution problem of urban streams. In addition, contaminated sediment remediation should be part of the overall plan.

Measuring and evaluating benefits of urban diffuse pollution control and water body/watershed restoration is difficult and the standard benefit-cost approaches do not work nor would be applicable. Improving the ecological quality of the resource generates private and public benefits that are direct and indirect, tangible and intangible. Among those residents who use the water resource for recreation activities such as hiking, sailing, fishing or swimming, an improvement in ecological quality can improve or even reinstate a recreational experience. Such benefits are direct. However, even local residents who are not currently users may want to improve the environmental quality of the water resource for themselves or their children's private future use (known as option value). Existence values are benefits that an individual receives from knowing that a resource is preserved or enhanced even though the consumer never intends to use the resources (Krutila and Fisher 1975, Mitchell and Carson 1989). Such existence benefits are divided into vicarious consumption (by significant others, relatives or close friends, and by general public), stewardship values (preservation or bequest) or even enhanced sense of civic pride resulting from improving or restoring a local environmental resource. These are direct benefits to consumers, even though the good is public in nature. A typical current practice of evaluating benefits is to count only active users of the water resource, for example, by estimating the number of recreational users and assigning a numeric value of a benefit to each user.

Measuring benefits associated with flood control projects by traditional cost/benefit analysis wherein the reduction of tangible flood damage is the benefit are incomplete for several reasons. First, they are based on the false premise that the only benefits of flood protection are those experienced by residents in the floodplain. Second, they fail to fully recognize some ecological benefits that may be derived from some ecologically enhancing storage oriented flood control projects. Third, they are incapable to include the intangible external cost of the ecological damage done to the stream corridor by channel modifications and floodplain development.

Socio-economic conflicts

It is important to briefly overview the numerous and often conflicting actors and interests that are affected by urban watershed management and seek to influence it.
First, federal, state, regional and local governments and supporting institutions (e.g., regional planning commissions, regional drainage agencies) are the most obvious and powerful agents of management. However, because watersheds and floodplains do not fall exactly within the geographical jurisdictional boundaries, problems arise. Also, different governmental organizations and units may have conflicting objectives, depending on their constituencies, interests, funding sources, relationship to other agencies, etc.

Second, due to the large expense associated with watershed management and preservation/restoration projects, many policymakers are hesitant to initiate proactive policies, especially those that may present a financial burden on population. Policymakers usually react to what they perceive to be the demands of their voting public and derive their policy concerns from stakeholders, public meetings, and media coverage of flooding and stream bad quality calamities. Without the intelligence of an unbiased and valid public opinion survey their perception of public concerns can be erroneous, since only motivated people will voice their concerns directly to policy makers or attend a meeting. News media coverage usually include salient events (e.g., flooding) rather than trends (e.g., progressive worsening of water quality and loss of the ecological value of a water resource due to urbanization). Thus concerns with flooding, driven by policymakers' perceptions of the media and public concerns, generally drive urban watershed projects. This was the primary driver of the Menomonee River, the Muddy River and Lincoln Creek projects. However, in the 1980s and before, citizens' participation on watershed (primarily flood control) projects was minimal and restricted mostly to citizens' advisory committees with few members. In the case of Lincoln Creek, in the early 1990s, the flood protection only project relying on fast conveyance ran into stiff opposition from the public and environmental groups that virtually stopped the project while trucks with concrete were being delivered. Without proactive environmental communication and knowledge of environmental benefits of stream corridor preservation/ restoration, the linkages between the ecological status and use of the water body for conveyance (and by the same reasoning for other purposes requiring hydraulic modification such as navigation or excessive water withdrawals) are blurred.

Finally, it was recognized at the end of the last century that efficient watershed management involves more than a reduction of the flood risk. It incorporates issues as diverse as ecological integrity, water quality, public health and safety, urban and rural development planning, and aesthetic/quality of life concerns. Watershed management encompasses a number of social, economic, ethical and environmental issues. Consequently, effective watershed management planning and policy formation require knowledge on the benefits and costs of management actions and public acceptance.

Ecocentric and anthropogenic values

Environmental values of urban watersheds are a special form of basic views about how things should be in the world and what should be done to make urban areas a better place (Norton 1995, Leopold 2001). They can be anthropocentric or biocentric. In the case of anthropogenic values, environmental improvement should be undertaken only for the material benefits of people. For biocentric environmental values, ecological improvements should be undertaken for the sake of nature itself apart from any material human benefits. The ecological restoration of a watershed, for example, should be undertaken if it benefits the species present whether or not there is any material benefit to human beings. As Leopold (2001) noted, out of 22,000 species of birds, fish and animals in Wisconsin only a few percent have any economic value, yet they deserve protection. This means that individuals with environmental and biocentric values could support ecological restoration even if neither they or anyone else experience added material benefits such as improved recreation opportunities, higher market values for riparian and near stream properties, or cleaner drinking water.

One of the major objectives of the research conducted at Marquette University (Milwaukee, WI), described in this article, was to investigate the role and extent of land ethic defined by Leopold (2001) (or environmental perspectives) as evident in the beliefs and attitudes of citizens in urban and urbanizing watersheds, in particular as related to conservation and ecological restoration under the threat of increased flooding caused by urbanization. The research estimated the citizens' willingness to pay to support stream restoration and sound flood protection as well as communication, attitudinal belief, and other psychological factors that may affect that support.

Method

A two wave phone scientific survey of more 1000 citizens residing in the watersheds of the Menomonee River and Oak Creek was conduced during the 1998-2001 period (Figure 1). The survey was preceded by several focus group sessions that tested the survey and significance of questions included in the questionnaire. The measure of the citizens' attitudes was the willingness (WTP) to pay for flood control and environmental restoration projects. A comprehensive research report (Novotny et al. 2001) also addressed the hydrologic impact of urbanization and developed measures of water body integrity – ecological risks that were then, in a simplified form, conveyed to the respondents of the survey. Given that residents in the Milwaukee metropolitan area have experienced several large (more than 100-year) flood events in the last 15 years (1986, 1997 and 1998), the issue of flood control has had a high public profile. Since flood control projects focus primarily on mitigation of flood risks or they may employ techniques that also improve the ecological integrity of the watershed, an understanding of the relative importance of these two objectives of watershed management is needed. The socio-economic study developed and conducted by the second and third author employed Contingent Valuation Method (CVM) to evaluate community support for watershed management practices. The CVM was used to estimate value for environmental improvements and flood control, relying on individual responses to hypothetical circumstances. A parallel analysis utilizing models of risk communication and testing at a more micro level of psychological variables that correlate to willingness to pay was also conducted. Finally, the survey also contained questions related to environmental ethics and its relation to WTP.

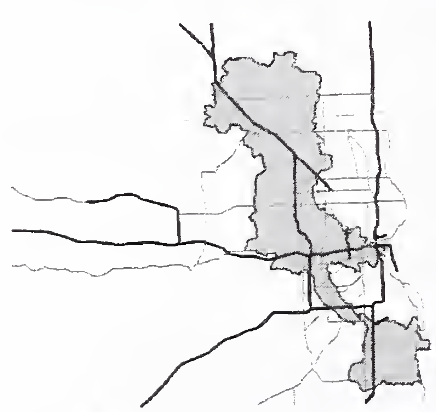

Figure 1. Menomonee River and Oak Creek watersheds.

The generalized model for the analysis of the survey is given by equation

Ln(WTP) = f(*demographic, residence controls, survey controls, attitude/value, risk,* ε)

where *demographic, residence controls, survey controls, attitude/value* and *risk* are vectors of variables contained in the model, and ε is the random error variable.

The surveys were conducted along three paths: (a) environmental path where respondents expressed their views and WTP for environmental restoration and preservation projects, (b) flood control path, and (c) combined path.

Results

Until recently, WTP studies have neglected psychological foundation of WTP and, instead, narrowly focused on demographic variables (Ajzen and Driver 1992). Our study found that the primary socio-demographic variables (respondent income, race/ethnicity, gender, age, dwelling location within the floodplain, and the number of inhabitants in the dwelling) bear weaker relationship with WTP for flood control projects than do variables based on the Theory of Planned Behavior (Ajzen and Driver 1992), specifically subjective norms ($r = 0.29$, $p<0.001$) and an overall index of cognitive structure ($r = 0.4$, $p<0.001$; $r = 0.46$, $p<0.001$, when the belief-evaluation compound items are also multiplied by a separate self-report measure of the importance of the outcome to the decision.

The findings from the survey in the flood control path revealed that:

- There is some evidence that WTP is higher among those at a higher risk of flooding, especially those living in the downstream portions of urban watersheds and those currently residing near but outside of the 100-year floodplain
- Demographic factors (especially income) and measures of environmental attitudes are important determinant of WTP, even after accounting for differential risk factors
- Potential problems with embedding suggests that voters in a hypothetical referendum on flood control may not carefully scrutinize the features of the flood control project when determining their level of support. Rather, given the existing perception at the time of the survey on the flooding problems in Milwaukee County, they may believe that it is important to take some action.

Ethic research

The survey evaluation revealed that strictly economical values such as income, play a comparatively minor role in WTP regression equations relative to psychological variables such as cognitive structure and subjective norms. Cognitive structure is in turn strongly related statistically to environmental attitudes and values. The research focused on evaluation of two types of ethical attitudes: environmental ethic and duty oriented ethics. Results of Griffin's research found that residents' perception of the actual efficacy of the project in bringing about the physical goals ("...help improve the health of the river" in the environmental path, and "...help hold the line against flooding" in the flood control path) were among the most important considerations, especially if they produced enduring benefits such as "...help support a long-term solution" and "...help future generations." Less salient was consideration of whether a flood control project might help people who live in the floodplain. Similarly, respondents on the average rated only as moderately important (i.e., roughly around the middle of a 0 to 10 scale of importance) economic considerations as to whether the project would be personally expensive and whether it would add significantly to one's taxes.

Environmental perspectives can play a significant role in determining cognitive structure and WTP. Not only is a broad measure of environmental beliefs an important influence on cognitive structure, but so is a measure of perceived taxpayer duty toward urban river cleanup as well. Thus key theoretical concepts from the environmental ethic play an important role in formulation of the public's WTP for urban watershed restoration. Those who see the environment in ethical terms appear to have a greater WTP for environmental improvement, at least in the case of urban watersheds.

Economic outcome

The regression results of the Marquette University research revealed that the models explained about 40% of the variation of the latent WTP variable. Clark found that for the ecological restoration value, respondent income does not have a statistically significant influence on real WTP. "Years of education" does positively affect the real WTP, and older respondents have lower real WTP, other factors being equal. Homeowners have higher WTPs, but the coefficient is significant only at the 10% level on one-tailed test. In the protest vote, respondents were unwilling to spend their own funds. The "subjective norms" index, as well as both cognitive structure measures, are positive and significant.

The habitat risk score was positive and significant. In the habitat test the responses were correlated to the quality of the habitat expressed by the index of physical integrity of the nearest section of the water body for which the index was available. This implies that a higher level of habitat quality leads to a higher WTP to pay for reducing ecological risk. This at first seems counterintuitive, since one may believe that higher ecological risk (and lower habitat quality) areas need more clean-up. However, an alternative interpretation is that respondents believed that less environmentally damaged areas require more funding to preserve their environmental integrity than did more damaged areas. That is, WTP is higher before an area is damaged than after the damage has occurred. Finally, the more the respondents visit the river, the higher is the WTP.

Two additional measures were included in this category: *an awareness of consequences*, a Likert-type question related to the belief that taxpayers have a duty to share the cost of improving the health of urban rivers (*taxpayer duty*) and a belief that nature should be preserved for its own sake apart from any human benefits (*biocentric ethic*). Only "taxpayer duty" was statistically significant and was positive. It is an expression of environmental duty rather than biocentric ethics, which apparently is less significant in the urban environment dealing with degraded rivers and favoring restoration rather than nature protection.

Given approximately equal positive valences to the project outcome that would result in flood protection or environmental restoration project, the survey yielded a significantly more heavily weighted compound for beliefs about the efficacy of environmental projects as compared to the flood control projects. This was confirmed by the actual present values calculated by Clark from the willingness to pay that yielded much higher WTP for environmental restoration than flood control. Willingness to pay for environmental restoration was 2.4 times higher than for flood control benefits.

The WTP model developed in the research is transferable to other location, at least in the same geographical region (Alp et al. 2002).

Conclusions

Although the primary drivers for urban stream management for policy makers is primarily flood control, projects that would focus solely on flood control may have negative net benefits and may not be acceptable to the public that appreciates more ecocentric values of stream restoration and preservation. However, because restoration of the water bodies and riparian floodplains and development of best management practices for storage and treatment of runoff (e.g., ponds and wetlands) have also significant flood control benefits, accomplishing both goals has been possible and met (or are proposed to be met) in the investigated watersheds.

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Contingent Valuation and Watershed Management: A Review of Past Uses and Possible Future Applications

Jennifer N. Duberstein, J.E. de Steiguer

Abstract

Contingent valuation is an economic tool used for estimating the value that a person places on environmental goods and services. It is particularly useful for estimating the values of non-market and non-use goods and services. Contingent valuation has a number of possible uses for environmental decision-making such as measuring willingness-to-pay for environmental changes, for risk assessment, in environmental litigation, in policy formulation, and for evaluating investments. Contingent valuation also has possibilities for evaluating watershed management options. This paper examines the uses and limitations of contingent valuation and its possible future applications in watershed management.

Positive aspects of contingent valuation include its hypothetical nature and its ability to measure option, bequest, and existence values. However, among problems associated with contingent valuation are a failure to address global impacts, boundary issues, asymmetric valuation of gains and losses, contingent valuation's hypothetical nature, strategic bidding behavior of respondents, and irrational responses. Many of these drawbacks are important considerations when using contingent valuation for watershed management decisions. Some of these shortcomings may be addressed by use of integrated decision-making, multi-criteria analysis tools, and post-survey debriefing interviews to determine respondent frame of reference.

Contingent valuation has clear values to watershed management, but it also has clear limitations. If conducted correctly, in many situations it can be expected to provide fairly accurate results. For valuing the longer reaching effects of management activities on a watershed, however, contingent valuation results may be less than accurate. In these cases, alternative methods should be explored.

Keywords: contingent valuation method, watershed management, non-use value, stated preference method

Introduction

Contingent valuation is an economic tool used for estimating the value that a person places on environmental goods and services. Contingent valuation is particularly useful for estimating the value of non-market and non-use goods and services. It is referred to as a "stated preference" method of valuation because it involves the survey of personal opinions of value regarding hypothesized, but unrealized, environmental changes.

Researchers interview a sample of the population to be affected by a particular action, and through a series of questions and analyses, estimate the respondent's value of the resource or action in terms of willingness to pay (WTP) or willingness to accept (WTA). Willingness to pay represents the most a person would be willing to pay to keep her quality of life, while willingness to accept represents the minimum that a person would be willing to accept to keep her quality of life at its original level when she loses the good. The average WTP/WTA obtained from the sample is then extrapolated across the entire affected population (often taking into account factors such as income, education level, and other socioeconomic variables) and used as the dependent variable in a regression model.

Contingent valuation has a number of possible uses for environmental decision-making, such as measuring willingness to pay for environmental

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changes (Eisen-Hecht and Kramer 2002), for risk assessment (Fried et al. 1999, Novotny et al. 2000), in litigation (Arrow et al. 1993, World Bank Institute 2002), in policy formulation (He et al. 2002), and for evaluating investments (Ardila et al. 1998). It is also commonly used as a tool in evaluating different watershed management options (Cruz et al. 2000, Pattanayak 2001, Eisen-Hecht and Kramer 2002).

In 1993, a team of economists headed by two Nobel laureates convened to discuss the utility of contingent valuation (Arrow et al. 1993) (the "Arrow-Solow Report"). The result was a set of recommendations for the use of contingent valuation in natural resources. The authors suggest that the more closely their guidelines are followed, the more reliable the results obtained from a contingent valuation study. This report is commonly viewed as a list of best management practices for the assessment of damages caused by environmental disasters, but also has a more general applicability to natural resources as a whole (Carson et al. 1996). Many natural resource contingent valuation studies adhere to the methodology outlined in the 1993 Arrow-Solow Report and use this adherence as an indication of the validity of their study (Eisen-Hecht and Kramer 2002, He et al. 2002).

There are several assumptions one must make in order for contingent valuation to be valid. The first is that the resource to be valued can be described in a scenario that is meaningful to the respondent, and that the respondent understands the resource as the researcher intends it to be understood (World Bank Institute 2002). Maps, computer presentations, and pictures might be used to accomplish this. The second assumption is that there is a payment vehicle: for willingness to pay, the vehicle might be a new user fee. For willingness to accept, it might be a tax refund. The third assumption is that the questioner has a method for measuring the respondent's value of the proposed change. There are three common methods for this. The first is open-ended questioning, where the questioner asks the respondent how much he would be willing to pay, for example, for improved water quality. The second method is iterative bidding. The questioner asks the respondent if he would be willing to pay, for example, $10 more a year for improved water quality. If the respondent answers affirmatively, the questioner continues to increase the bid until the respondent answers no. The third, and most accepted method is dichotomous choice, or referendum, where the questioner offers the respondent a random price, to which she must answer either yes or no. Often a follow-up question or a combination of these methods is used to narrow the willingness to pay (or accept) price range (Carson et al. 1995).

Contingent Valuation Method: Strengths

Contingent valuation has a number of strengths. First, it can be used to value multiple destination recreation trips, as many other non-use valuation tools cannot (Loomis 2002). Second, contingent valuation is hypothetical in nature, so it can be used to measure the effects of an irreversible change without actually making the change (Loomis 2002). Second, it can be used to measure option values, or the value that one places on a resource for the option of having it to use in the future (Loomis 2002). Finally, it is the only method that can measure bequest value, which is the value one places on a resource in order to be able to pass it on to future generations, and existence value, which is the value one places on a resource just for knowing that it exists (Loomis 2002, World Bank Institute 2002).

Contingent Valuation Method: Issues

Despite these strengths, there are a number of problems with contingent valuation. First and foremost, any survey research is susceptible to a variety of different errors. Survey design, including question wording and question order, can all affect accuracy and can cause bias in survey results (Dillman 2000).

Although the Arrow-Solow Report is regarded as the industry standard, there are those who do not agree with the findings. Harrison (2001), for example, asserts that the report is "generally lacking in logic and empirical foundation," and encourages researchers to think important issues through on first principles. The Report does not seem to be infallible, either. Carson et al. (1995) tested the Report's assertion that timing of interviews could have a significant effect on the reliability of survey results. They found this to be untrue, and concluded that willingness to pay measures did not seem to be significantly sensitive to interview timing.

Projects implemented locally can have long reaching effects. Contingent valuation tends to focus its efforts on the communities in which the proposed change will take place and neglects to take into account the effects on a global scale (Westra 2000). But for issues affecting watersheds, these effects can have significant impacts, and contingent valuation will fail to measure them. It is very difficult to place boundaries around environmental issues (O'Neill and Spash 2000), and the population sampled for a contingent valuation survey may not be a good sample of the underlying affected population. So for projects of more than local significance, use of the contingent valuation method is questionable. Asymmetry in valuation of gains and losses, depending on the respondent's point of view, creates interesting issues for contingent valuation (O'Neill and Spash 2000). We tend to put more value on things we already own (willingness to accept), and less value on things we have to purchase (willingness to pay), so depending on which criteria a survey is using, the measure can over- or underestimate the actual value to respondents.

Although the hypothetical nature of contingent valuation is a strength, it is also a weakness (Bateman and Langford 1997, Westra 2000, Morrison 2002, World Bank Institute 2002). The situations described by contingent valuation surveys are not real, and just because something is predicted to occur doesn't mean that it will. A potential Pareto improvement is an economic principle which states that if an action has "winners" (those whose quality of life is improved by the action) and "losers" (those whose quality of live is diminished by the action), it is possible that such a situation could exist where the winners could potentially compensate the losers and still have a gain remaining (Loomis 2002). In reality, just because it is possible for winners to compensate losers, it doesn't mean that they will. Westra (2000) also notes that although contingent valuation surveys are conducted with the caveat that respondents must have a full understanding of the issue at hand, that is seldom the case. Especially in cases that involve environmental risk, economically interested parties have a vested interest in protecting and promoting their products and operations. This can lead to half-truths and misinformation in the survey process.

Strategic bidding is also an issue, and can introduce bias into contingent valuation studies (Kuriyama 1999, Morrison 2002). An underbid may be indicative of the fact that someone isn't willing to state his actual value for a resource because he believes it should be available at no cost. An overbid might represent a respondent's strategy to give a higher than reality price to something in hopes that the inflated response will influence the final results of the survey. Irrational responses, or responses that make no sense, can also confound contingent valuation surveys. Respondents might state a positive willingness to pay for something they think will have no effect (He et al. 2002). They may also not be willing to put a price on something that clearly has a value to them. For instance, if they feel something to be a basic human right, respondents may see it as an act of betrayal to put a verbal price on it, despite the fact that it has value to them (O'Neill and Spash 2000).

Yea saying, or a tendency of the respondent to agree with the interviewer regardless of his true views, can also be an issue (He et al. 2002). Social pressures may motivate some of these responses. Studies have shown that people are less willing to voice their undesirable positions when they are aware that they are being tested (i.e., in a survey situation)(Singleton and Straits 1999). This so-called social desirability effect has the potential to bias the validity of any contingent valuation study.

A further problem with contingent valuation is that it can give respondents unrealistic expectations of what is going to happen. Especially with willingness to accept contingent valuation surveys, respondents can be led to expect that they will receive a payment at some point in the near future, based on their responses to the survey. In practice, it is rarely the case that any payment is actually granted (Westra 2000). Particularly when working with populations in developing and impoverished areas, this "promise" of money that never comes can lead to a decline in trust of researchers and can have negative impacts on the future of conservation efforts in the region. This is an especially important factor to consider for watershed conservation efforts in developing countries.

Contingent Valuation Method: Improvements and Alternatives

There are many ideas for improvement of the valuation of natural resources, but no all-encompassing answers. Integrated decision-making seems to be critical (O'Neill and Spash 2000). An integrated approach that involves all stakeholders, relevant policy makers, and hard scientific evidence in a decision process is critical to gaining an understanding of the true value of a resource (Westra 2000).

Although economically interested parties are stakeholders and should be included in the decision process, in order to make sure respondents receive as accurate and nonpartisan information as possible, disinterested third parties should conduct impact assessments and design and conduct contingent valuation surveys (Westra 2000).

Multi-criteria analysis (MCA) tools take into account the fact that some things aren't measurable by money, and the resource requirements and effects of different watershed management alternatives may be comparable in several different dimensions, but without a single unit of measure (O'Neill and Spash 2000). Analytic Hierarchy Process is an example of MCA. In this process, stakeholders choose preferred management options based on pairwise comparisons of several alternatives. Preferences of respondents for one option over another are the measuring rod instead of dollars. Methods such as this can help to eliminate bias caused by the pressure of having to choose a dollar value.

People's decisions can be based on concerns about legitimate procedures and the fairness of the distribution of burdens and benefits (O'Neill and Spash 2000). This can be independent of concerns about maximizing total welfare. Questions should be worded such that respondents don't compromise their ethical beliefs by giving a truthful answer. Multi-criteria analysis combined with deliberative methods (such as stakeholder focus groups) may help elicit a more accurate value of the resource (O'Neill and Spash 2000).

Although the contingent valuation methodology certainly has drawbacks, there are ways to help counteract these issues. If values placed on a resource depend on the frame of reference of the respondent (i.e., WTP vs. WTA), it becomes very important to elicit not only the respondent's value of the resource, but also his frame of reference. This will help explain why he puts this value on the good in question. It also becomes critical to have demographic data to help estimate the frame of reference of the population as a whole.

Boundary issues can be minimized by using a distance decay function when extrapolating WTP/WTA to an entire population. Rubin et al. (1991) used this method to perform a benefit-cost analysis of the Northern Spotted Owl. According to their paper, WTP decreases with distance from the affected area. Rubin et al. estimated that WTP decreased approximately 10% for every 1,000 miles distance, and used this estimate to extrapolate the value of the spotted owl to their entire affected population (in this case, the United States).

Irrational responses may not be the result of failure to understand the survey, but instead may represent very real indications of willingness to pay, and why respondents are (or are not) willing to pay. Reconsider the example of someone stating a positive willingness to pay for something that she thought would have no effect. There are conceivable situations in which citizens could be willing to contribute to something that they didn't think would necessarily have immediate, on-the-ground effects, but that they thought might influence the future of conservation. According to accepted contingent valuation theory, these irrational responses are discarded with the assumption that the respondent didn't properly understand the survey. It becomes very important to understand exactly why respondents answered as they did. Post-survey debriefing sessions are a necessity if one is to clearly comprehend respondents' understanding of the proposed scenario and the meaning behind irrational responses (Hanemann 1994). In addition, careful survey design and administration can help to alleviate some of the bias-causing mistakes common in survey research.

Contingent Valuation as a Tool for Watershed Management: An Example

There are many examples of the utility of contingent valuation for evaluating different options in watershed management. Generally, a set of alternatives is developed and presented to the public in the form of a contingent valuation survey. Based on public response to the survey, researchers can then perform a cost-benefit analysis to determine which alternative is the most preferred.

The literature is full of a variety of different examples of using contingent valuation for watershed resources. In order to demonstrate this use, we consider a study conducted by Eisen-Hecht and Kramer (2002) in the Catawba Basin, located in North and South Carolina. This study was an analysis of the cost-effectiveness of maintaining the current level of water quality in the Catawba River Basin. Researchers estimated economic benefits using the contingent valuation method by surveying a total of 1,085 area residents. They calculated a mean willingness to pay of $139/household, and compared it with the estimated cost of the proposed management plan over a 10-year period. Using net present value and a variety of discount rates, researchers found that the potential benefits would outweigh the costs by more than $95 million. Researchers used dichotomous choice (referendum) for eliciting a value from respondents. They then used logistic regression to find which social and demographic factors affected respondents' willingness to pay. They asked a wide range of questions, and found that the cost of the management plan had the most direct affect (people were more willing to pay for lower cost management plans), but a number of other variables also had both significant positive and negative effects, such as belief that management plan was likely to succeed, membership to an environmental or conservation organization, whether or not respondents trusted universities, and respondent's household income.

Potential Problems with the Study

Overall, the study conducted by Eisen-Hecht and Kramer reads much like a checklist of how to conduct a contingent valuation study. One potential problem, which in this particular case doesn't seem to have had a negative impact, is that researchers didn't appear to address any larger scale management implications. In a situation like this, which aims to make environmental improvements to an ecosystem, the project may have longer reaching positive effects, and so benefits may have been underestimated. Since this example appears to be a winner, in any case, this omission may not be important. For a project where the decision isn't as clear-cut, however, or where there could be long-reaching negative effects, consideration of all stakeholders, costs, and benefits is vital.

Conclusions

Contingent valuation has clear values to watershed management, but it also has clear limitations. It is a tool that can be used for valuation of natural resources, and if conducted correctly, can be expected to provide fairly accurate results. For valuing the longer reaching effects of management activities on a watershed, however, it can be difficult to include all relevant stakeholders, and contingent valuation results may be less than accurate. In these cases, alternative methods should be explored.

There are a number of means suggested to help counter the shortcomings of contingent valuation methodology. Acknowledgement of all stakeholders, careful survey design and administration, and post-survey debriefings (particularly for examining the reasoning behind irrational responses) are all methods that help improve the process of valuation of watersheds, and the use of contingent valuation as a means for doing so. Most importantly, researchers need to be aware of the limitations of contingent valuation and the knowledge that the situation in which contingent valuation is to be used can have important ramifications for the accuracy of the test. In these situations, integrated decision-making and multi-criteria analysis tools may help make more accurate management decisions.

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