Proceedings of the Fourth Interagency Conference on Research in the Watersheds

Observing, Studying, and Managing for Change

Scientific Investigations Report 2011–5169

U.S. Department of the Interior
U.S. Geological Survey

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p. 1: (Prince William Sound, Alaska). *Don Becker, U.S. Geological Survey*

p. 22: (Prince William Sound, Alaska). *Don Becker, U.S. Geological Survey*

p. 48: Spawning coho (Issaquah Creek, Washington). *Dan Bennett (Any use is licensed under Creative Commons 2.0)*

p. 93: (Taken on flight from Fairbanks to Prince William Sound, Alaska). *Don Becker, U.S. Geological Survey*

p. 107: (Denali National Park, Alaska). *Don Becker, U.S. Geological Survey*

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p. 131: Sampling stream water (Denali National Park, Alaska). *Dennis G. Dye, U.S. Geological Survey*

p. 140: "Another Day at the Office" (Solomon Gulch, Valdez, Alaska). *Johnse S. Ostman, U.S. Geological Survey*

p. 148: Permafrost erosion measurement (along the Arctic coast, Alaska). *Christopher Arp, U.S. Geological Survey*

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Back cover: (Prince William Sound, Alaska). *Don Becker, U.S. Geological Survey* science for a changing world

Proceedings of the Fourth Interagency Conference on Research in the Watersheds

Observing, Studying, and Managing for Change

Edited by C. Nicolas Medley, Glenn Patterson, and Melanie J. Parker

Scientific Investigations Report 2011–5169

U.S. Department of the Interior
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U.S. Department of the Interior
KEN SALAZAR, Secretary

U.S. Geological Survey
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U.S. Geological Survey, Reston, Virginia: 2011

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The extended abstracts and conference papers in this volume that were written by U.S. Geological Survey authors were reviewed and approved for publication by the Survey. Extended abstracts and conference papers submitted by researchers from academia and from State and other Federal agencies are published as part of the proceedings but do not necessarily reflect the Survey's policies and views and might not adhere to USGS editorial standards.

Preface

These proceedings contain the abstracts, manuscripts, and posters of presentations given at the Fourth Interagency Conference on Research in the Watersheds—Observing, Studying, and Managing for Change, held at the Westmark Hotel in Fairbanks, Alaska, September 26–30, 2011. The conference was jointly hosted by the Bureau of Land Management and the National Park Service.

Watersheds face resource impacts driven by accelerated change related to land use, population, and climate. About every three years a conference is held to bring together watershed researchers, observers, and managers to share scientific advances and management strategies. This year, the Fourth ICRW took a wider perspective on watershed science and examined some pressing issues of watershed science and management in our largest and perhaps most vulnerable state, Alaska. The purpose of the conference was to better understand the processes driving change and help managers incorporate societal needs and scientific uncertainty in the management of natural resources.

The conference echoed similar themes to the last, highlighting the challenges of managing watersheds based on available science when considerably uncertainty remains regarding the hypothesized relationships between observed environmental changes and their ultimate effects. For example, while the scientific case for anthropogenic climate change has been well presented, confirming possible cause and effect relationships between climatic change and physical and ecological impacts in highly variable, natural systems continues to represent a scientific challenge. This goal becomes even more difficult when superimposed upon a long history of natural resource and land management practices that have fundamentally changed the physical, chemical and biological processes important in maintaining naturally functioning ecosystems. Designing and implementing studies to better understand watersheds and clearly communicating the findings to decisionmakers will be the primary challenge for natural resource scientists and managers into the foreseeable future.

The decision to hold the Fourth ICRW in Alaska was intended to highlight these challenges, and specifically the potential changes associated with climate change. The relatively pristine environment of Alaska perhaps provides the best opportunity to confirm predicted cause and effect relationships associated with climate change, absent the confounding influence of anthropogenic disturbances characteristic of more populated areas.

The Fourth ICRW presented a mix of science and management. The conference was loosely structured to reflect important aspects of the scientific process that lead to scientifically defensible management decisions. Abstracts for presentations or posters were invited on the following topics, related to both pristine and altered watersheds:

1. Observing Watersheds: Inventory and monitoring. Instrumentation. Study design and implementation. Remote sensing. Data collection and management.

2. Studying Watersheds: Hypothesis testing and experimental approaches. Results, analysis and interpretation of data. Trends, relationships, causes and effects and predictions. Ecosystem modeling. Cumulative and indirect effects. Potential drivers of change including climate, land use, population growth and water demand, wildfire, disease, insects, agriculture, mining, and energy development. Impacted goods and services including quantity and quality of water, biota and habitats, controls on carbon, nutrient, and sediment fluxes, and aesthetics. Opportunities for education and recreation.

3. Managing Watersheds for Change: Approaches to adaptive management. Incorporation of uncertainty in decisionmaking. Prioritization of management actions. Collaborative approaches. Ecosystem restoration and strategies for adaptation, mitigation, and enhancement of resiliency.

The Fourth ICRW conference program contained several major elements that highlighted the theme of the conference; Federal agency representative science policy presentations; technical talks and poster presentations; invited speaker keynote addresses; a selection of all-day field trips in the Fairbanks area; and a plenary session.

All presenters were invited to report on their professional and personal experience with watershed research and management during the technical talks and poster presentations of the conference. Each was also invited to provide an abstract, manuscript, or poster for inclusion in the conference proceedings. The contributions contained within these proceedings do not include all presentations given at the conference. Only those submitted by presenters that met the U.S. Geological Survey Scientific Investigations Report peer review and agency approval standards were published. The organizers thank the many people who contributed manuscripts and abstracts to these proceedings and to the attendees whose interest and energy made the Fourth Interagency Conference on Research in the Watersheds a success.

Looking ahead, the Fifth ICRW will be hosted by the Center for Watershed Sciences, Southern Research Station, U.S. Forest Service and will be held at the Santee Experimental Forest located in Charleston, South Carolina, in September or October 2014, to coincide with the 50th anniversary of watershed research at the facility. Updates and information will be made available on the conference website *http://www.hydrologicscience.org/icrw/*. We look forward to seeing you there.

Proceeding Editors

C. Nicolas Medley
National Park Service

Glenn Patterson
National Park Service and
U.S. Geological Survey liaison

Melanie Parker
U.S. Geological Survey

Fourth Interagency Conference on
Research in the Watersheds

Acknowledgments

Conference Chair

Jim Renthal, Bureau of Land Management
Bill Jackson, National Park Service, Water Resources Division

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Charles Noss, U.S. Environmental Protection Agency
Glenn Patterson, National Park Service, Water Resources Division
Paul F. Schuster, U.S. Geological Survey
Jade Soddell, U.S. Bureau of Reclamation
Rob Striegl, U.S. Geological Survey
Carl C. Trettin, U.S. Forest Service
Rick Webb, U.S. Geological Survey

Permafrost Workshop

Torre Jorgenson, University of Alaska, Fairbanks
Misha Kanevskiy, University of Alaska, Fairbanks
Yuri Shur, University of Alaska, Fairbanks

Keynote Speakers

Jeff Arnold, U.S. Army Corps of Engineers, Institute of Water Resources, Senior Scientist
Jessica Cherry, University of Alaska, Fairbanks, International Arctic Research Center
Ellen Wohl, Colorado State University
Susan Moran, U.S. Department of Agriculture, Agricultural Research Service, Southwest Watershed Research Center
Rob Striegl, U.S. Geological Survey
Scott Guyer, North Slope Science Initiative

Welcome Address

Jon Waterhouse, Yukon River Inter-Tribal Watershed Council
Bob Winfree, National Park Service, Alaska Regional Science Advisor, Anchorage

Agency Presentations

Charles Noss, U.S. Environmental Protection Agency
Jerad Bales, U.S. Geological Survey
Ben Kennedy (tentative), Bureau of Land Management
Mark Walbridge, U.S. Department of Agriculture, Agricultural Research Service
Rick Hooper, Consortium of Universities for the Advancement of Hydrologic Science, Inc.
Andy Loranger (tentative), U.S. Fish and Wildlife Service
Deb Hayes and Mary Beth Adams, U.S. Forest Service
Gary Machlis (tentative), National Park Service

Session Moderators

Ben Kennedy—Alaskan Water Quality
Don Campbell—Climate and Hydrology
C. Nicolas Medley—Ecohydrology
Susan Moran—Cargon, Nitrogen, and Agriculture
Scott Davis—Forested Watersheds
Jim Renthal—Lakes, Wetlands, and Soil Moisture
Mary Beth Adams—Water Quality Monitoring
Chuck Noss—Data and Modeling
Rick Webb—Geomorphology and Watershed Management

Field Trip Leaders

Denali National Park (preconference)—Denny Capps, National Park Service
Permafrost Tunnel—Tom Douglas, U.S. Army Corps of Engineers
Yukon Crossing—Dennis Keill, Bureau of Land Management (retired); Paul Schuster, U.S. Geological Survey; Hans Arnett
Bonanza-Poker Creeks—Jay Jones and Kenji Yoshikawa, University of Alaska, Fairbanks
Chena Geothermal Springs—John Schaake, U.S. Army Corps of Engineers; Bernie Karl, Chena Lake Hot Springs Resort (owner)

Proceedings Editors

C. Nicolas Medley, National Park Service, Water Resources Division
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Melanie Parker, U.S. Geological Survey

Proceedings Layout and Design

Mari L. Kauffmann, (Contractor, ADC Management Services, Inc.)
Melanie Parker, U.S. Geological Survey

The organizers would like to thank the many unnamed people, including the panel presenters, who helped in so many ways to make the conference a success.

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Conference Papers and Extended Abstracts

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26–30 September 2011, Fairbanks, Alaska

C. Nicolas Medley, Glenn Patterson, and Melanie J. Parker, editors

Conference Program

Preconference Permafrost Workshop

Monday, September 26, 8:00 AM

Building Management Competency in Permafrost Terrain—Torre Jorgenson, University of Alaska, Fairbanks, AK

Monday, September 26

4:00 PM	Arrival / Check in / Registration
6:30 PM	Opening Reception—Hors d'oeuvres and no-host bar
7:00 PM	Welcome Speech—Jon Waterhouse, Yukon River Inter-Tribal Watershed Council
7:30 PM	Reception continues

Tuesday, September 27

8:00 AM	Call to Order and Orientation—Jim Renthal, Bureau of Land Management
8:10 AM	Welcome—Bob Winfree, Alaska Regional Science Advisor, National Park Service
8:30 AM	Introduction of Keynote Speaker, Jeff Arnold—Tom Douglas, U.S. Army Corps of Engineers
8:40 AM	First Keynote Address—Jeff Arnold, Senior Scientist at the U.S. Army Corps of Engineers Institute of Water Resources
9:45 AM	Break

Agency Presentations

10:00 AM	Chuck Noss, U.S. Environmental Protection Agency
10:30 AM	Jerad Bales, U.S. Geological Survey
11:00 AM	Ben Kennedy (tentative), Bureau of Land Management
11:30 AM	Mark Walbridge, Agricultural Research Service
12:00 PM	Lunch
	Second Keynote Address—Hydroclimate of the Seward Peninsular in Western Alaska: A Century of Change—Jessica Cherry, International Arctic Research Center, University of Alaska, Fairbanks, AK

Tuesday, September 27 (Continued)

	Concurrent Session I		
	Session 1A: Alaskan Water Quality Moderator: Ben Kennedy	**Session 1B: Climate and Hydrology** Moderator: Don Campbell	**Session 1C: Ecohydrology** Moderator: C. Nic Medley
1:30 PM	Hydrogeochemistry of Antimony and Arsenic in Watersheds of the Kantishna Hills District, Denali National Park and Preserve, Alaska—T.P. Trainor	Potential Climate Change Effects on Water Tables and Pyrite Oxidation in Headwater Catchments in Colorado—R.M.T. Webb	Predicting the Impact of Glacier Loss on Fish, Birds, Floodplains, and Estuaries in the Arctic National Wildlife Refuge—M. Nolan
1:55 PM	Use of Pore Water as Part of Contaminated Sites Management: Case Studies in Kotzebue and Fairbanks, Alaska—J. Fish	Utilizing Long-Term ARS Data to Compare and Contrast Hydroclimatic Trends from Snow and Rainfall Dominated Watersheds—D.C. Goodrich	Land Use and Salmon Habitat: A Comparison of North Pacific Watershed Parameters—S.F. Loshbaugh
2:20 PM	Assessment of High-Flow Events and Water Quality in the Birch Creek and Nome Creek Placer-Mined Watersheds near Central, Alaska, 2008–2010—B.W. Kennedy	Analysis of Trends in Climate, Streamflow, and Stream Temperature in North Coastal California—M.A. Madej	Intrinsic Potential: A Tool for Identifying Salmon Habitat at the Watershed Scale—G.H. Reeves
2:45 PM	Application of Updated Approaches for the Reclamation of Placer-Mined Lands in the Harrison Creek Watershed near Central, Alaska—H.R. Arnett	Evidence of Climate Change in the Streamflow and Water Temperature Record in the Missouri River Basin—M.T. Anderson	Evaluating Biodiversity Response to Forecasted Land Use Change: A Case Study in the South Platte River Basin, Colorado—W.G. Kepner
3:10 PM	Break	Break	Break
3:40 PM	Developing a Long-Term Aquatic Monitoring Network in a Complex Watershed of the Alaskan Arctic Coastal Plain—M.S. Whitman	Long-Term Climate Change Controls Stream and Riparian Area Response to Disturbance in the Semiarid Great Basin—J.C. Chambers	Modeling Impacts of Environmental Change on Ecosystem Services across the Conterminous United States—P. Caldwell
4:05 PM	Permafrost and Active Layer Dynamics Inferred from Major Element Geochemical Signatures in Six Arctic Alaskan Rivers—T.A. Douglas		The Urban Fishery: An Application of System Robustness—M.B. Krupa
4:30 PM	Monitoring Water Quality in Alaskan National Parks: Development and Application of RIVPACS Empirical Models for the Assessment of Ecological Condition—T. Simmons		Quantifying Uncertainty in Ecosystem Studies (QUEST): Using Long-Term Data from Small Watersheds—M.B. Adams

Tuesday, September 27 (Continued)

	Poster Session and Reception		
5:30 PM	Hors d'oeuvres and no-host bar		
	Using Ground-Based Geophysics in the Yukon Flats, Alaska, to Characterize Permafrost and Subsurface Features Critical to Hydrologic Models of Subarctic Systems—H.R. Best	Restoring and Protecting a National Treasure: An Overview of the Chesapeake Bay Program—A.S. Burnett	Making Long-Term Watershed Research Accessible—K.J. Cole
	Building a Hydrology Science Plan for the Arctic Landscape Conservation Cooperative—B.T. Crosby	Renewable Energy Locations for Existing and Potential Facilities within BLM Leased Land—S. Davis	An Investigation of Carbon Dynamics in Beaver Creek, Alaska, Using in situ Sensors—M.M. Dornblaser
	Influence of Organic Matter on *Cryptosporidium parvum* Oocyst Mobility in Watersheds Characterized by Variable-Charge Soils—R.W. Harvey	Evaluating Cumulative Effects of Logging and Potential Climate Change on Dry-Season Flow in a Coast Redwood Forest—J. Lewis	The Yukon River Basin Indigenous Observation Network: Preliminary Results from Baseline Datasets Highlighting Climate Variation Indicators—L.M. Mackey
	Outreach in the Yukon River Basin: Connecting Science and Community—P.F. Schuster	Using Remotely Sensed Brightness Temperatures to Infer Snowmelt Onset and Runoff in High Latitude Drainage Basins—K.A. Semmens	Modeling Coastal Plain Drainage Ditches with SWAT—A.M. Sexton

Wednesday, September 28

6:00 AM Breakfast service begins at Northern Lights Restaurant in Westmark Hotel

Field Trips (Box Lunch)

More information on p. 9.

7:00 AM Yukon Crossing

Caribou-Poker Creeks Research Watershed—Bonanza Creek Long-Term Ecological Research (LTER) Program

Chena River Flood Diversion Project and Chena Lake Hot Springs Resort—Geothermal Energy

5:30 PM Return to Fairbanks

Thursday, September 29

8:00 AM Third Keynote Address—Steep Streams: What's New, What's Problematic—Ellen Wohl, Colorado State University, Fort Collins, CO

Thursday, September 29 (Continued)

	Concurrent Session II		
	Session 2A: Carbon, Nitrogen, and Agriculture Moderator: Susan Moran	**Session 2B: Forested Watersheds** Moderator: Scott Davis	**Session 2C: Lakes, Wetlands, and Soil Moisture** Moderator: Jim Renthal
9:00 AM	Seasonality in Water, Carbon, and Nitrogen Fluxes from an Upland Boreal Catchment Underlain by Continuous Permafrost—J.C. Koch	Quantifying Hydrologic Impacts of Vegetation Treatments in the Bates Creek Watersheds—G.B. Paige	Classification, Mapping, and Management of Wetlands in Alaskan Watersheds with Rapidly Growing Populations—M. Gracz
9:25 AM	Improved Nitrogen Management Utilizing Ground-Penetrating Radar: A Nine-Year Investigation—T. Gish	Long-Term Forest Management and Climate Effects on Streamflow—S.G. Laird	Will the Arctic Coastal Plain Wetlands Disappear?—A.K. Liljedahl
9:50 AM	Water, Energy, and Carbon Flux Observations from Agricultural Research Service Watersheds and Agro-Ecosystem Experimental Sites—J. Alfieri	Effects of Forest Cover and Environmental Variables on Snow Accumulation and Melt—M. Dobre	Measuring Soundscapes Relative to Global Change in Interconnected Wetlands and Uplands along North American Environmental Gradients—W. Sadinski
10:15 AM	Integrating Watershed- and Farm-Scale Models to Target Critical Source Areas While Maintaining Farm Economic Viability—T. Veith	Headwater Variability across the Rain-Snow Transition in California's Sierra Nevada: Stream Discharge, Runoff Timing, and Sediment Yield—C.T. Hunsaker	Lake Districts of the Koyukuk National Wildlife Refuge—K. Lehmkuhl Bodony
10:40 AM	Break	Break	Break
11:00 AM		Contrasts in Carbon and Nitrogen Ecosystem Budgets in Adjacent Norway Spruce and Appalachian Hardwood Watersheds in the Fernow Experimental Forest, West Virginia—M.B. Adams	Using SAR to Characterize the Winter State of Ponds and Lakes In Arctic Alaska—J. Grunblatt
11:25 AM		Whole Watershed Manipulations of Deposition Chemistry at Bear Brook Watershed in Maine and the Fernow Experimental Forest, West Virginia—M.B. Adams	Reclamation Activities in the Climate Change and Water Working Group (CCAWWG)—L. Brekke

12:15 PM Lunch

Fourth Keynote Address—Response of Southwestern Grasslands to Precipitation and Temperature Extremes of the Early 21st Century Drought—Susan Moran, Agricultural Research Service, Southwest Watershed Research Center, Tuscon, AZ

Thursday, September 29 (continued)

	Concurrent Session III		
	Session 3A: Water Quality Monitoring Moderator: Mary Beth Adams	**Session 3B: Data and Modeling** Moderator: Chuck Noss (tentative)	**Session 3C: Geomorphology and Watershed Management** Moderator: Rick Webb
1:30 PM	Water Quality Effects of Fire Retardant Application in a Tundra Lake near Hughes, Alaska—B.W. Kennedy	Hydrologic Response to Climate Change and Habitat Resiliency Illustrated Using Fine-Scale Watershed Modeling—L.E. Flint	Temporal and Spatial Distribution of Landslides in the Redwood Creek Basin, Northern California—M.A. Madej
1:55 PM	Long-Term Pesticide Volatilization Monitoring at the Catchment Scale—J.H. Prueger	Simulation of Hydrologic Response to Climate and Landscape Change Using the Precipitation Runoff Modeling System in the Apalachicola–Chattahoochee–Flint River Basin in the Southeastern United States—J.H. LaFontaine	Holocene Extraordinary Floods and the Social Impacts in the Weihe River Basin, China—C. Huang
2:20 PM	Reconnaissance Investigation of Emerging Contaminants in Effluent from Wastewater Treatment Plant and Stormwater Runoff in the Columbia River Basin—J. Morace	Parameterization of the Yukon River Basin Using the Data of Wolf Creek Research Watershed—O. Semenova	Paleochannels as Reservoirs for Watershed Management—R.P. Gupta
2:45 PM	Monitoring and Modeling Hydrologic and Water Quality Responses to Disturbance Resulting from Natural Gas Development in Muddy Creek, Wyoming—S.N. Miller	Spatiotemporal Analyses of Simulated Biophysical Processes in the Chippewa River Watershed, Minnesota—A.A. Jaradat	The Effectiveness of Aerial Hydromulch as an Erosion Control Treatment in Burned Chaparral Watersheds, Southern California—P.M. Wohlgemuth
3:10 PM	Break	Break	Break
3:40 PM	Pesticide Monitoring in Watersheds: Balancing Cost and Uncertainty—E.A. Pappas	Instrumentation of Crow Creek Watershed for Short- and Long-Term Hydrologic Studies in the Northern Intermountain West—G.B. Paige	The National Watershed Boundary Dataset (WBD), a Framework for all Watershed Science—K.M. Hanson
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4:30 PM	Use of Early Agency Coordination to Efficiently Navigate the Permitting Process for Complex Stream- and River-Related Projects—H.R. Arnett		Spatiotemporal Analysis of Surface and Subsurface Soil Moisture for Remote Sensing Applications within the Upper Cedar Creek Watershed—G.C. Heathman

Thursday, September 29 (continued)

7:00 PM	Banquet and Awards
	Fifth Keynote Address—Research on the Yukon River—Rob Striegl, U.S. Geological Survey, Boulder, CO

Friday, September 30

Agency Presentations

7:45 AM	Jennifer Arrigo, Consortium of Universities for the Advancement of Hydrologic Science, Inc., Senior Program Manager
8:15 AM	Jeff Adams, U.S. Fish and Wildlife Service
8:45 AM	Deb Hayes and Mary Beth Adams, U.S. Forest Service
9:15 AM	Break
9:30 AM	National Park Service (TBA)
10:00 AM	Panel Discussion—Approaches and Tools for Future Management Challenges
10:40 AM	Looking Past and Forward—Rick Webb, U.S. Geological Survey
11:00 AM	Plans for the 5th ICRW—Chuck Noss, U.S. Environmental Protection Agency
11:15 AM	Lunch
	Sixth Keynote Address—Current Activity of the North Slope Science Initiative—Scott Guyer, North Slope Science Initiative
12:30 PM	Final Comments and Adjournment of the 4th ICRW—Jim Renthal, Bureau of Land Management

Field Trip Information

Yukon Crossing

The group will visit the Permafrost Tunnel (16 miles north of Fairbanks near Fox and Goldstream Creek), built by the U.S. Army Corps of Engineers–Cold Regions Research and Engineering Laboratory, nearly 50 years ago. The entrance is representative of the typical seasonally frozen soil of the Fairbanks area but the interior of the tunnel is permanently frozen and does not require structural support.

Next we'll visit the Erickson Creek fire (2003) which burned 118,000 acres and discuss Alaska fire ecology and issues related to buried and elevated pipelines at the ecosystem-permafrost transition. We will view differences between south- and north-facing slopes on the landscape along the 140 miles of the Elliott and Dalton Highways between Fairbanks and the Yukon River (a river nearly 1,900 miles in length from Canada to the Bering Sea.

7:00 AM	Depart from Westmark Hotel parking lot; traveled north on the Elliott Highway
7:30–8:30 AM	Stop at the Trans-Alaska Oil Pipeline and continued north on Elliott Highway to view more of the Pipeline (caribou not guaranteed in tour)
8:30–9:30 AM	Visit Hess Creek overlook, 21 miles from Fairbanks on Elliott Highway. View past wildfire: Erickson Creek fire, 2003, burned 118,000 acres.
9:30–10:30 AM	34 miles on Elliott Highway—View Trans-Alaska Oil Pipeline and discuss buried and elevated pipelines at the ecosystem-permafrost transition.
10:30 AM–12:00 PM	18 miles north on Elliot Highway—View black spruce forest at Tolovana River Bridge
	13 miles north on Elliott Highway (84 miles from Fairbanks)—Livengood (Elliott Highway becomes the Dalton Highway/Haul Road) Observe a fire ecology photo stop (29 miles north of Livengood). See the Bureau of Land Management publication *Birds of the Dalton Highway*.
12:00–1:00 PM	56 miles north on Dalton Highway, close to the Yukon River crossing—Hotspot Cafe for box lunch

Presentation—Success Story on Intergovernmental Coordination for Complex Stream and River Related Projects: Obtaining the Permitting Process by Early Involvement of Regulatory Agencies—Hans Arnett and Sara Lindberg

1:00–2:30 PM	Visit Yukon Bridge viewing station and Yukon River crossing. Another good fire ecology site is north of the river.
2:30–5:30 PM	Return 140 miles to Fairbanks, Westmark Hotel

Field Trip Information

Caribou-Poker Creeks Research Watershed
Bonanza Creek Long-Term Ecological Research (LTER) Program

The purpose of this field trip is to learn about ecological research on taiga and floodplain forests including the influence of slope aspect, the absence or discontinuity of permafrost and its thaw depths, leaf litter, seeds, fire effects, nutrient cycling, carbon cycles, insects and disease infestations, hydrologic effects, changes from vertebrate herbivores, and the successional, physical, and biogeochemical changes on landscapes and species growth. Tied to these topics, the group will address the influences and exchanges of methane, carbon dioxide, water, and energy and how they may influence climate. Degradation of oil spills on organic soils and their vegetative recovery and resilience will be discussed.

7:00–7:45 AM	Depart from Westmark Hotel; traveled 13 miles to the Bonanza Creek Experiment Station
7:45–10:00 AM	Talk on the diversity of the LTER project and long-term databases
10:00–11:00 AM	Travel to the Trans-Alaska Oil Pipeline. View and discuss buried and elevated pipelines at the ecosystem-permafrost transition
11:00 AM–12:00 PM	Visit Permafrost Tunnel (16 miles north of Fairbanks)

Note: The Permafrost tunnel was built by the U.S. Army Corps of Engineers–Cold Regions Research and Engineering Laboratory. Late Pleistocene syngenetic permafrost is exposed in the walls and ceiling. It consists of ice- and organic-rich silty sediments penetrated by ice wedges. Evidence of long-continued syngenetic freezing under cold-climate conditions includes the dominance of lenticular and microlenticular cryostructures throughout the walls, ice veins and wedges at many levels, the presence of undecomposed rootlets, and organic-rich layers that reflect the former positions of the ground surface. Fluvio-thermal modifications are indicated by bodies of thermokarst-cave ('pool') ice, soil and ice pseudomorphs, and reticulate-chaotic cryostructures associated with freezing of saturated sediments trapped in underground channels.

12:00–1:30 PM	Travel to the Caribou-Poker Creeks Research Watershed (31 miles north of Fairbanks) for lunch
1:30–4:30 PM	Visit Caribou-Poker Creeks Research Watershed

Here we will review a variety of watershed, fire, and forest succession conditions from south and north slopes and within floodplains. Available information on stream macroinvertebrates, benthic organic materials, flow rates, water/streambed temperatures, stream and groundwater chemistry, denitrification rates, soil nitrogen pools, the National Atmospheric Deposition Program, and snow surveys for the Caribou-Poker Creeks Research Watersheds will be discussed.

4:00–5:00 PM	Return to Fairbanks, Westmark Hotel

Field Trip Information

Chena River Flood Diversion Project and Chena Lake Hot Springs Resort—Geothermal Energy

The group will visit with Bernie Karl to hear the story of using geothermal energy to create water reaching about 165°F for heating and to generate electricity. After Karl purchased the resort in 1998, geothermal energy reinvented the way energy was consumed, replacing the 1,000 gallons of diesel fuel that had been used every month. Many of Karl’s ideas stem from a desire to find alternative ways to use and reuse resources he already has at his fingertips. After acquiring the 400-acre resort, Karl began trapping water from the underground hot springs, which produce enough power to heat the facility's greenhouses year-round. We also will see a portable geothermal power plant that has saves over $200,000 a year and has partnered with the U.S. Department of Energy that funded half of a $1.4 million exploration project to find and characterize the geothermal resources at Chena Hot Springs.

7:00–10:30 AM	Depart Fairbanks; travel to flood control project Stop before Chena Lake—U.S. Army Corps of Engineers personnel will lead discussion of the project’s function, leakage problems and corrective actions. We will discuss the history of the 1967 flooding of Fairbanks.
10:30–11:30 AM	Travel to Chena Lake Hot Springs Resort.
11:30 AM–12:30 PM	Lunch at restaurant at Chena Hot Springs, lunch at restaurant, Bernie and Connie Karl, owners of resort, will lead discussions and a tour. Tour includes an overview of the geothermal power plant, geothermal exploration project, ice museum and absorption chiller, and greenhouses and gardens.
12:30–3:00 PM	Continue tour of geothermal renewable energy development, geothermal exploration project, reservoir engineering, geology, heatflow, alternative energy, and ice museum.
3:00–5:00 PM	Travel to the Trans-Alaska Oil Pipeline. View and discuss buried and elevated pipelines at the ecosystem-permafrost transition

Note: The Permafrost tunnel was built by the U.S. Army Corps of Engineers–Cold Regions Research and Engineering Laboratory. Late Pleistocene syngenetic permafrost is exposed in the walls and ceiling. It consists of ice- and organic-rich silty sediments penetrated by ice wedges. Evidence of long-continued syngenetic freezing under cold-climate conditions includes the dominance of lenticular and microlenticular cryostructures throughout the walls, ice veins and wedges at many levels, the presence of undecomposed rootlets, and organic-rich layers that reflect the former positions of the ground surface. Fluvio-thermal modifications are indicated by bodies of thermokarst-cave ('pool') ice, soil and ice pseudomorphs, and reticulate-chaotic cryostructures associated with freezing of saturated sediments trapped in underground channels.

5:00–5:30 PM	Return to Fairbanks, Westmark Hotel

Alaskan Water Quality

Use of Pore Water as Part of Contaminated Sites Management: Case Studies in Kotzebue and Fairbanks, Alaska

James Fish

Abstract

Pore water is generally defined as water between grains of sediment, but it can have its origin as either surface water or groundwater. For contaminated sites management in Alaska, pore water is often more specifically defined as saturated zone water in the subsurface that is hydrologically connected to surface water. It can serve as a transport mechanism between contaminated groundwater and surface water bodies and thus can be both an ecological and human health exposure pathway to contaminants. In addition to soil and groundwater sampling and analyses for site assessments, pore water sampling is being used at two contaminated sites under management by the Alaska Department of Environmental Conservation's Contaminated Site Program to evaluate contaminant exposure pathways and regulatory compliance. In Kotzebue, pore water data indicate that contaminant plumes originating from subsurface soils and groundwater and containing diesel-range organic petroleum hydrocarbons (DRO) and benzene, toluene, ethylbenzene, and xylene (BTEX) compounds are reaching both Kotzebue Sound and Kotzebue Lagoon. In Fairbanks, subsurface chlorinated solvents from a former dry cleaner are migrating to the Chena River, causing an apparent water quality violation. This presentation will present data on the two case studies that use pore water to understand exposure pathways as part of risk evaluation and regulatory compliance for contaminated site management.

Keywords: pore water, hyporheic zone, contaminated sites, groundwater–surface water interface

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Introduction

Assessment of a contaminated site typically involves delineation of soil and groundwater contamination. Contaminated groundwater is often delineated to identify a source and a leading edge of a contaminant plume following predominant groundwater flow direction and gradient. Accordingly, a groundwater cleanup point of compliance is normally placed downgradient of the leading edge of a contaminant plume to meet Alaska Department of Environmental Conservation (ADEC) groundwater cleanup levels (see Table C of 18 AAC 75.345; Alaska Department of Environmental Conservation 2008 b) and to ensure the protection of surface water or sediments (see 18 AAC 70; Alaska Department of Environmental Conservation 2009). In cases where contaminated groundwater potentially meets surface water, consideration must also be given to surface water quality criteria and standards as part of contaminated sites management. Groundwater discharge to surface water is often subsurface flow within hyporheic, hypolentic, or subtidal zones (Conant 2000, U.S. Environmental Protection Agency 2008, Environment Agency 2009). It is governed by advective flow and is under strong influence of geomorphology, fluvial (or tidal) processes, and subsurface strata permeability (Conant 2000, Environment Agency 2009). Contaminants in groundwater that discharges into surface water may ultimately be diluted by large volumes of receiving waters or by continuous fluvial processes, masking their exposure potential (e.g., being sediment-bound or bioaccumulating in organisms). The groundwater–surface water interface (GSI), generally defined as the location in the saturated zone containing a mixture of both surface water and groundwater (e.g., hyporheic flow), can be a "sentinel" monitoring location to assess contaminated groundwater movement into surface water bodies (Conant 2000, U.S. Environmental Protection Agency 2008, Environment Agency 2009). Depending on collection method, pore water may represent a snapshot of groundwater–surface water exchange or a time-integrated average of predominant subsurface flow (Kalbus et al. 2006). The proportion of groundwater to surface water may also change seasonally depending on location and aquifer characteristics. Prior to establishing a point of compliance for contaminated groundwater cleanup or monitoring, pore water data is often useful to assess GSI dynamics that may affect the fate and transport of contaminants in the subsurface, as well as to evaluate contaminant exposure pathways to human and ecological receptors.

The presence of contaminants in pore water has been problematic for regulatory agencies not only because pore water is not exclusively groundwater, but also because it does not fulfill the regulatory definition of surface water nor qualify as a surface water point of compliance (i.e., just below the water surface). It is, however, included in sediments regulated under water quality standards (Alaska Department of Environmental Conservation 2009). ADEC recently used pore water data to demonstrate migration of petroleum contaminants to Kotzebue Sound and Kotzebue Lagoon and chlorinated solvents to the Chena River in Fairbanks, resulting in an updated policy and interdivisional cooperative approach to assessing contaminant fate and transport associated with contaminated sites and impaired water bodies (Alaska Department of Environmental Conservation 2011).

Kotzebue Area-Wide

ADEC began work in 2008 to understand area-wide contamination and exposure risks in Kotzebue at both the city and airport locations. Widespread (>10 hectares) petroleum contamination in groundwater exists beneath the city from fuel releases near the Kotzebue Elementary School and former hospital, dating back to the 1970s or earlier. Such large-scale soil and groundwater contamination resulted from releases of approximately 100,000 to 200,000 gallons of heating fuel oil spilled over a period of many years, assumed to be largely from an underground fuel transfer pipeline near the school and former hospital (Janzen and Kane 1987). During the late 1980s, more than 50 monitoring wells and two fuel recovery systems were installed in the area of the fuel releases. Although approximately 40,000 gallons of fuel were reportedly recovered by 1986 (Janzen and Kane 1987), contamination continues to persist in the subsurface. Hydrocarbon sheens observed on Kotzebue Sound have been reported periodically to ADEC, suggesting contaminant migration from groundwater to surface water.

During summer 2010, the Alaska Department of Transportation and Public Facilities began installing approximately 4,400 linear feet of a sheet pile seawall and making substantial roadway improvements to expand Shore Avenue (a roadway that parallels the shore of Kotzebue Sound). The effects of the seawall installation on groundwater discharge dynamics and contaminant transport are unknown. Multiple contaminated sites are also located along the paved apron of the Ralph Wien Memorial Airport in Kotzebue. Fuel tank upgrades begun in the mid 1990s at the airport revealed petroleum contamination in groundwater that remains today. Little is known about the prevailing hydrogeological conditions and groundwater dynamics or about the fate and transport of subsurface contamination at the airport.

Pore water data was used to assess potentially complete exposure pathways that remain from historic citywide fuel releases, as well as exposure pathways that may exist at the airport, and to better understand the risks to human health and ecological receptors. Pore water was also used to establish baseline information of groundwater–surface water interactions and contaminant migration into both Kotzebue Sound and Kotzebue Lagoon.

Wendell Avenue, Fairbanks

Historical releases of tetrachloroethene (PCE) from former dry cleaning operations have occurred at 314 Wendell Avenue in Fairbanks, likely as onsite spills and solvent disposal to wood stave underground sewer lines. Two sewer lines exit the building on either side and connect to a sewer main under Wendell Avenue. The greatest soil contaminant concentrations have been found just west of the building, beneath the driveway, and as well as near a known splintered sewer line in this area. Site investigations also suggests there is significant contamination underneath the building itself and in a smaller source zone near the sewer line east of the building. Releases and disposal have created two vadose zone source areas of PCE-contaminated soil, as well as a groundwater plume containing chlorinated solvents that extends downgradient in a northwesterly direction towards the Chena River. Soil contamination, at concentrations above ADEC soil cleanup levels for chlorinated solvents, extends from the ground surface to the water table at approximately 8 to 10 feet below ground surface (bgs), with greatest contaminants levels at 2 to 4 feet bgs (Oasis Environmental 2009, 2010).

The primary contaminants of concern in both soil and groundwater are PCE, trichloroethylene (TCE), cis-1, 2-dichloroethylene (cis-DCE), and trans-1, 2-dichloroethylene (trans-DCE). Vinyl chloride (VC) has not been detected in soil or groundwater, but chlorinated ethanes are also present in the subsurface. Site characterization activities identified the primary contaminant exposure pathways as inhalation (of outdoor air and indoor air via vapor intrusion), subsurface soil contact and ingestion, groundwater contact and ingestion, and possibly sediment and surface water contact and ingestion (Oasis Environmental 2009, 2010). A subslab depressurization (SSD) system is currently installed within the building at 314 Wendell Ave. to mitigate vapor intrusion and remediate soil contamination beneath the building by creating a vacuum-induced negative pressure beneath the foundation slab. A soil vapor extraction (SVE) system is currently under construction to address shallow soil contamination through volatilization and vapor removal from the two known source zones exterior to the building. In both systems, a series of extraction wells installed in the source areas draw chlorinated solvent vapors from the soil pore spaces and exhaust them to the atmosphere. The intent of this vadose zone treatment is to remove soil contamination, thereby limiting vapor intrusion risks and contaminant migration to the underlying groundwater table.

A series of monitoring wells installed downgradient of the source zones are used to monitor groundwater geochemistry and contaminant distributions. Geochemical data collected from 2008 to the present indicate seasonally reduced groundwater conditions, under the prevailing influence of the regional Tanana aquifer, with predominant groundwater flow towards the west-northwest. High river stage events in the Chena River are known to reverse the groundwater gradient in locations adjacent to the river (Hinzman et al. 2000). This phenomenon has been observed at the Wendell Avenue site at least twice per year (i.e., spring and late summer). The seasonal recharge of Chena River surface water into the Tanana River aquifer potentially creates periods of aerobic conditions in groundwater (i.e., dissolved oxygen >2 mg/L). Contaminant data from groundwater indicates PCE undergoes microbial reductive dechlorination to TCE and cis-DCE/trans-DCE. These contaminants are generally found in greater concentrations than PCE downgradient along the groundwater plume towards the Chena River. VC has not been detected in groundwater. Accordingly, molecular biological information (qPCR with 16s rDNA) indicates the presence of indigenous anaerobic chlororespiring microorganisms in groundwater (i.e., *Dehalococcoides* spp.). However, reverse transcriptase polymerase chain reaction analyses (RT-qPCR) have indicated a low transcriptional expression (and in some wells an absence) of the reductive dehalogenase gene *vcrA*, encoding for the enzyme vinyl chloride reductase. Thus, groundwater contaminants as partially degraded daughter products of PCE biodegradation may discharge into the Chena River. Pore water was collected from the banks of the Chena River to investigate contaminant migration into the river and the viability of the sediment and surface water contact and ingestion exposure pathways.

Methods

Drive point sampling (as described for PushPoint samplers; Lewis 2007) was used to collect pore water during October 2008 and May 2010 from the shore of Kotzebue Sound near the vicinity of the large historic fuel spill (across from the elementary school), along Shore Avenue (location of the seawall project), and from airport property (along Kotzebue Sound and Kotzebue Lagoon). Drive points (AMS™ screened disposable stainless steel Anchor Point tips fitted to 3/16-inch Teflon tubing; AMS, Inc., American Falls, ID, http://www.ams-samplers.com/) were hand-driven using a hollow tube implant drive and fence post hammer. Drive points were developed as mini-groundwater wells, using low-flow groundwater sampling methods (Puls and Barcelona 1996). At Wendell Avenue in Fairbanks, samplers made of a stainless steel well screen enclosed in Colorado sandpack and housed within a slotted 6-inch section of PVC pipe were used to collect pore water. Samplers were buried below the water line in hand-dug holes on the banks of the Chena River and allowed to equilibrate in situ overnight prior to being purged and sampled following low-flow sampling procedures referenced above.

Pore water samples from Kotzebue were analyzed for diesel-range organic petroleum hydrocarbons (DRO) by Alaska Method AK 102 and for benzene, toluene, ethylbenzene, and xylene (BTEX) compounds and polynuclear aromatic hydrocarbons (PAHs) by U.S. Environmental Protection Agency (U.S. EPA) Method 8260C. Total aromatic hydrocarbons (TAH) and total aqueous hydrocarbons (TAqH) were calculated following Alaska Water Quality Standards (Alaska Department of Environmental Conservation 2009). Pore water samples from Wendell Avenue were analyzed for PCE, TCE, cis-DCE, trans-DCE, and VC by U.S. EPA Method 8260B. All analytical results were compared to ADEC groundwater cleanup levels (Alaska Department of Environmental Conservation 2008 b), Alaska Water Quality Standards (Alaska Department of Environmental Conservation 2009), and Alaska Water Quality Criteria for Toxic and Other Deleterious Organic and Inorganic Substances (Alaska Department of Environmental Conservation 2008 a).

Results

In Kotzebue, DRO and benzene were detected in pore water samples above ADEC groundwater cleanup levels in multiple locations along Shore Avenue and at the airport along both Kotzebue Sound and Kotzebue Lagoon (Figures 1, 2; Table 1).

Figure 1. DRO in pore water sampling locations along Shore Avenue in Kotzebue; circles (●) indicate DRO <1.5 mg/L or non-detect; triangles (▲) indicate DRO ≥1.5 mg/L. Bold black lines indicate the historic underground fuel transfer pipeline.

Figure 2. DRO in pore water sampling locations at the Ralph Wien Memorial Airport in Kotzebue; circles (●) indicate DRO <1.5 mg/L or non-detect; triangles (▲) indicate DRO ≥1.5 mg/L.

Many pore water sampling locations containing contaminants above cleanup levels were adjacent to known contaminated sites, such as the historic fuel release site(s) near the elementary school (Figure 1). However, some locations (e.g., Kotzebue Sound at the airport) did not have a distinct source area identified and may indicate comingled groundwater contaminant plumes from many sources. Accordingly, TAH and TAqH concentrations in pore water exceeded Alaska Water Quality Standards for surface waters in similar locations as those exceeding ADEC groundwater cleanup levels (Table 1). TAH and TAqH standards were also exceeded in additional locations, given the additive nature of the indices (e.g., TAH is the sum of concentrations of individual BTEX compounds). However, neither DRO nor benzene was detected in a sea water sample collected adjacent to pore water sampling locations (Table 1). Nonetheless, pore water data confirmed contamination in the GSI and indicate the potential for exposure of human and ecological receptors to contaminated sediments and possibly surface water.

Table 1. DRO, benzene, TAH and TAqH in pore water samples collected in Kotzebue. Samples exceeding groundwater cleanup levels or water quality standards are in **bold**. (N/A, not analyzed; data compiled from Shannon and Wilson 2009, 2010, in press)

Sample location	DRO (mg/L)	Benzene (µg/L)	TAH (µg/L)	TAqH (µg/L)
	ADEC cleanup level			
	1.5	*5.0*	*10*	*15*
Shore	<0.71	**5.45**	**22.70**	**24.21**
Shore	**13.1**	1.43	**94.99**	**120.3**
Shore	**18.9**	3.94	**98.19**	**140.75**
Shore	**24.2**	**10.1**	**108.57**	**211.84**
Shore	**19.8**	**22.1**	**75.16**	**113.05**
Shore	**13.1**	**7.81**	**31.65**	**101.95**
Shore	**19.5**	1.55	**98.97**	**126**
Shore	1.37	0.48	9.49	**21.33**
Shore	**1.76**	<0.40	2.81	**151.56**
Seawater	<0.80	<0.40	0	0
Airport	**5.94**	4.33	4.33	N/A
Airport	1.09	**9.98**	9.98	9.98
Airport	**1.94**	**132**	**136.33**	**136.33**
Airport	**2.33**	**41.6**	**41.6**	**41.6**
Airport	**2.05**	**21.5**	**21.5**	**22.42**
Airport	0.66	**9.36**	**11.66**	11.66

At Wendell Avenue in Fairbanks, pore water data confirmed groundwater discharge of chlorinated solvents to the Chena River (Figure 3). To date, only TCE (at 5.7 µg/L) has exceeded the ADEC groundwater cleanup level (5.0 µg/L), in one of fifteen pore water samples; cis-DCE and trans-DCE have also been detected, but below cleanup levels. VC has not been detected in pore water. TCE in pore water also exceeded Alaska Water Quality Criteria for Toxics and Other Deleterious Organic and Inorganic Substances, based on drinking water standards (5.0 µg/L; Alaska Department of Environmental Conservation 2008 a).

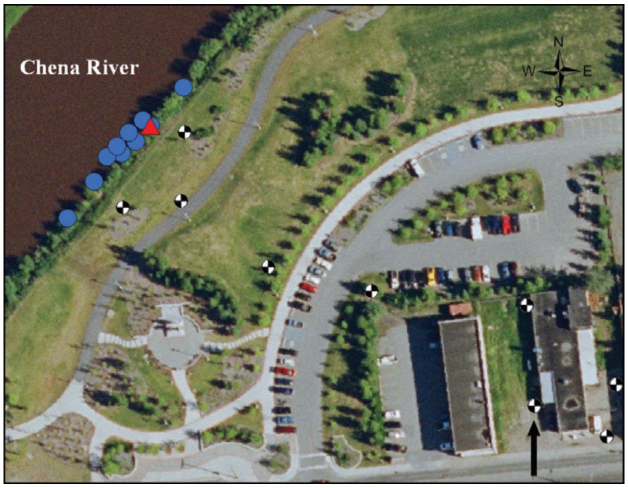

Figure 3. Locations of monitoring wells ( ) at Wendell Avenue, Fairbanks, and pore water samples (●) collected October 2010. TCE in pore water-3 (▲) was 5.7 µg/L. Arrow (↑) points to monitoring well in source area at former dry cleaner (building east).

Pore water results are consistent with groundwater investigations that suggest some level of microbial reductive dechlorination of PCE occurs in groundwater downgradient from contaminant source areas at Wendell Avenue, resulting in TCE that exceeds ADEC groundwater cleanup levels. As in Kotzebue, pore water data indicates contaminants are reaching surface water, completing dermal contact and ingestion exposure pathways for surface water and potentially for sediments as well. Ecological receptors (e.g., aquatic life) are also likely at risk for contaminant exposure. For example, areas of groundwater discharge through river bottom sediments (upwelling locations) are often preferred fish spawning habitats and are common in alluvial rivers of Alaska (Durst 2000). While there is currently no Alaska aquatic life criterion for TCE, discharge of groundwater containing TCE or other chlorinated contaminants may render aquatic organisms as ecological receptors to contaminant exposure.

Conclusions

Management of contaminated sites by ADEC includes an evaluation of risks to both human health and ecological receptors, usually depicted as a conceptual site model (CSM). A CSM describes how people and biota may come into contact with contaminants at a location. Specifically, a CSM identifies exposure pathways, both current and future ways humans and biota may be exposed to contaminants; migration routes, or mechanisms of contaminant transport (through soil, groundwater, biota, etc.); and potential receptors, be they humans, plants, or other animals (Alaska Department of Environmental Conservation 2010). In contrast to a risk assessment, a CSM does not quantify risk but rather identifies real or potential opportunities for contaminant exposure and aides managers in mitigating or minimizing exposure across completed (or viable) pathways. A CSM is required by Alaska statutes for all contaminated sites overseen by ADEC (Alaska Department of Environmental Conservation 2008 b).

Pore water data were used to develop a more complete CSM for area-wide contaminated locations in Kotzebue and for a contaminated former dry cleaner in Fairbanks. Results from pore water sampling demonstrate open exposure pathways where contaminants are reaching surface waters, thus creating exposure pathways to both human and ecological receptors.

Progress towards remedial action is being made at each site. In Kotzebue, responsible parties are being engaged to address further site characterization and cleanup requirements, and data is being collected from monitoring wells at the airport to understand groundwater dynamics and contaminant migration. At Wendell Avenue, a remedial system is operating to address soil contamination and vapor intrusion mitigation. Future remedial action may also include groundwater treatment (e.g., substrate addition) to encourage further reductive dechlorination or cometabolic/direct oxidation of chlorinated contaminants.

Additional monitoring and remedial efforts at both sites may also include continued use of pore water to monitor discharge of contaminated groundwater to surface waters. Future pore water data may come from sentinel monitoring wells located near surface water bodies (e.g., Kotzebue Sound) to capture hyporheic exchange within the GSI. Such monitoring can be used to evaluate whether or not exposure pathways remain viable and serve as points of compliance for cleanup actions. Pore water data, although not exclusively surface water, may also be useful for classifying surface waters as impaired water bodies.

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Application of Updated Approaches for the Reclamation of Placer-Mined Lands in the Harrison Creek Watershed near Central, Alaska

Hans R. Arnett, Patrick L. McMahon, P.E.

Abstract

In 2004, the Bureau of Land Management (BLM) embarked upon a project to reclaim portions of approximately 11 miles of abandoned placer-mined land in the Harrison Creek valley near Central, Alaska. Placer-mined land reclamation efforts performed up to that time generally had been accepted to be inadequate in restoring disturbed valley ecosystems. Historically, reclamation efforts have focused on attempts to design stable stream channels.

During the first Phase of the Harrison Creek Reclamation Project, the BLM contracted with USKH Inc. to develop updated placer-mined land reclamation approaches with applications towards the reclamation of Harrison Creek. Work under this phase included performing literature reviews and field investigations and conducting interviews with parties experienced in placer-mined valley reclamation. The information gathered under these tasks was used to develop a reclamation study that (a) summarizes major issues involved in placer-mined stream valley reclamation; (b) provides an assessment and review of past and current reclamation practices, approaches, and techniques; (c) presents recommendations regarding the field assessment of stream channel and valley stability and the recovery potential in previously reclaimed valleys; and (d) outlines the resulting recommended updated reclamation approaches with particular respect to the reclamation of Harrison Creek.

Updated reclamation approaches were developed to meet the following goal: Perform reclamation activities that promote and accelerate the steady long-term recovery of instream, riparian, and terrestrial ecosystems in a placer-mined valley. As outlined in the study, this goal can be accomplished using the following five general reclamation approaches:

1. Remove sources of excessive sediment introduction into the stream,
2. Construct floodplains,
3. Promote natural revegetation processes,
4. Supply adjustable channels that will trend toward dynamic equilibrium, and
5. Remove mining-related structures, infrastructure, and waste.

The updated reclamation approaches were applied to the development of design plans for the Phase I reclamation of a portion of the Harrison Creek valley, with construction in 2006. Lessons learned from the 2006 reclamation design and construction were applied to the Phase II and III designs that were constructed in 2009. Further design and construction refinements were made to the Phase IV reclamation design, based on lessons learned from the Phase II and III design and construction efforts. The construction of the Phase IV reclamation design was completed in autumn 2010.

Keywords: placer mining, stream reclamation, updated approaches, Bureau of Land Management, Harrison Creek

Introduction

In 2004, the Bureau of Land Management (BLM) embarked upon a project to reclaim portions of approximately 11 miles of abandoned placer-mined land in the Harrison Creek valley near Central, Alaska. Placer-mined land reclamation efforts performed up to that time generally had been accepted to be inadequate in restoring disturbed valley ecosystems. This paper provides a brief overview of placer mining and its

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effects, past reclamation approaches, the development of updated reclamation approaches, and the application of those new approaches to the reclamation of approximately 2.7 miles of the Harrison Creek valley.

Background

The following sections provide brief discussions on the placer mining process and its effects, as well as on previously used reclamation approaches. For a thorough discussion of these topics the reader is referred to Arnett (2005).

The Placer Mining Process

Placer mining consists of the removal of layers of overlying non-gold-bearing materials in order to access and process gold-bearing black sands that have been concentrated near the underlying bedrock surface. Non-gold-bearing materials include vegetation, soil, and alluvial gravels, known collectively as overburden.

In earlier periods of mining, no provisions were made for collecting or controlling sediments washed from overburden or gold-bearing gravels, and the sediments were allowed to wash downstream. Hydraulic mining methods, although relatively uncommon today, were once widely used as a means to strip off overburden materials. High-pressure jets from water cannons (also known as monitors or hydraulic giants) washed overlying gravel deposits away, exposing the gold-bearing materials. The water cannons would also be used to move gold-bearing gravels to the sluice box, wash the gravels through the sluice box, and then remove the tailings (non-gold-bearing gravels) from the site.

The relatively recent promulgation of government regulations regarding water quality standards and reclamation requirements has changed the way that placer mining is conducted. Current regulations require the stockpiling of stripped vegetation, soil, and overburden and the controlling of sedimentation by routing process water through settling ponds that remove finer particles through the settling of suspended sediments before the water is discharged to the adjacent stream.

In modern placer mining, mechanical means are used almost exclusively to prepare and work the mine site. Heavy earth-moving equipment such as bulldozers, front-end loaders, scrapers, or excavators are used to remove and stockpile organic material, soil, and overburden and to construct infrastructure such as access roads and settling ponds.

Placer mining has taken place periodically within the Harrison Creek drainage since 1895. Large stretches of the drainage were abandoned after the completion of mining and left in an unreclaimed condition.

Effects of Placer Mining

Environmental effects of past placer mining can be summarized in the following general categories: water quality reduction, stream channel and floodplain instabilities, vegetation loss and slow revegetation rates, habitat loss, topographic disturbance and aesthetic degradation effects, groundwater changes, loss of permafrost, and enhanced aufeis formation. The persistent reduction of downstream water quality is the most commonly noted environmental effect associated with placer-mined lands, the two most important components of which are increased levels of turbidity and high sediment concentrations.

Unstable stream channels are a common characteristic of both abandoned and reclaimed placer-mined lands. Many factors contribute to the instability of stream channels in placer-mined watersheds. These factors fall under the following categories: changes in the longitudinal profile of the valley, lack of adequately sized floodplains, changes in sediment textures and composition, changes in the amount and type of vegetation present, changes in stream hydrology and sediment transport characteristics, and the extension of instabilities upstream and downstream of mined areas.

Rates of natural revegetation on sites disturbed by placer mining are extremely slow. Some placer mine spoils have little or no vegetative cover 50 or more years after the last episode of mining. The two most commonly noted limiting factors are the loss of moisture-retaining fines and the lack of water for plant growth.

The characteristic instabilities and degraded water quality of stream channels in placer-mined valleys contribute to the loss and degradation of instream habitat for fish and invertebrates. The loss of floodplains and floodplain vegetation reduces terrestrial habitat within riparian corridors.

Abandoned placer mine sites are also characterized by the presence of large, randomly placed, coarse-grained, and sparsely vegetated spoils piles. These spoils can make it difficult or impossible to restore the pre-mining topography of a valley and can also result in an unnatural and barren-looking landscape.

Fine-grained sediments deposited on stream beds in placer-mined drainages reduce the hydraulic contact between the surface water and groundwater of the stream, which can result in higher specific conductance and significantly lower dissolved oxygen concentrations in the groundwater of mined streams. The higher specific conductance and lowered dissolved oxygen concentrations can have negative effects on fish eggs in spawning gravels, and the reduction of hydraulic contact can leave the stream perched above the groundwater table.

In many Interior Alaska drainages with climax vegetation communities, underlying permafrost may help stabilize smaller stream channels by limiting stream bank erosion. This stabilizing factor is lost when vegetation is stripped from the surface and the underlying permafrost melts. Placer mining can increase the formation of aufeis (icings) in drainages. Deposits of aufeis can completely fill channels and in some cases can completely cover valley bottoms. Spring breakup floodwaters perched on top of thick, widespread aufeis deposits can cause erosion of areas well away from the active stream channel. Aufeis deposits can persist well into the summer, which can significantly reduce the growing season on underlying surfaces.

Previously Used Reclamation Approaches and Goals

Government regulations requiring placer miners to reclaim mine sites have been in place for less than 35 years. Because it is a relatively new subject, well-established approaches for the successful reclamation of placer-mined lands do not exist. There is a general consensus among experts in placer-mined land reclamation in Alaska that the reclamation approaches implemented to date have been generally inadequate, whether performed by placer miners or with sponsorship from government agencies.

Any discussion of the success or failure of placer-mined land reclamation efforts must address the specific goals of reclamation. Most stated reclamation goals include some variation on the desire to return natural processes and habitats to the disturbed site within some undetermined length of time and to minimize sediment discharge and protect against riparian erosion. Associated with these general goals of reclamation is the frequently stated assumption that successful reclamation of a site can be achieved through the construction of a stable channel. The idea is that if a stable channel is provided, then normal alluvial, floodplain, and revegetation processes will be encouraged, water quality will improve, and riparian and instream habitats will gradually return to a natural state. Most placer mine reclamation approaches developed to date have addressed the construction of a stable channel as a primary goal. As previously noted, however, post-reclamation stream channels are frequently unstable.

A stable channel is normally defined as a channel with a stable dimension, pattern, and profile that maintains channel features over time and neither degrades or aggrades. It is a commonly stated reclamation goal to design and construct a stable channel with a natural stream pattern, cross-sectional geometry, longitudinal profile, and the sediment transport characteristics of the original stream.

The fact that stable channel design approaches do not appear to work has driven efforts to further refine these approaches. This continued focus on channel stability and design, however, has distracted attention from the larger issue of valley and floodplain stability. It is unlikely that any channel can become stable in an unstable valley. Furthermore, the idea that a newly constructed stable channel can be designed and constructed in a recently placer-mined area ignores the critical role that floodplain vegetation plays in the stability of channels and the time scale of vegetation recovery in placer-mined valleys.

Based on the literature review, site visits, and interviews conducted during the course of this study, it has been concluded that failed attempts to design and construct stable channels within recently placer-mined valleys have been based on unrealistic reclamation goals. These attempts have failed largely because of an inability to predict the long-term equilibrium condition toward which a particular stream channel is heading. Given the profound adverse effects of placer mining on valleys, streams in placer-mined valleys will probably trend toward equilibrium conditions that are different from pre-mining conditions. It may be that channels in dynamic equilibrium in reclaimed valleys will be wider, shallower, less sinuous, and steeper in gradient and will display a braided or split channel pattern more commonly than streams in otherwise similar unmined basins.

Updated Approaches for the Reclamation of Placer-Mined Lands

Updated approaches for the reclamation of placer-mined lands have been developed to meet the following goal: Perform reclamation activities that promote and accelerate the steady long-term recovery of instream, riparian, and terrestrial ecosystems in a placer-mined valley. This goal can be met by providing stable valleys and floodplains that are conducive to natural processes of revegetation, as well as adjustable channels that are trending toward a long-term dynamic equilibrium.

Reclamation efforts should be aimed toward producing a revegetated valley that contains an adjustable, dynamically stable channel with gradually improving water quality characteristics, flowing within a vegetated floodplain between stable, revegetated valley walls. This end result will be accomplished by the reconfiguration of the placer-mined valley in order to promote and accelerate natural recovery processes. Because of the time scale of these recovery processes and the inability to understand precisely where these processes are heading, it should be accepted that more than one episode of reclamation construction may be required to adjust valley slopes, floodplains, and channels in order to promote the steady long-term recovery of ecosystems. The desire to perform a one-time reclamation project that leaves in place a stable reconstructed channel is understandable but unrealistic.

Because the reclamation of placer-mined valleys is a relatively recent occurrence, and because natural recovery rates are so slow, the final configuration of a fully recovered, reclaimed valley is unknown. The time scales of natural recovery processes in Interior Alaskan basins are such that parties involved in the reclamation of a particular site are unlikely to see the eventual full recovery of that site in their lifetimes.

Floodplain construction and the promotion of natural revegetation are the two most important components of the approaches developed in this study. As the valley tends toward the development of a stable, well-vegetated floodplain, long-term gradual improvements in channel stability and water quality will occur.

Because the equilibrium form of channels in reclaimed placer-mined valleys is not clearly understood, channels should be designed and constructed or modified to assure adjustability. Adjustability will allow channels to attain a state of dynamic equilibrium and respond to changes and perturbations occurring within the basin such as upstream mining effects. Non-biodegradable erosion control and energy dissipation structures such as riprap revetments or drop structures should be avoided since they will adversely affect long-term adjustability.

It is assumed that as the channel adjusts over time, natural instream habitat will gradually be created and improved. Given the nature of changes to the system induced by placer mining, however, the stream may never reproduce the instream habitat conditions that existed prior to mining. If instream habitat structures are desired, they should not be constructed or placed so that they inhibit the adjustability of the channel. Furthermore, habitat structures should not be placed in a channel until vegetation has established on the adjacent floodplain. The presence of floodplain vegetation will be an indicator that some degree of channel and floodplain stability has been achieved and, therefore, that instream habitat structures will have some chance of success.

Updated approaches have been developed for the eventual reclamation of abandoned placer-mined lands in the Harrison Creek valley; however, they are also anticipated to be useful for the reclamation of either abandoned or recently mined lands throughout much of Alaska. The approaches have been developed so that they can be implemented by placer miners in a cost-effective manner and therefore do not require engineering and design efforts beyond those normally required of a miner for obtaining approval of a reclamation plan.

The steady long-term recovery of instream, riparian, and terrestrial ecosystems within a placer-mined valley can best be accomplished using the following five general approaches:

1. Remove sources of excessive sediment introduction into the stream,
2. Construct floodplains,
3. Promote natural revegetation processes,
4. Supply adjustable channels that will trend toward dynamic equilibrium, and
5. Remove mining-related structures, infrastructure, and waste.

Application of Updated Approaches in the Harrison Creek Watershed

The updated approaches described above were applied to the design of four phases of reclamation construction projects in the Harrison Creek valley. Phase I construction was completed in 2006, Phases II and III were completed in 2009, and Phase IV was completed in 2010. The Phase I design resulted in the reclamation of three valley segments totaling 1.46 miles. Phase II resulted in the reclamation of a 0.40-mile segment of the valley. Phase III comprised two segments of valley reclamation totaling 0.33 miles. Phase IV also comprised two segments of valley reclamation, totaling 0.49 miles. In total, 2.68 miles of the Harrison Creek valley have been reclaimed since 2006.

By designing and constructing the project in phases, it has been possible to apply lessons learned in earlier phases to the design and construction of later phases. The design process for Phase I was carried out using the approaches outlined in the previous section. The grading portion of the design specified a general recontouring of the valley to eliminate channel division and existing steep, sparsely vegetated spoils piles; to provide adequate flood plain width; and to provide surface hydraulic connectivity between the valley walls and the valley floor. The design also included provisions for preserving small areas covered by thick vegetation within the construction limits. The design concept and process leaned heavily on the use of photogrammetric contour data, aerial photographs, and hydraulic modeling of the proposed channel and floodplain. Finally, at the direction of the BLM, the design was developed to allow construction with the use of only large bulldozers for earth moving. Construction of Phase I was completed in 2006.

A site visit was made in the summer of the following year to select areas for future reclamation and to review the finished product with BLM representative Rodd Moretz, who had performed the construction inspection for the project. Observations made during this site visit indicated that the Phase I reclamation was largely a success. Colonization by pioneering vegetation was noted on most recontoured surfaces, and the valley appeared to be moving toward a more natural condition.

There were a number of lessons learned during the 2007 site visit. First, because it had not been possible to inspect all of the proposed reclamation areas prior to commencement of the Phase I design effort, there was misinterpretation of portions of the aerial photography and contour data. Misinterpretation of the preexisting edge of the channel resulted in the floodplain being constructed too low in some locations, causing channel braids to form in the constructed floodplain. The lesson learned is that it is essential that a site visit be performed prior to the completion of design in order to assure proper interpretation of aerial photography and mapping and to assure that design elevations are correct for constructed floodplains. A second lesson learned was that future phases of work need to include the clear establishment of an access road through the site that keeps recreational vehicles off newly created floodplain surfaces.

The third lesson learned from the Phase I reclamation efforts was that attention needs to be paid to how recontoured surfaces tie in to preexisting surfaces, particularly valley walls, so that closed depressions are not formed and avenues are not provided for overbank flows to bypass the reclaimed portion of the valley. Old settling ponds turned out to be too soft to be adequately reclaimed, since heavy construction equipment became mired while trying to push fill into the ponds.

Other lessons learned during the Phase I reclamation efforts were that the preservation of small areas of well-established vegetation within the construction limits unnecessarily complicates the reclamation efforts, and that the spreading of cleared, grubbed, and stockpiled vegetation and topsoil onto recently constructed surfaces speeds the rate of natural revegetation. However, the amending or mixing of this material into the upper portion of the newly created surfaces is not recommended.

Designs were produced for Phases II and III of the Harrison Creek Reclamation project that incorporated the lessons learned during Phase I construction. The design concept for the two phases was established entirely during the course of the 2007 site visit, eliminating any potential for misinterpretation of photographic or contour data. Design floodplain elevations were based on ordinary high water marks surveyed in the field rather than hydraulic modeling. Equipment needs were assessed in the field based on the demonstrated strengths and limitations of construction efforts made using only large bulldozers for Phase I. The access road was specifically included in the design and placed along the valley margin, well away from the new floodplain. The intent of the design was specified more carefully on the plans in order to improve the tie in with the existing topography and valley walls. Areas of existing vegetation within the construction limits were specified to be salvaged and spread onto the finished surface.

Following the completion of Phase II and III construction in 2009, another site visit was made in coordination with BLM representative Rodd Moretz, who had also done the construction inspection on Phases II and III. Like the 2007 site visit, sites were selected for future reclamation, design concepts were developed in the field, and lessons learned were discussed. Because the lessons learned from the Phase I efforts had been applied to the designs of Phases II and III, the elevation of constructed floodplains were properly specified in the designs, and the reclaimed valley sections better addressed the routing of hillside drainage to the stream and the tie-ins of newly constructed surfaces to preexisting topography. An inspection of the Phase I construction sites showed good natural revegetation occurring on all surfaces, particularly floodplains. Application of lessons learned from Phase I also resulted in a more efficient construction process under Phases II and III.

A review of the Phase I, II, and III sites shows that constructed floodplains are quicker to revegetate than constructed slopes, and that the surfaces that revegetate the fastest are those on which salvaged vegetative material has been spread.

Lessons learned from the Phase II and III construction efforts included the need to place additional design emphasis on the placement of post-construction recreational vehicle access and the accommodation of hillside drainage to prevent creation of impassable mud holes on new access roads. Future phases should allow for a wider road corridor to allow recreational vehicle drivers to negotiate around hazards during wet periods.

The issue of swell in mining spoils became an issue for the balancing of cuts and fills in some areas of the Phase II and III construction. The design called for abandoned bypass channels to be filled with mining spoils excavated from floodplain construction and for the flattening of tall spoils piles. The significant swell associated with placer mining spoils (with increase in volume up to 30 percent more than in situ materials) meant that material volumes were greatly reduced once spoils piles were moved. The end result was that in some cases there were not enough spoils available to adequately fill the abandoned bypass channels. This change in volume needs to be taken into account when material is to be taken from one area to fill a low spot in another area.

The lessons from the Phase II and III construction were applied to the Phase IV design, constructed in 2010.

Summary

Previous efforts to reclaim placer mined valleys generally have been inadequate. An updated reclamation approach has been developed, based on extensive literature reviews, field investigations, and interviews. This approach was applied over the course of three design and construction sequences within the Harrison Creek valley, allowing the lessons learned from each effort to refine the reclamation design and construction methods of each subsequent phase. The resulting reclaimed 2.68 miles of Harrison Creek Valley show good signs for full recovery through natural processes.

Acknowledgments

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Developing a Long-Term Aquatic Monitoring Network in a Complex Watershed of the Alaskan Arctic Coastal Plain

M.S. Whitman, C.D. Arp, B. Jones, W. Morris, G. Grosse, F. Urban, R. Kemnitz

Abstract

The Arctic Coastal Plain (ACP) of northern Alaska consists of an extremely low gradient, lake-rich landscape that is characterized by a complex network of aquatic habitats and surface features strongly influenced by permafrost dynamics. Much is unknown about the form, function, and ecological conditions in this unique hydrologic setting. Amplified climate change and landscape responses in the Arctic further complicate the capacity to separate natural variability from land use effects that may occur with petroleum development. A comprehensive, multi-disciplinary review and analysis of recent studies and initial inventory and monitoring in the Fish Creek watershed on the ACP provided guidance to develop a framework for future aquatic monitoring and integrated research. The result is an established network of stream and lake sites for physical, chemical, and biological data collection that is intended to be sustainable over a long-term period and contribute to understanding Arctic aquatic ecology in the context of climate change and assist science-based land management decisions.

Keywords: Arctic coastal plain, fish, lakes, permafrost, streams, watershed monitoring

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Introduction

Watershed responses to climate change in the Arctic are of increasing interest for managing land use and forecasting affects on fish and wildlife. Greater snow and rain precipitation coupled with increased winter sublimation and summer evapotranspiration (Serreze et al. 2003, Richter-Menge et al. 2008) make projecting annual and seasonal water balance uncertain. In areas of continuous permafrost, such as northern Alaska's Arctic Coastal Plain (ACP), the dynamic hydrologic cycle is constrained near or at the surface, causing expansive, complex, and intermittently connected surface water drainage networks. For example, lakes expand and coalesce in some areas and drain in others (Smith et al. 2005, Jones et al. 2009, Marsh et al. 2009). Similarly, the duration and timing of stream-lake connectivity has been shown to be shifting in opposite directions in various Arctic regions (Woo and Guan 2006, Lesack and Marsh 2007). Permafrost warming and thermokarst erosion play a critical role in such drainage network response to climate change with water availability and temperature acting as important feedbacks (McNamara et al. 1999, Lawrence and Slater 2005). Despite the anticipation of continuing hydro-climatological shifts, few watersheds on the ACP in Alaska have long-term comprehensive environmental monitoring, accentuating the need to establish sustainable watershed observatories in this region.

The Fish Creek watershed (4,676 km^2) in the northeast National Petroleum Reserve - Alaska (NPR-A; Figure 1) is representative of the hydrologically complex ACP, characterized by deep continuous permafrost with a shallow active layer, a high density of thermokarst lakes, drained thermokarst lake basins (DTLB), low-order beaded streams along thermally degraded ice wedges, and higher-order alluvial channels that flow into the Beaufort Sea. These physiographic characteristics, oil and gas exploration and planned development, and the presence of established climate stations made the Fish Creek watershed an advantageous location to pursue a multi-disciplinary observation network.

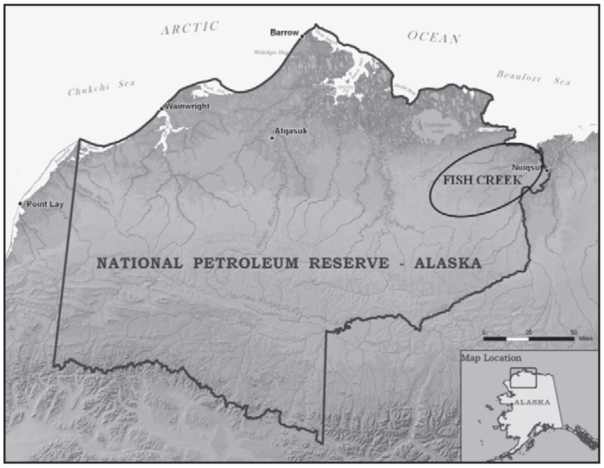

Figure 1. Location of Fish Creek watershed in Alaska.

We utilized a variety of work in the watershed over the past decade to improve understanding of the complex aquatic habitats, establish baseline datasets, evaluate protocols, and identify parameters best suited to the objectives of assessing land use effects and climate change effects on the ACP. Conceptual modeling of Arctic ecological processes related to climate change (Martin et al. 2009) and oil and gas activities (Noel et al. 2008) provided guidance. Initial sites were selected to capture the array of aquatic habitat types in catchments with varying degrees of anticipated development. Ultimately, an assessment of scientific information, land-use projections, operating costs, and anticipated future supporting resources led to the network of sites in place for 2011.

An approach of incremental implementation and refinement was fundamental in developing a valid monitoring network intended to be sustainable over a long time period. Described here are a selection of some of the studies that we used to do this, which included work that focused on drainage network structure, geomorphology, streamflow regimes, water quality, and biological communities.

Methods

A spatial analysis of the Fish Creek watershed drainage network quantified the variability and extent of waterbody types. High-resolution aerial photography was used in conjunction with the U.S. Geological Survey (USGS) National Hydrography Dataset in a geographic information system (ArcMap) to analyze stream and lake extent. Stream channels were categorized as beaded, alluvial, colluvial, or unclassified. Lake basins and drained lake basins were categorized as headwater (outlet only) or flowthrough (inlet and outlet).

The geomorphology of beaded stream systems was investigated to improve understanding of formative processes and variability among streams of this type. This study was accomplished by combining reach-scale topographic and thaw-depth surveys, aerial reconnaissance, and analysis of high-resolution aerial photography. Descriptive characteristics of the five gaged systems were generated based on drainage area; gradient; pool (bead) size, density, shape, and depth; thermokarst conditions (thaw depths); the relative position and area of interconnected lakes; and runoff.

Three main channel (higher-order) alluvial streams have been gaged since 2002 (Fish and Judy Creeks and the Ublutuoch River). Additionally, five beaded tributary streams (Bills, Oil, Crea, Blackfish, and Bear Trio Creeks) have been gaged since 2008. Pressure transducers record water level every fifteen minutes, with discharge measurements made throughout the annual hydrologic cycle.

Water quality studies included continuous monitoring with sondes (Yellow Springs Instruments 6600-V2) as well as broad-scale laboratory constituent analyses. Sondes were deployed in five beaded streams during the open-water season from 2008 to 2010. Sensors included conductivity, pH, dissolved oxygen, turbidity, chlorophyll *a*, and temperature. Water samples were collected (U.S. Geological Survey, variously dated) during the summer of 2010 at five beaded streams and one lake outlet. Samples were analyzed for major ions, trace metal cations, nutrients, organic carbon, and a suite of organic compounds at the USGS National Water Quality Laboratory in Denver, CO.

Because of oil and gas activities, fish studies were conducted in the watershed primarily by industry contractors and Alaska Department of Natural Resources (ADNR). Species inventories conducted from 2000 to 2010 largely consisted of sampling with fyke nets (e.g., MJM Research 2004). In coordination with early inventory efforts, ADNR conducted radio telemetry work on Arctic grayling (*Thymallus arcticus*), broad whitefish (*Coregonus nasus*), and burbot (*Lota lota*) (Morris 2003).

Macroinvertebrate samples were collected from streams in 2006 and 2010. A Petite Ponar dredge was used to collect three subsamples from the streambed at study sites (Burton and Pitt 2001). Each subsample was rinsed through a 500-µm mesh sieve, composited into a single sample, and preserved with 95 percent ethanol. A D-frame kick net (500-µm mesh) was used to collect three subsamples from emergent vegetation (adapted from U.S. Environmental Protection Agency 1997). A sweep sampling technique was utilized with each subsample consisting of approximately three 1-m sweeps. Subsamples were composited for each site and preserved with 95 percent ethanol. Samples were processed at the Bureau of Land Management–Utah State University National Aquatic Monitoring Center.

Phytoplankton was collected for chlorophyll *a* analysis by pumping stream water through a 0.7-µm glass fiber filter and adding $MgCO_3$ to prevent degradation of chlorophyll (Moulton et al. 2002). Different quantities of water (≤1.0, 1.5, and 2.0 L) were filtered during 2004, 2006, 2009, and 2010 to help determine the minimum volume required for acceptable detection limits during laboratory processing. Three samples were collected at each location and frozen until they were processed at Alaska Department of Fish and Game, Division of Habitat in Fairbanks, AK, or Analytica, Inc., in Juneau, AK.

Results

The spatial analysis of the Fish Creek watershed drainage network showed a channel drainage density of 0.48 km/km^2 and lake surface area of 17 percent, with much of the remaining land surface covered by DTLB. Alluvial and beaded channels each account for about 44 percent of the total stream length in the watershed, and lake basins cover 18 percent of the linear drainage distance (Table 1). Most beaded streams initiate from thermokarst lakes (61 percent) or DTLBs (29 percent). Approximately 53 percent of beaded streams terminate in alluvial channels, while others terminate in oxbow lakes, thermokarst lakes, or fluvial transition zones.

Evaluation of beaded stream geomorphology revealed that about half have coalesced or irregular pools indicative of older channels with greater thermokarst degradation, while the others have distinct pools or a series of connected thaw pits that likely denote younger or slowly developing channels. Channels with more coalesced beads had lower gradients and occurred in catchments with lower lake area extent. In the five gaged beaded stream systems, there was considerable variation in baseflow-runoff, which also corresponded to bead coalescence (r=-0.79) and percent lake area (r=+0.90).

Table 1. Drainage network classification for Fish Creek watershed.

Class	Percent class	Subclass	Percent subclass	Length (km)
Stream channels	77.4	--	--	2,345
		Beaded	44.0	1,031
		Colluvial	2.0	47
		Alluvial	44.7	1,048
		Unclassified	9.3	219
Lake basins	18.4	--	--	557
		Headwater	26.9	150
		Flowthrough	73.1	407
Drained lake basins	4.2	--	--	129
		Headwater	60.2	78
		Flowthrough	39.8	51

Most annual runoff in ACP alluvial and beaded streams occurs during spring breakup. Typically, there are few, if any, substantial rainfall peaks during the summer in these low gradient watersheds. Flow generally recedes from snowmelt peakflow through freeze-up, although in some years late summer rain is heavy enough to result in a water level rise (Figure 2). Baseflow conditions are most consistent from mid-July to mid-August, which is the targeted indicator period for collecting annual discrete samples of various types.

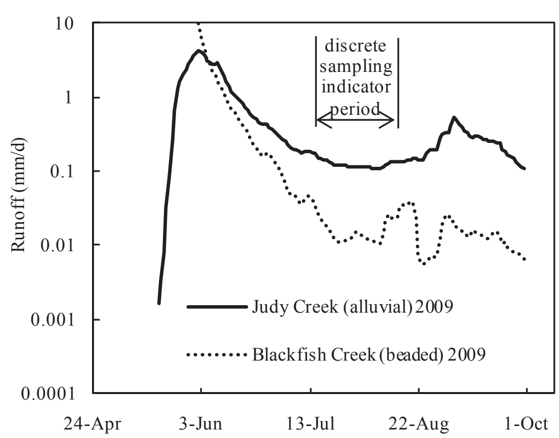

Figure 2. Open-water hydrograph for 2009 of a higher-order alluvial stream and a tributary beaded stream.

Select results from continuous water quality monitoring are presented in Table 2. Various water sample results from the five gaged beaded streams related to productivity include phosphorous (mean = 0.005 mg/L), orthophosphate (mean = 0.008 mg/L), total nitrogen (mean = 0.374 mg/L), and organic carbon (mean = 5.84 mg/L). All organic compound results were less than reportable values, except for a detection of trichloromethane at 0.2 µg/L in the sample and the replicate from Oil Creek.

Table 2. Select results for continuously monitored water quality parameters (June 14 to August 26, 2010).

Site	Temp[1] (°C)	Turb[2] (NTU)	SC[3] (µS/cm)	Chl a[4] (RFU)
	Minimum–Maximum			
Blackfish Creek	3.4–17.7	≤0.1–1.9	68–149	≤0.1–1.90
Crea Creek	2.8–18.6	≤0.1–6.5	76–160	≤0.1–3.60
Bear Trio Creek	2.2–17.8	0.3–15.8	78–177	≤0.1–2.00
Oil Creek	2.0–19.2	≤0.1–7.4	98–261	≤0.1–2.00
Bills Creek	3.0–18.9	≤0.1–3.2	87–201	≤0.1–2.10

[1]Temperature (±0.15), [2]Turbidity (±0.3), [3]Specific conductivity (±1.0), [4]Chlorophyl *a* (relative fluorescence units, ±0.1%)

A total of 16 fish species have been captured in the watershed, with Arctic grayling, broad whitefish, least cisco (*Coregonus sardinella*), and ninespine stickleback (*Pungitius pungitius*) being most prevalent. Dolly Varden (*Salvelinus malma*) and Pacific salmon (chum, pink, chinook, and sockeye) (*Oncorhynchus* spp.) are only found occasionally. Other species include humpback whitefish (*Coregonus pidschian*), Arctic cisco (*Coregonus autumnalis*), round whitefish (*Prosopium cylindraceum*), burbot, lake trout (*Salvelinus namaycush*), Alaska blackfish (*Dallia pectoralis*), and slimy sculpin (*Cottus cognatus*). Notably, species detection at a given location often varied greatly. Telemetry helped explain the variability of fish presence, with Arctic grayling, burbot, and broad whitefish making frequent movements among multiple habitats (Figure 3).

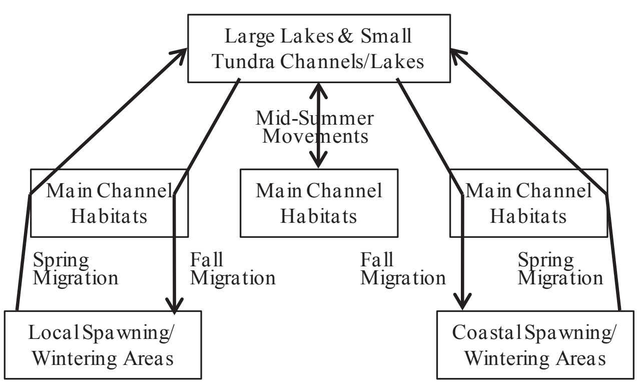

Figure 3. Simplified diagram of typical broad whitefish seasonal movements (adapted from Morris 2003).

Macroinvertebrate community richness was significantly different between emergent vegetation and streambeds at the genus (p<0.001) and family (p<0.001) level (Figure 4), with emergent vegetation being the richest habitat. This habitat also had higher diversity (p<0.01). Chironomidae dominated streambeds, while emergent vegetation communities were dominated by Valvatidae, Chironomidae, and Limnephilidae.

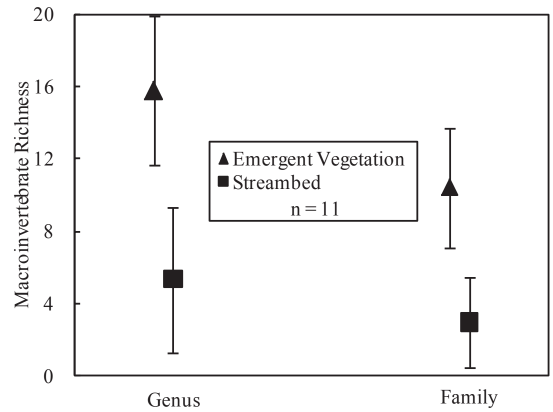

Figure 4. Mean macroinvertebrate richness (±1 standard deviation) for emergent vegetation and streambeds (2006).

Phytoplankton samples consisting of ≤1.0 L of filtered stream water did not contain enough absolute chlorophyll *a* in the extraction solution to satisfy a minimum reporting limit (MRL) based on a detection level of 0.1 mg/m³ (Figure 5). Filtering either 1.5 or 2.0 L largely resulted in valid samples, with only a small proportion being <MRL. However, samples collected by filtering 2.0 L of water were substantially more difficult to collect in the field because of filter clogging. Among the valid samples, the mean summer chlorophyll *a* concentration for Fish Creek watershed streams is 1.58 mg/m³.

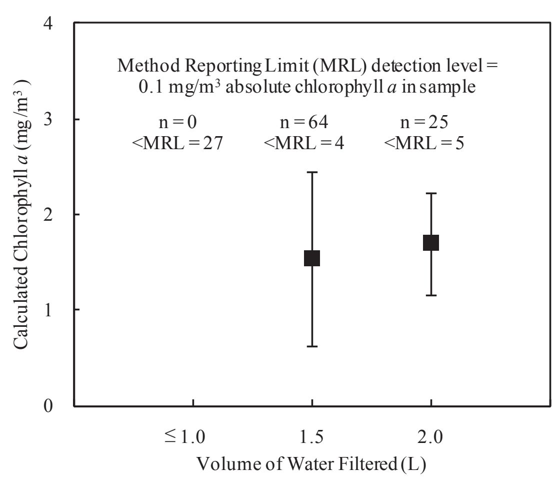

Figure 5. Mean chlorophyll *a* concentrations (±1 standard deviation) for phytoplankton samples at different volumes.

Table 3. Number of fixed monitoring sites where annual data is intended to be collected.

Aquatic habitat	Total sites	WL[1]	D[2]	T[3]	SC[4]	pH	DO[5]	Turb[6]	Chl *a* trend[7]	Chl *a* lab[8]	Nutr[9]	OC[10]	MI[11]	Zoop[12]	Fish	AL[13]
Alluvial stream	3	3	3	3	2	2	2	2	2	3	2	2	2	--	--	--
Beaded stream	5	5	5	5	5	5	5	5	5	5	5	5	5	5	1	5
Headwater lake	4	4	--	4	1	1	1	--	1	4	1	1	4	4	--	--
Flowthrough lake	3	3	--	3	1	1	1	--	1	3	1	1	3	3	--	--

[1]Water level, [2]Discharge, [3]Temperature, [4]Specific conductivity, [5]Dissolved oxygen, [6]Turbidity, [7]Chlorophyll *a* probe, [8]Chlorophyll *a* sample, [9]Nutrients, [10]Organic carbon, [11]Macroinvertebrates, [12]Zooplankton, [13]Active layer depth

Conclusions

The drainage network classification, geomorphology analysis, and streamflow monitoring collectively contributed to insights regarding watershed-scale hydrologic behavior, waterbody connectivity, and formative processes, concepts principle to evaluating potential ecosystem shifts related to climate change or land use effects. The initiation of most beaded streams from thermokarst lakes or DTLBs suggests that lakes are the dominant control on initial channel formation and evolution. The strong correlation of beaded stream baseflow runoff with bead coalescence and percent lake area indicates that proportionally higher lake area results in higher water yield in the short term and is indicative of older or more developed drainage networks in the long term. The classification of waterbodies also guided site refinement to make the network more representative of the hydrological processes occurring in ACP watersheds.

Water quality parameters monitored in situ showed that beaded streams generally have low productivity and low suspended sediment loads throughout the summer season based on relative fluorescence and turbidity values. Higher turbidity values observed at Bear Trio Creek were attributed to unstable banks in the upper catchment that were identified as the origin of the sediment inputs. Among all parameters, upper temperature limits are likely the primary constraining factor for fish suitability in beaded streams. Future water quality monitoring will largely involve tracking parameters by instrumentation, with expanded efforts to evaluate alluvial stream sites. Water sampling will vary based on yearly resources, although a set of priority constituents most relevant to climate change projections was selected for annual monitoring and includes nutrients and organic carbon.

While fish community is utilized in many regions as an indicator of stream health (Simon 1998), it is not a suitable option on the ACP because of high spatial and temporal variability. The best strategy for evaluating ecological relationships involving Arctic fish is focused research within the scope of the monitoring framework. Other biological studies provided baseline data and helped identify sampling best suited to ACP aquatic habitats. For example, long-term macroinvertebrate sampling will focus on emergent vegetation. Richest-habitat targeting is a well-founded option for monitoring (Karr and Chu 1999) and is more cost effective than multi-habitat sampling.

The strategy of the Fish Creek watershed aquatic monitoring program from 2011forward includes maintaining a set of fixed sites (Table 3), establishing an intensive sampling study catchment, and continuing integrated research. The fixed-site network includes streams and lakes characteristic of the primary habitat types in the watershed. Parameters not monitored continuously at those locations will be collected during a baseflow indicator period, which is a sound monitoring program approach (Karr and Chu 1999). More frequent sampling to investigate temporal trends will occur in the intensive sampling catchment, Crea Creek. As integrated research is an essential component of an effective monitoring program (Mulder et al. 1999), this will continue in Crea Creek and, as resources allow, at a greater spatial extent throughout the watershed.

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Monitoring Water Quality in Alaskan National Parks: Development and Application of RIVPACS Empirical Models for the Assessment of Ecological Condition

T. Simmons, J.D. Ostermiller

Abstract

Historically, assessment of water quality focused on chemical measures, and this is still the case for nearly all water quality programs in U.S. national parks. Over the last 20 years, however, the emphasis at the State and Federal level has shifted to biological measures of water quality, with the goal of assessing the ecological integrity of aquatic ecosystems. Because benthic macroinvertebrates are ubiquitous in aquatic environments, occur at high densities, are relatively inexpensive to collect, and respond sensitively to environmental stressors, most biological assessment programs rely on these organisms as indicators of ecosystem condition. The River Invertebrate Prediction and Classification System (RIVPACS) is a robust biological assessment tool that uses natural environmental gradients to predict the species composition of benthic macroinvertebrates that would be expected to occur in the absence of anthropogenic stress. Deviations from the expected species composition then constitute a measure of biological impairment. We used macroinvertebrate and environmental data from 66 streams in Denali National Park and Preserve and Wrangell–St. Elias National Park and Preserve to develop and test a preliminary RIVPACS model for these parks. We will discuss the development of this model and its application as a tool for the contemporary biological assessment of water quality in streams and rivers in Alaskan national parks. We will also discuss the potential of using this approach to detect the ongoing effects of climate change in otherwise pristine streams and rivers.

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Climate and Hydrology

Potential Climate Change Effects on Water Tables and Pyrite Oxidation in Headwater Catchments in Colorado

Richard M.T. Webb, M. Alisa Mast, Andrew H. Manning, David W. Clow, Donald H. Campbell

Abstract

A water, energy, and biogeochemical model (WEBMOD) was constructed to simulate hydrology and pyrite oxidation for the period October 1992 through September 1997. The hydrologic model simulates processes in Loch Vale, a 6.6-km² granitic watershed that drains the east side of the Continental Divide. Parameters describing pyrite oxidation were derived sulfate concentrations measured in pore water and stream water in Handcart Gulch, a naturally acidic watershed in the Colorado Mineral Belt. Average monthly differences in precipitation and temperature between current and future climates, as predicted by using six global circulation models and three carbon-dioxide emission scenarios, were input into WEBMOD to identify possible shifts in the quantity and quality of the water flowing from the watershed for the period 2005 through 2100. Initial results suggest that increased air temperatures will result in earlier snowmelt compared to current conditions. Average sulfate concentrations and acidity in streams draining hydrothermally altered terrain may decrease as water tables rise in response to greater overall precipitation and earlier snowmelt, although a net increase of sulfate load was simulated as a result of greater overall discharge. Evapotranspiration is expected to increase but not enough to offset the increase in precipitation.

Introduction

Concentrations of metals in the water and sediments in Handcart Gulch and other naturally acidic streams in the Colorado Mineral Belt are often found at levels that are toxic to fish and other aquatic organisms (Figure 1; Church et al. 2009, Schmidt et al. 2009). The present study uses numerical simulations to examine how climate change may affect the oxidation of pyrite and its acid-producing potential in these streams. Because hydrologic observations in Handcart Gulch are in the preliminary stages, the geochemistry determined for Handcart Gulch was inserted into a hydrologic model of Loch Vale that was built using temperature, precipitation, and discharge observations for the water years 1993 through 1997.

The Loch Vale watershed drains the eastern slope of the Colorado Front Range in Rocky Mountain National Park near Estes Park, CO. Within the 6.6-km² watershed (Figure 1), elevations rise from 3,000 meters (m) above mean sea level (amsl) at the watershed outlet (the Loch) to 4,000 m amsl along the Continental Divide. The watershed contains a small cirque glacier and a rock glacier that occupy the topographically shaded northeastern facing slopes. In western North America, snowpack has declined in recent decades, and this trend is expected to continue through the 21st century (Pederson et al. 2011). Mean annual air temperatures at Loch Vale have risen 1.2–1.4°C since 1990, resulting in earlier spring melts and the release of water previously locked up in permafrost above 3,460 m amsl (Clow et al. 2003).

Handcart Gulch is a naturally acidic stream that drains hydrothermally altered terrain on the eastern flank of the Continental Divide 90 km south of Loch Vale (Caine et al. 2008). Elevations in the watershed vary from 3,200 m at the gage to 3,700 m amsl. Similar to Loch Vale, a rock glacier flows down the valley. Using cations, including H_3O^+, is an order of magnitude greater at Handcart Gulch than at Loch Vale. Of course, alpine watersheds are not steady state; instead they display large variations of flow and concentration as snowpack builds through the winter and melts in the summer. Numeric watershed models that simulate

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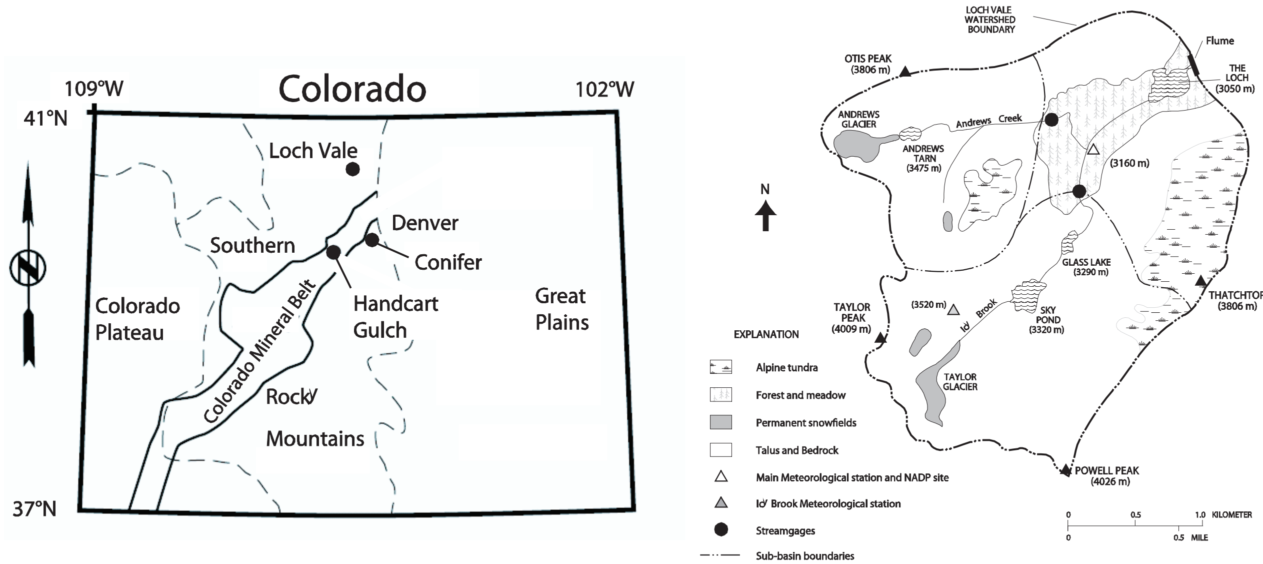

Figure 1. Location of Loch Vale and Handcart Gulch in relation to major geomorphic and mineralogic provinces in Colorado (modified from Church et al. 2009) and descriptive map of the Loch Vale watershed (modified from Campbell et al. 1995).

steady-state estimates described below, the export of hydrologic and geochemical fluxes and states on a daily time step provide an important tool to better understand the cause and effect of these variations.

Methods

Possible changes in the hydrology and pyrite oxidation of mineralized watersheds were simulated by shifting the temperature and precipitation observed for water years 1993 through 1997 to the temperature and precipitation predicted by six general circulation models (GCMs), each running three different carbon dioxide emission scenarios for the 21st century. The construction of a Handcart Gulch hydrologic model is in the preliminary stages, reflecting the limited observations of flow and water quality at the basin outlet. So for this study, Loch Vale was used as a hydrologic proxy for Handcart Gulch, which has similar elevations and air temperatures. Pyrite concentrations and weathering rates estimated for Handcart Gulch were added to the Loch Vale model to simulate annual variations in water quality in alpine and subalpine watersheds underlain by mineralized deposits.

WEBMOD

The Water Energy and Biogeochemical Model (WEBMOD) is a semidistributed watershed model that simulates hydrology and geochemistry on a daily timestep (Figure 2; Webb et al. 2006).

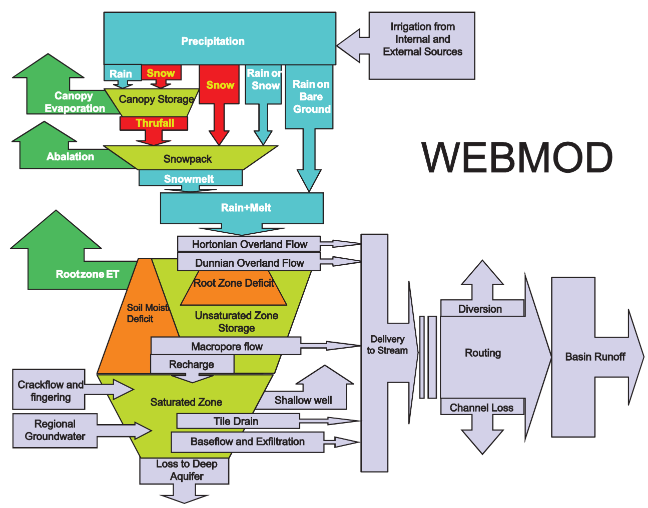

Figure 2. Reservoirs and fluxes of WEBMOD. Infiltration, transpiration, and wetting of the root zone by groundwater are not shown to simplify the schematic.

Insolation, temperature, precipitation, potential evapotranspiration, and canopy interception and evaporation are distributed to hillslopes by using algorithms from the U.S. Geological Survey's Precipitation Runoff Modeling System (PRMS; Leavesley et al. 1983, Leavesley and Stannard 1995). Snowpack accumulation and melt are simulated with the National Weather Service's Hydro-17 snow module (Anderson 1973). Hillslope processes including overland flow, infiltration, evaporation and transpiration, shallow return flow, and base flow are simulated with the variable source area model TOPMODEL (Beven and Kirkby 1979, Wolock 1993, Beven 1997). Hillslope discharge is routed to the basin outlet by using a unit hydrograph (Clark 1945). Mixing ratios from the hydrologic model are used to simulate the transient aqueous geochemistry in each hydrologic compartment using PHREEQC (Parkhurst et al. 1999, Charlton and Parkhurst 2011).

Hydrologic Model

The Loch Vale watershed was discretized into six hillslopes representing southern (warmer) and northern (cooler) exposures for Andrews Creek, Icy Brook, and the Loch area downstream from the confluence. Five years of meteorological observations from a weather station in the watershed were used to drive the simulations of evapotranspiration, snowpack processes, hillslope processes, and runoff from October 1992 through September 1997. Annual precipitation during the period averaged 128 cm with 60 percent falling mostly as snow from late autumn through spring (November through May). WEBMOD process parameters were adjusted to reproduce the dominant hydrologic processes in the basin including the accumulation of snowpack in the winter, the initial rise of soil moisture in the spring, and melting of the snowpack in the spring and summer (Figure 3). The sharp rise in observed runoff beginning in the middle of May marks the breakup of ice in the flume at the outlet of the Loch. There are no ice dam dynamics in WEBMOD, so simulated runoff increases earlier than observed runoff. Runoff decreases as the snowpack becomes depleted, first near the Loch and progressing up Icy Brook and Andrews Creek. The temperature lapse rate between two weather stations in the watershed, the Main weather station (3,160 m amsl) and the Icy Brook weather station (3,520 m amsl), for water year 1994 was approximately 10°C/km in the winter and 6°C/km during the summer.

Simulated snowpack begins accumulating in September and reaches a maximum of approximately 60 cm of snow-water equivalence at the end of April (Figure 4). A basin average snow-water equivalence of 60 cm is equal to 1.7 m of snow depth using an average density of 35 percent. Because of the extreme relief and high winds, the depth of snowpack varies greatly, with basin-wide snow-covered area rarely exceeding 75 percent.

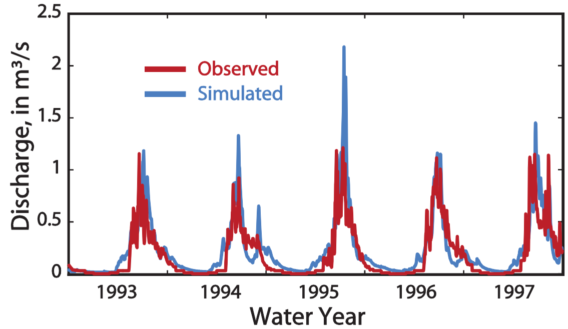

Figure 3. Observed versus simulated discharge for Loch Vale for the water years 1993–1997. The abrupt beginning of runoff observed in the spring corresponds to the melting of the ice dam in the flume.

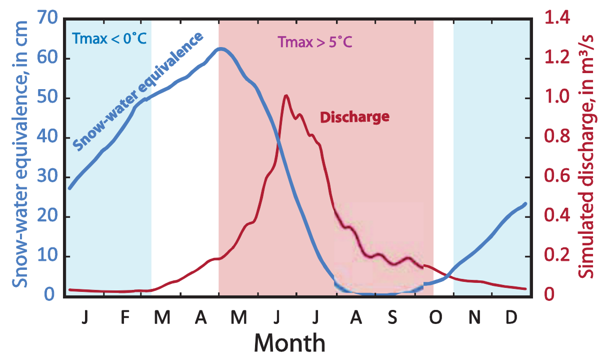

Figure 4. Average daily simulated snow-water equivalence and discharge at Loch Vale for the water years 1993–1997. Values are moving averages centered on 9-day periods. Shading indicates daily maximum temperatures at the Main weather station (blue, <0°C; white, 0°C<tmax<5°C; red, >5°C).

The onset of partial thawing of the watershed begins in March, as documented by rising soil moisture saturation that corresponds to rising spring temperatures and rain on snow events (Figure 5). Soil moisture reaches a maximum during spring melt and then drops in response to evapotranspiration (ET) demand through the summer. Soil moisture begins to rise in late summer, possibly in response to monsoonal moisture and (or) decreased transpiration rates due to the closing of stoma by stressed conifers. Sap-flow sensors installed in lodgepole pines at an elevation of 2,500 m near Conifer, CO, indicated a transpiration season of April 8–October 12 in 2000 and April 18–September 28 in 2001 (Bossong et al. 2003).

In the WEBMOD simulations for the higher and cooler elevations of Loch Vale, transpiration was set to begin at the beginning of June and end at the beginning of August. Because the area covered by tundra, grass, and conifers is limited for Loch Vale, transpiration is a relatively minor fraction of overall ET in the model simulations. The reader should note the water-table fluctuations of less than a meter, shown in Figure 5, are computed for porosities typical of sandy clay loams or silty loams found in valley bottoms (porosity 0.4 with a field capacity of 0.2). Porosities in the WEBMOD implementation of TOPMODEL are constant for a hillslope, but the water-table fluctuations in the fractured rocks near the Continental Divide could be several tens of meters given measured porosities less than 0.05.

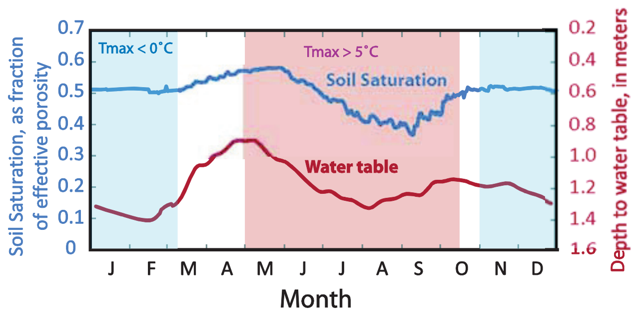

Figure 5. Average daily observed soil saturation for water years 1998–2002 and simulated depth to water table for water years 1993–1997. Values are moving averages centered on 9-day periods. Shading indicates daily maximum temperatures at the Main weather station (blue, <0°C; white, 0°C<tmax<5°C; red, >5°C).

Pyrite Weathering

Sulfate concentrations in precipitation falling on the Rocky Mountains are usually less than 20 µeq/L (approx. 1 mg/L) (National Atmospheric Deposition Program 2007). After interacting with the biotite gneiss and the Silver Plume granite (Cole 1977), slightly higher concentrations are measured in the waters exiting the Loch, implying that there is a certain amount of pyrite in the watershed (Mast 1992). In contrast, if the precipitation falls on hydrothermally altered terrain, sulfate in the springs and streams draining that terrain often exceed 1,000 µeq/L (approx. 50 mg/L). Figure 6 shows the dissolution of pyrite when exposed to oxygen as occurs in a mine tailings pile (Diehl et al. 2008). The dominant hydrothermal alteration assemblage at Handcart Gulch is quartz-sericite-pyrite (QSP) with an average concentration of about 8 weight-percent fine-grained, disseminated pyrite, FeS_2, and quartz-pyrite veinlets (Manning et al. 2009).

The three most sensitive factors in simulating pyrite weathering and the production of acidic waters in mineralized deposits are the mineral area to volume (A/V) ratio, the partial pressure of oxygen in the unsaturated zone, and the total mass of pyrite available for oxidation. Heterogeneities in abundance of pyrite and fracture apertures at both large and small scales make the knowledge of these data at all points in the watershed impossible. The values assigned to these parameters in the model, such as 5.0 dm^2/dm^3 for the A/V ratio, therefore, represent 'effective' values that are calibrated such that simulated values for pH and sulfate concentrations match those observed in the monitoring wells and and stream samples from Handcart Gulch (Manning et al. 2010). In these model runs, the initial mass of available pyrite is assigned an arbitrary value of 1.0 mol/kg, sufficient to ensure that the available pyrite will not be a limiting factor in the production of acidic waters during the simulation period of 5 years. Whereas pyrite abundance is not a limiting factor, the availability of oxygen is. In a thick unsaturated zone where pyrite is present, weathering consumes oxygen faster than diffusion can replace it, resulting in decreased oxygen and reduced weathering rates with depth (Figure 7).

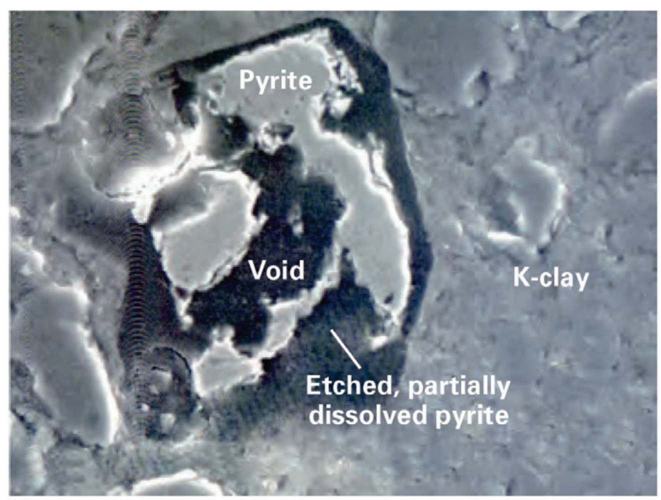

Figure 6. Partial dissolution of pyrite grain from the Dinero Mine waste pile (Diehl et al. 2008)

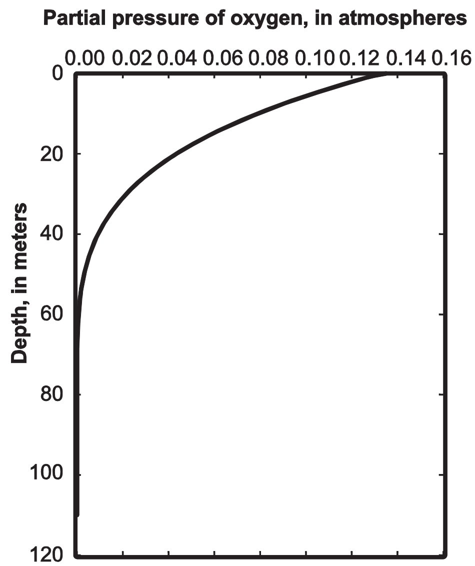

Figure 7. Plot showing decreasing oxygen with depth typical of values measured in deep unsaturated zones in the Handcart Gulch watershed.

To represent this process, pore waters in the unsaturated zone were equilibrated with oxygen partial pressures from 0.08 to 0.003 atmospheres depending on the their thickness.

The simplest form of pyrite oxidation was assumed:

$$FeS_{2(s)} + 3.5\ O_2 + H_2O \rightarrow Fe^{2+} + 2\ SO_4^{2-} + 2\ H^+$$

As a result of the oxidation of pyrite, the pH of the perennial stream draining the Handcart Gulch varies between 3 and 4. Using an average pH of 3 ($[H^+]$ = 1 meq/L), a sulfate concentration of 50 mg/L ($[SO_4^{2-}]$~1 meq/L), and a mean discharge of 46 L/s (unpublished data for water year 2005) at the outlet of a watershed area of 3.6 km², the annual export of hydrogen from Handcart Gulch exceeds 4,000 eq/ha/yr with similar yields of sulfate.

Williamson and Rimstidt (1994) added extensive laboratory observations to the existing body of literature to arrive at the following empirical empirical kinetic rate expression for the above reaction (appropriate for the pH range 2–10):

$$r'_{25°C} = 10^{-8.19(\pm 0.10)} * \frac{m_{DO}^{0.5(\pm 0.04)}}{m_{H^+}^{0.11(\pm 0.01)}},$$

where $r'_{25°C}$ is the rate of pyrite weathering in mol/m^2/s, and concentrations, m, of dissolved oxygen and hydrogen (as a hydronium ion H_3O^+) are in mol/kg of water at 25°C. The rate exponent of -8.19 describes the rate at 25°C for 1 m² of pyrite surface area per kg of solution. The pyrite kinetic weathering rate used in the WEBMOD simulations was

$$r'_{2°C} = 10^{-9.14} * \frac{5.0 m_{DO}^{0.5}}{m_{H^+}^{0.11}}.$$

The exponent of -9.14 was computed by using the Arrhenius equation (Arrhenius 1889) with an activation energy for pyrite of 65 kJ/mol to convert the rate constant at 25°C to 2°C, an average soil temperature at Handcart Gulch. The coefficient of 5.0 is the effective surface area to volume (A/V) ratio, in PHREEQC units of dm²/dm^3, estimated for the QSP deposits at Handcart; A/V equals 100.0 dm²/dm^3 (≅ 1 m²/kg) in the Williamson and Rimstidt paper.

Using transition-state theory (Eyring 1935), the computed kinetic reaction rate is further reduced as pyrite approaches saturation:

$$r = r'_{2°C} * (1 - SR^n),$$

where r is the rate of pyrite weathering in mol/m^2/s, $r'_{2°C}$ is the empirical rate constant in mol/m^2/s at 2°C, SR is the saturation ratio that ranges from 0.0 for no dissolved ions to 1.0 when the solution is in equilibrium with pyrite, and n is a coefficient to account for other factors, set to 1 in this case.

Coupled Hydrology and Geochemistry

The unsaturated and saturated zones were initialized with major-ion concentrations similar to those measured in samples from monitoring wells in the Handcart Gulch watershed. Both zones were assigned identical pyrite abundance (1 mol/kg), A/V ratios (5.0 dm²/dm^3), and kinetic expressions. Unsaturated zones were in equilibrium with depleted atmospheric oxygen as described above. Oxygen in the saturated zone, however, was limited to that contained in recharge water draining from the unsaturated zone; diffusion of oxygen into the saturated zone was considered negligible. After initializing the water quality and geochemistry, the daily simulations begin with the atmospheric deposition using concentrations measured at Loch Vale, one of the closest sites where precipitation chemistry is routinely measured (National Atmospheric Deposition Program 2007). Simulated fluxes between compartments shown in Figure 2 were used to provide mixing ratios for a series of forward-feeding batch reactions: Precipitation > Canopy > Snowpack > Overland Flow > Unsaturated Zone > Saturated Zone > Stream. On days with no precipitation, transpiration would move water and solutes to the canopy. For all compartments the batch reactions proceed as follows: inputs are conservatively mixed with existing storage, the mixed solution is exported to downstream compartments, and reactions then take place in the final volume. During the reaction calculations, ions released by the oxidation of pyrite can precipitate out as goethite, kaolinite, montmorillinite, and (or) calcite when thermodynamically stable.

Net exports of sulfate will closely match the shape of the hydrograph (Figure 4) as the loads are dependent on discharge, which varies over orders of magnitude, and concentration, which is predicted to vary less. Simulated sulfate concentrations in the unsaturated zone varied between 20 and 60 mg/L with the greatest concentrations in late winter and late summer, coincident with lower water tables (Figure 8, Figure 5). Groundwater concentrations varied less, between 41 and 44 mg/L, and are predicted to lag the peak concentrations in the unsaturated zone by more than a month. The acidity of the streamwater in WEBMOD is a mixture of overland flow, the unsaturated zone, and groundwater (Figure 2); the simulated stream acidity varies throughout the year, reflecting relative contributions from each source (Figure 9).

Climate Models and Energy Scenarios

Greenhouse gas emissions continue to increase as fossil fuels are consumed by a global population that has grown from less than 2 billion in 1900 to almost 7 billion today. Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level (Intergovernmental Panel on Climate Change 2007). The main uncertainty is the degree of warming and the direction and magnitude of changes in precipitation. To address these uncertainties, six global climate models were run with three carbon dioxide emission scenarios using the climatologic downscaling and data synthesis of Hay et al. (2011) and Markstrom et al. (in press). Global climate models consist of coupled atmospheric and oceanic general circulation models sometimes linked with additional models that simulate aerosols, sea ice, glacial melting, land cover, and vegetation models. The acronym GCM that was originally used for general circulation models is now commonly used, as it is here, to describe the coupled global climate models. All models in this study (Table 1) are sensitive to the loads of greenhouse gases emitted by natural and anthropogenic sources.

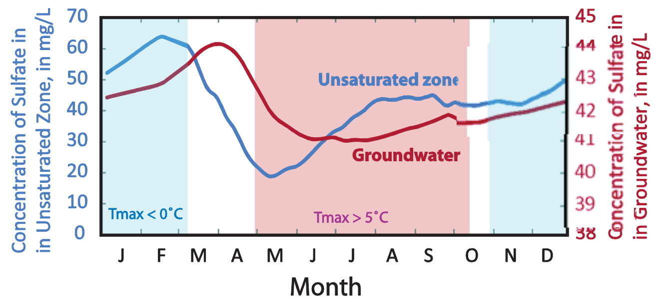

Figure 8. Average daily simulated concentration of sulfate in the unsaturated zone and groundwater for the the water years 1993–1997 for the Loch Vale hydrologic model with a hypothetical pyrite-bearing deposit. Values are moving averages centered on 9-day periods. Note that variations in unsaturated zone concentrations are ten times those simulated for groundwater. Shading indicates daily maximum temperatures at the Main weather station (blue, <0°C; white, 0°C<tmax<5°C; red, >5°C).

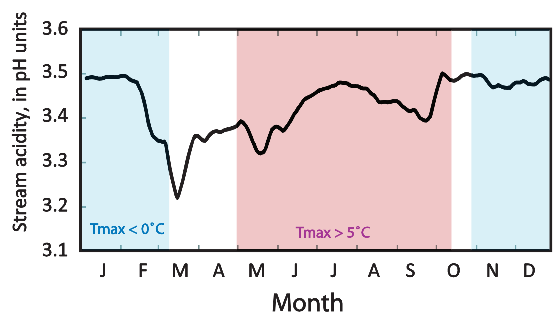

Figure 9. Average daily simulated stream acidity for the water years 1993–1997 for the Loch Vale hydrologic model with a hypothetical pyrite-bearing deposit. Values are moving averages centered on 9-day periods. Shading indicates daily maximum temperatures at the Main weather station (blue, <0°C; white, 0°C<tmax<5°C; red, >5°C).

Each GCM was validated by simulating the climate for the 20th century. In all cases, matching temperatures observed in the second half of the century required the input of anthropogenic greenhouse gases. The 20th century simulation, referred to as 20C3M for each GCM, forms the baseline to compute temperature and precipitation changes for each emission scenario of the 21st century. The WEBMOD simulation for water years 1993–1997 forms the hydrologic and geochemical baseline to quantify potential shifts in hydrology and oxidation of pyrite during warming climate conditions through the 21st century.

Table 1. GCM outputs used in this study from the World Climate Research CMIP3 multimodel dataset archive. (BCCR-BCM2.0, Bjerknes Centre for Climate Research Bergen Climate Model; CSIRO Mk3.0, Commonwealth Scientific and Industrial Research Organisation Mark version 3.0; INM-CM3.0, Institute of Numerical Mathematics Coupled Model, version 3.0; MIROC3.2(medres), Model for Interdisciplinary Research on Climate 3.2, medium-resolution version)

GCM*	Developer
BCC-BCM2.0	Bjerknes Centre for Climate Research, Norway
CCSM3	National Center for Atmospheric Research, USA
CSIRO-Mk3.0	Commonwealth Scientific and Industrial Research Organisation, Australia
CSIRO-Mk3.5	Commonwealth Scientific and Industrial Research Organisation, Australia
INM-CM3.0	Institute for Numerical Mathematics, Russia
MIROC3.2(medres)	National Institute for Environmental Studies, Japan

* CMIP3 GCM documentation, references, and links can be found online at http://www-pcmdi.llnl.gov/ipcc/model_documentation/ipcc_model_documentation.php.

Table 2. Three emission scenarios simulated by each of the six GCMs to simulate climate conditions for the 21st century (from Intergovernmental Panel on Climate Change 2007).

Scenario*	Description
20C3M	20th century climate used to determine baseline (1992–1997) conditions
A2	Very heterogeneous world with high population growth, slow economic development, and slow technological change
A1B	Very rapid economic growth, a global population that peaks in mid 21st century and rapid introduction of new and more efficient technologies with a balanced emphasis on all energy sources
B1	Convergent world, with the same global population as Emission Scenario A1B but with more rapid changes in economic structures toward a service and information economy that is more ecologically friendly

The three greenhouse gas emission scenarios used in this study (Table 2) range from pessimistic to optimistic in terms of slowing the rate of increase of annual emissions though a combination of slowing population growth, conservation practices, and new technology.

For each GCM and emission scenario, the average monthly difference in precipitation and temperature were computed between the baseline (20C3M for 1992–1997) and the 95 21st-century 5-year periods (2001–2005, 2002–2006, ..., 2095–2099). Those monthly differences were applied to the WEBMOD data to create 95 ‘future’ 5-year simulations of hydrology and pyrite weathering for the six GCMs and three emission scenarios (1,710 model runs). Differences in monthly means of simulated hydrology and pyrite weathering were computed for the last 4 years of each model run; simulations of the first year were excluded to diminish artifacts resulting from initial conditions. To illustrate seasonal differences in watershed processes, baseline model runs for each GCM/scenario were compared to the monthly statistics for model runs centered on the years 2030, 2060, and 2090. Possible interannual variations and long-term trends were documented by plotting the means and range for each variable.

Results

All GCM/emission scenarios predict that average air temperature in Colorado will rise throughout the century; the mean increase for scenario B1 is approximately 2°C, whereas scenario A2 predicts 4°C (Figure 10).

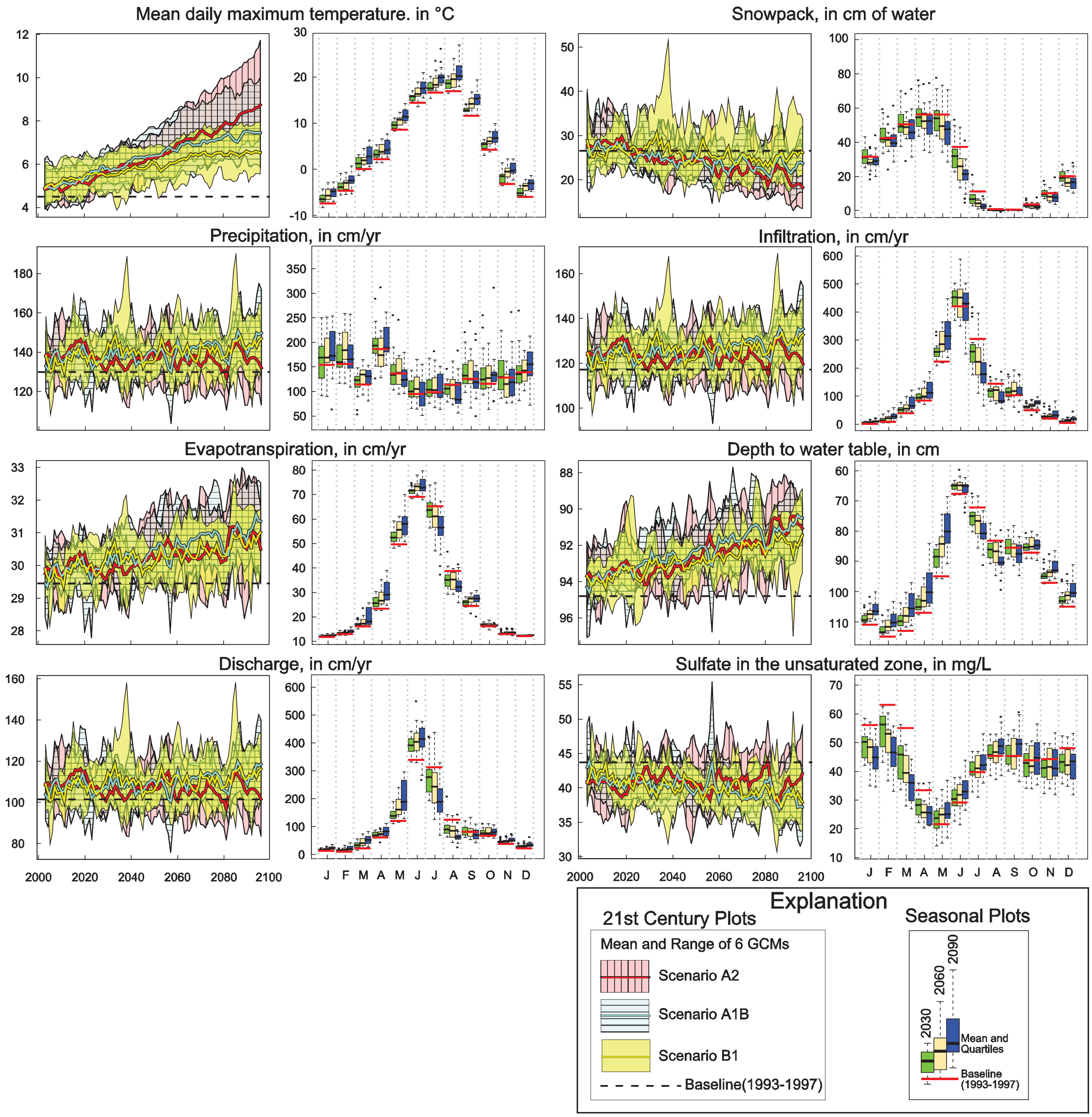

Figure 10. Interannual and seasonal differences between simulations of hydrology and sulfate production for climate observed for water years 1993–1997 and climate predicted by six global climate models using three different emission scenario.

The Clausius-Clapeyron equation (Clapeyron, 1834) dictates that a 2°C increase in average air temperature would result in approximately a 14 percent increase in water-holding capacity in the atmosphere (approx. 7%/K). The absolute increases in moisture amount should be greater at lower latitudes, but the additional moisture may not be available to extratropical storms creating precipitation over the Rocky Mountains (Trenberth et al. 2003). The GCMs simulating the A1B and the B1 scenarios predict an increase in mean annual precipitation of about 20 cm over the century (an increase of 15 percent over the mean baseline precipitation of 130 cm). The interannual trend of increasing precipitation is rather noisy, reflecting the complex interplay between albedo and the evaporation, advection, and precipitation of moisture around the globe. A plot of seasonal precipitation for all GCMs and scenarios for the years 2030, 2060, and 2090 shows most of the increase in precipitation occurring in the winter months. With the increased temperatures, a greater percentage of spring and fall precipitation may fall as rain, resulting in shallower snowpacks that will begin melting earlier in the spring. The shift in form (rain/snow) and timing of precipitation has significant implications for watersheds that respond to the buildup and melt of the seasonal snowpack.

Increasing temperature and precipitation would result in steadily increasing ET. Evapotranspiration is expected to exceed baseline conditions for all months except for July and August, when ET would be severely limited by available soil moisture. In response to the shift in the volume and timing of snowpack, the discharge from the watershed is expected to begin earlier and reach its peak earlier in the summer as available snowpack is depleted.

Given winters with more precipitation and warmer temperatures than present, infiltration is expected to increase for all months except for July and August when snowpack is depleted, precipitation is lower, and any available precipitation would be used to meet the high ET demand.

The increase in overall precipitation and infiltration that exceeds the increase in ET would result in a rising water table, again for all months except for July and August. A rising water table would reduce the volume of pyrite oxidation zone (for water table depths <50 m) and thus reduce overall concentrations of sulfate in the unsaturated and saturated zones. Concentrations would decrease for all months except for a slight increase from May through August as the water table lowers in response to depleted snowmelt. Overall, the loads of sulfate exported from the watershed would be greater than baseline.

Conclusions

A WEBMOD model was used to assess the effects of climate change on high-elevation mineralized Rocky Mountain watersheds. Baseline conditions describe a watershed with 60 percent of precipitation falling as snow with melting and runoff from May through September. In simulations using predicted climate changes, the percentage of precipitation falling as rain increased as did overall infiltration that recharges groundwater. Greater recharge resulted in reduced sulfate concentrations in the unsaturated and saturated zones for much of the year, although a net increase of sulfate load was simulated as a result of greater overall discharge. Evapotranspiration is expected to increase but not enough to offset the increase in precipitation. The simulations presented here assume no changes to the vegetation; if the vegetation shifts to one capable of transpiring significantly greater amounts of water given the same temperature and water availability, then the predicted rise in water table becomes less certain. The inclusion of more realistic plant and transpiration models into WEBMOD are high priorities.

Acknowledgments

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Utilizing Long-Term ARS Data to Compare and Contrast Hydroclimatic Trends from Snow and Rainfall Dominated Watersheds

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Abstract

The U.S. Department of Agriculture–Agricultural Research Service, Northwest and Southwest Watershed Research Centers have operated the Reynolds Creek Experimental Watershed (RCEW) in southwestern Idaho and the Walnut Gulch Experimental Watershed (WGEW) in southern Arizona since the 1950s. Each watershed is densely instrumented with a variety of hydrometeorological instrumentation and has multiple gauged subwatersheds spanning a range of spatial scales. These watersheds have yielded an extensive knowledge base of watershed processes over multiple decades of use as outdoor hydrologic laboratories. Both research centers have published data reports in *Water Resources Research* describing the RCEW and WGEW and their associated characteristics and observational databases. Precipitation and runoff generation in RCEW is dominated by snow and snowmelt processes, while WGEW is dominated by thunderstorm-generated rainfall during the summer monsoon. Mean annual temperatures in the continental United States have increased from 1 to 3°C since the establishment of these experimental watersheds and this change has affected water supply, hydrology, and watershed response at both locations. This study compared and contrasted hydroclimatic variables at these experimental watersheds, including temperature, precipitation, and streamflow. Monthly, seasonal, and annual data of temperature, precipitation and runoff were tested for significant trends in these variables.

Keywords: experimental watersheds, trends, temperature, precipitation, runoff

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Introduction

To understand how variations in climate, land use, and land cover will affect water supply, ecosystems, and natural resources, we must have access to long-term hydrologic and climatic databases. Data from watersheds that include significant human activities, such as grazing, farming, irrigation, and urbanization, are critical for determining the signature of human-induced changes on hydrologic processes and the water cycle. One of the primary components of effective watershed research is a sustained, long-term monitoring and measurement program. The U.S. Department of Agriculture–Agricultural Research Service (USDA–ARS) Experimental Watershed Network is one such example (Goodrich et al. 1994, Slaughter and Richardson 2000). Two of the ARS Experimental

Watersheds in the Western United States will be featured in this study with the intent of expanding the analysis to other ARS watersheds in the future. They are the Reynolds Creek Experimental Watershed (RCEW), a 239-km^2 drainage in southwestern Idaho near Boise, and the Walnut Gulch Experimental Watershed (WGEW; 149 km^2) in southeastern Arizona near the town of Tombstone. These and the other ARS watersheds are operated as outdoor hydrologic laboratories in which watershed research is supported by long-term monitoring of basic hydroclimatic parameters.

RCEW consists of its drainage area and a third-order perennial stream draining north to the Snake River and ranges in elevation from 1,101 m above mean sea level (amsl) to 2,241 m amsl. About 77 percent of the watershed is under public ownership, with the remainder being privately owned. Primary land use of the watershed is livestock grazing with some irrigated fields along the creek at lower elevations. There is wide diversity in local climate, geology, soils, and vegetation across the Reynolds Creek landscape. Annual precipitation varies from 230 mm at the northern lower elevations to over 1,100 mm in the higher southern regions where 75 percent or more of annual precipitation occurs as snowfall. The ecology and hydroclimatology of RCEW are representative of much of the interior mountain west and Great Basin.

WGEW ranges in elevation from 1,220 to 1,950 m amsl. Its streams are ephemeral with uplands of desert shrubs dominating the lower two thirds of the watershed and desert grasses dominating the upper one third. The climate at WGEW is classified as semiarid with a mean annual temperature at Tombstone of 17.7^{o}C and mean annual precipitation of 312 mm. The precipitation regime is dominated by the North American Monsoon with about 60 percent of the annual total coming during July, August, and September. Summer events are localized short-duration, high-intensity convective thunderstorms, and winter storms are generally slower moving frontal systems. Virtually all runoff is generated by summer thunderstorm precipitation and runoff volumes, and peak flow rates vary greatly with area and on an annual basis.

More detailed descriptions of both RCEW and WGEW have been presented in special sets of papers in *Water Resources Research* (Slaughter et al. 2001, Moran et al. 2008). Research at RCEW continues to be supported by monitoring runoff at 9 weirs, 32 primary and 5 secondary meteorological stations, 26 precipitation stations, 8 snow courses and 5 snow study sites, 27 soil temperature and moisture measurement sites with 5 subsurface hill-slope hydrology sites, and 5 eddy covariance systems. WGEW contains 30 instrumented watersheds for runoff, of which 17 also monitor sediment, 88 precipitation gauges, 3 meteorological stations, 2 eddy covariance stations, and 24 soil moisture monitoring sites (5 with depth profiles). The objective of this study is to carefully examine observations common to both experimental watersheds to quantify the magnitude of climate warming and concurrent changes in precipitation and streamflow over the past 40 to 50 years.

Methods

Observations

Observations of daily maximum and minimum temperature (Tmax, Tmin), precipitation, and runoff were selected from RCEW and WGEW for slightly different time periods depending on installation of instrumentation, but most were initiated in the early to mid 1960s. For RCEW, temperature, precipitation, and runoff were analyzed for the period 1962–2006. Because of significant elevation change in RCEW and its importance to rain-snow precipitation phases, three meteorological stations and runoff weirs were selected, covering an elevation range of over 900 m. In RCEW, precipitation, Tmin, and Tmax observations were co-located. The low-elevation site (RC-Low, site 076, elevation 1,207 m) is located in a relatively broad, flat valley bottom only 108 m above, but nearly 10 km distant from, the RCEW outlet weir. Site vegetation is Wyoming big sagebrush and is typical of low elevations in RCEW. The mid-elevation site (RC-mid, site 127) is located on the eastern side of the basin near the midpoint elevation of the watershed (1,718 m). Site vegetation is dominated by low sagebrush and is typical of mid-elevation vegetation on the eastern side of RCEW. The high-elevation site (RC-high, site 176, elevation 2,093 m) is near the southern rim of the RCEW in an exposed area where a few trees and larger shrubs offer limited shelter from the wind. Site vegetation is a mix of shrubs including mountain big sagebrush, snowberry, and buckbrush. Adjacent to the site are Douglas fir and a few Aspen trees.

For WGEW, the long-term temperature observations are located adjacent to the ARS field headquarters in Tombstone. It is also adjacent to a large surface mining operation, and because of concerns about its affect on the temperature recording station, the Tombstone station was compared to 10 nearby long-term temperature records in Cochise County obtained from the National Climate Data Center. For the period 1961–2009, no significant differences between the Tombstone and adjacent records were found, but to minimize the impact of data gaps (approximately 10 percent of the data are missing) daily data were averaged for the 10 stations, not including any missing values. For precipitation and runoff, three subwatersheds were selected, spanning a range of spatial scales (Table 1). Precipitation records were analyzed for the period 1957–2010 and runoff for the period 1964–2010.

Table 1. Streamflow measurement stations.

Experimental watershed	Name	Elev. (m)	Drainage area (km^2)
RCEW	Reynolds Mtn. East	2,022	0.39
	Tollgate	1,404	55.0
	Outlet	1,099	238.0
WGEW	WG11	1,427	6.35
	WG6	1,334	81.5
	WG1	1,219	131.0

In RCEW, point measurements of precipitation were used from the same meteorological stations where temperature was recorded. For WGEW, areal average precipitation depths were computed over the three subwatersheds using the dense rain gauge network (approx. 0.570 gauges per 1 km^2) according to the procedure described in Goodrich et al. (2008).

Trend Analysis

Each daily time series was aggregated into months, seasons of a water year, and water years for trend analysis. The three-month seasons, starting from the beginning of the water year, are October, November, and December (OND); January, February, and March (JFM); April, May, and June (AMJ); and July, August, and September (JAS). Trends in air temperature, snow, and streamflow data were computed using two methods: least square (LS) linear regression and Sen's slope (SS) estimator. Although the LS method is in common use, the slope of the regression can be sensitive to autocorrelation and extreme values.

To address this problem, Hirsch et al. (1982, 1991) proposed Sen's slope estimator (Sen 1968), a nonparametric method to detect and estimate the magnitude of temporal trends in hydrologic data. This method computes slopes between all data pairs and estimates the overall representative slope as the median value among all possible slope values.

Following the analysis presented by Nayak et al. (2010), significance of these trends will be evaluated using the nonparametric Mann-Kendall statistic at α = 0.10, 0.05, 0.01, and 0.001 levels (Hirsh and Slack 1984, Yue et al. 2002 a). This statistic has the advantage of testing for consistency in the direction of change for temporally ordered data and is unbiased by the magnitude of change. Two methods will be applied to reduce the influence of autocorrelation on statistical significance of trends in time series data. First, the Mann-Kendall test with prewhitening (MK-PW), as suggested by Zhang et al. (2001), will be applied to eliminate the effects of serial correlation in the Mann-Kendall test. Second, the trend free prewhitening (MK-TFPW) approach (Yue et al. 2002 b) will be applied to minimize the effect of the MK-PW approach on the magnitude of the slope and significance of the trend present in the original data series. This approach also nonparametrically scores the significance of the trend with a sequence of symbols from weak to strong— +, *, **, *** —corresponding to the MK-TFPW test at significance levels of α = 0.10, 0.05, 0.01, 0.001, respectively.

Results

Temperature

In Figure 1, the average annual Tmin and Tmax temperatures are plotted as a function of time for the mid-elevation RCEW station (RC-mid, elevation 1,652 m) and for the average of the 10 Cochise County stations in and surrounding WGEW (WG-CC) with computed trend lines from the Sen's slope estimation method.

There was a significant increasing trend in temperature for the annual average series with a slightly stronger trend in the Tmin series (Table 2). At the seasonal level, significant increasing trends in Tmin were observed for all seasons and stations except the RCEW-Low station and the WG-CC stations during the water year season OND. Significant trends in Tmax were weaker in the case of the WG-CC data: MK-TFPW of * instead of ** or ***. In RCEW, the Tmax trends were significant in the summer (JAS) at all three elevations and in the fall (OND) and winter (JFM) at RC-High. In RCEW, the increasing temperatures had a profound effect on changing the phase of precipitation from snow to rain with the mid- and low-elevation

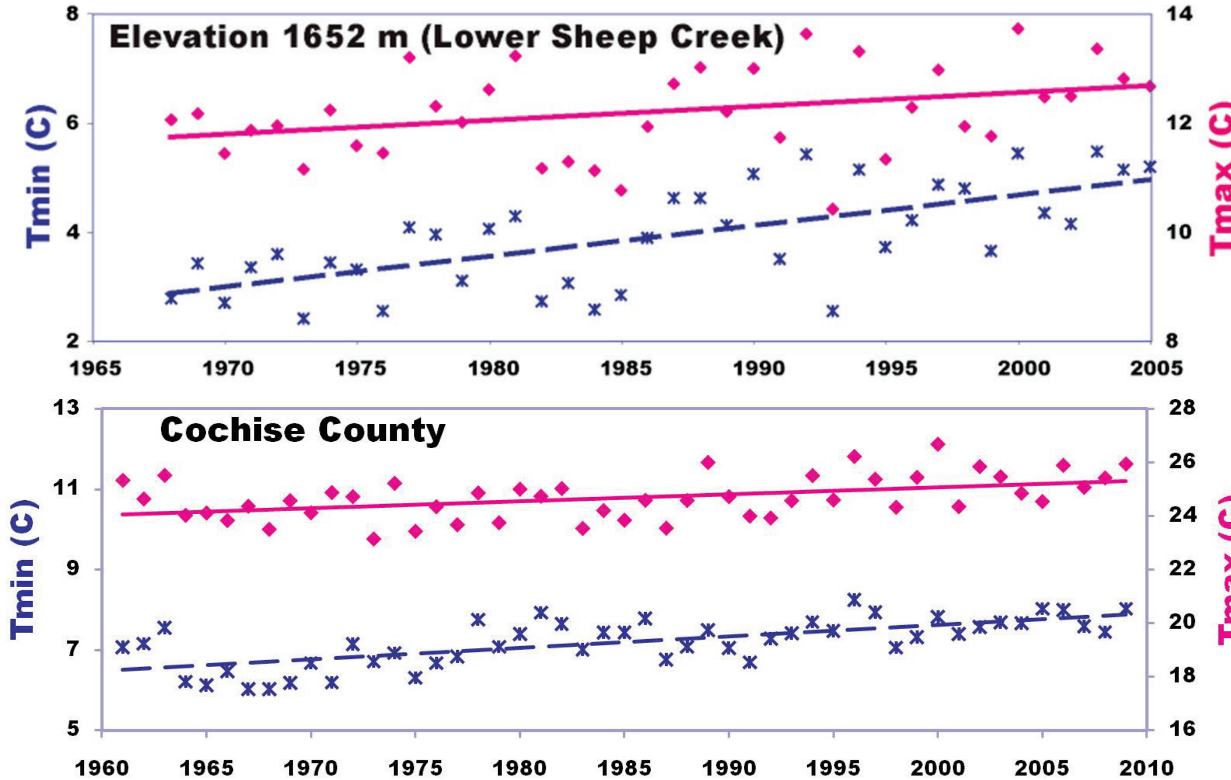

Figure 1. Average annual maximum and minimum daily temperatures for RC-Mid and WG-CC.

stations becoming dominated by rainfall in the later record. Nayak et al. (2010) discusses this result in more detail as well as its effect on snow-water equivalent trends.

Table 2. Annual average trends in Tmin and Tmax.

		Tmin		Tmax	
Station	Elev. (m)	Slope (°C/dec)	Trend sig.[a]	Slope (°C/dec)	Trend sig.[a]
RC-High	2,093	0.45	**	0.35	**
RC-Mid	1,652	0.57	**	0.29	**
RC-Low	1,200	0.36	**	0.20	*
WG-CC[b]	--	0.29	***	0.26	**

[a](MK-TFPW) test significance level: * for $\alpha = 0.05$, ** for $\alpha = 0.01$, and *** for $\alpha = 0.001$

[b]10 Stations in and around the WGEW in Cochise Co.

Precipitation

Very few significant trends in precipitation were detected for either RCEW or WGEW. In WGEW, only a weak (+) to moderately (*) significant trend was found for February precipitation over each of the three subwatersheds. However, precipitation in February comprises just over 5 percent of the annual average and is characterized by low-intensity frontal storms that rarely generate any runoff. An increasing trend in non-summer precipitation for a group of WGEW raingages for the period from 1956 to 1996 was also found by Nichols et al. (2002). However, when an additional 10 years of data were analyzed by Goodrich et al. (2008), there was no trend, due to the influence of multiyear droughts in the more recent data.

Few statistically significant temporal trends ($\alpha = 0.10$) were found in RCEW. A slight decline during the summer months of approximately 1 percent was the only significant trend. Note that summer precipitation represents a small fraction of annual precipitation, so the affect on other seasons is not significant (at the 90-percent level). The decrease in summer precipitation represents a redistribution of water year precipitation to fall and spring at mid to high elevations (Nayak et al. 2010).

Runoff

At RCEW, most of the flow during a water year occurs from March to June. During these months, nearly 90 percent of annual streamflow occurs at the high-elevation Reynolds Mountain East (RME) weir, 82 percent at the mid-elevation Tollgate (TG) weir, and 70 percent at the RCEW outlet weir. Variation of annual runoff volume is very large, with standard deviations of at least 50 percent of the annual mean at all sites. The Sen's slope values were negative over the 1962–2006 period of record for all three weirs (Table 3). However there were no significant temporal trends. At the monthly scale, streamflow has shifted toward earlier periods with a shift to late winter and early spring flows at the RME and TG weirs. Streamflow has increased in March and April and decreased in May and June. At the RCEW outlet weir, consistency in trends is less conclusive, with an increase in May flow as the only significant trend (* with $\alpha = 0.05$). It should be noted that spring and summer diversions to irrigation below the TG weir likely confounded trend analysis at the outlet weir. Examined in the context of elevation, there is a strong gradient to this shift. The high-elevation RME weir exhibits a weak but significant (+ with $\alpha = 0.10$) increase in flow in March and April, but the mid-elevation TG weir has significant (* with $\alpha = 0.05$) increase in April flow. The outlet weir exhibits a significant (*) increase in flow in May (Nayak et al. 2010).

Table 3. Water year stream discharge trends.

Watershed	Mean (10^6 m^3/WY)	Stream discharge	Sen's slope (10^6 m^3/decade)
Reynolds Mtn. East	0.21	0.10	–0.008
Tollgate	13.4	7.5	–0.75
Outlet	17.1	12.4	–1.66
WG11	0.066	0.10	–0.013
WG6	0.40	0.44	–0.040
WG1	0.37	0.38	–0.038

In WGEW, on average, there are roughly nine runoff events per year from each of the subwatersheds. Virtually all runoff occurs from the summer monsoon, and infrequent runoff occurs in the early fall as a result of tropical cyclones. While the onset of the monsoon is variable from year to year, the summer months of July, August, and September (JAS) typically contain the monsoon season and therefore almost all runoff production. Like RCEW, the Sen's slope values were negative over the 1964–2010 period of record for all three subwatersheds during this season (Table 3). The trends for WG1 and WG11 were found to be weakly (+) and moderately (**) significant, respectively, and the trend for WG6 was not significant. At the monthly level, a strongly significant (**) decreasing trend was found for all three subwatersheds for the month of September, which appears to be counterintuitive to the findings of Grantz et al. (2007). They found a significant delay in all stages of the monsoon in recent decades with a decrease in rainfall during July and an increase in rainfall during August and September. However, intrastorm precipitation intensity is the primary determinant of runoff generation.

Conclusions

RCEW and WGEW both exhibit moderate to strongly significant trends of increasing temperature that are in agreement with other studies of temperature trends in the western United States and Canada (e.g., Trenberth et al. 2007). The rate of increase in Tmin at the annual temporal scale (+0.29 to 0.57°C per decade) is greater than Tmax (+0.2 to +0.35°C per decade). In RCEW, this has resulted in the crossing of important thermal thresholds. Consequently, the snow season is at least a month shorter than it was in the mid 1960s (Nayak et al. 2010). As rain becomes a larger proportion of the total annual precipitation, runoff occurs earlier in the year.

Changes in precipitation and runoff in the WGEW were less pronounced. While there were moderately significant trends of decreasing runoff in September, it is unclear if this is related to changes in the seasonal onset of the monsoon or a change in rainfall intensity. These decreases will be investigated in more detail in future work. Sensitivity of watershed response will also be examined using runoff to rainfall ratios for each of the watersheds. This study will require improved methods to determine the spatial distribution of precipitation, which is especially challenging in the RCEW where wind-driven redistribution of snow is common. Trends in extremes will also be investigated. Extension of this work to ARS and other experimental watersheds across a wide range of hydroclimatic regions will then be undertaken.

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Analysis of Trends in Climate, Streamflow, and Stream Temperature in North Coastal California

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Abstract

As part of a broader project analyzing trends in climate, streamflow, vegetation, salmon, and ocean conditions in northern California national park units, we compiled average monthly air temperature and precipitation data from 73 climate stations, streamflow data from 21 river gaging stations, and limited stream temperature data from salmon-bearing rivers in north coastal California. Many climate stations show a statistically significant increase in both average maximum and average minimum air temperature in early fall and midwinter during the last century. Concurrently, average September precipitation has decreased. In many coastal rivers, summer low flow has decreased and summer stream temperatures have increased, which affects summer rearing habitat for salmonids. Nevertheless, because vegetative cover has also changed during this time period, we cannot ascribe streamflow changes to climate change without first assessing water budgets. Although shifts in the timing of the centroid of runoff have been documented in snowmelt-dominated watersheds in the western United States, this was not the case in lower elevation coastal rivers analyzed in this study.

Keywords: streamflow, climate, temperature, precipitation, northern California

Introduction

In north coastal California, daily, seasonal, and decadal variations in abiotic drivers (e.g., precipitation, fog, streamflow, and temperatures of air, ocean, and streams) regulate many ecological processes, including the distribution of vegetation and wildlife and frequency of disturbances from fires, floods, landslides, and biotic pests. However, the exact nature of the linkages between abiotic drivers and the direct and indirect effect of these drivers on species of concern and their habitat are not well understood.

In addition to needing greater understanding of the basic linkages between abiotic and biotic ecosystem elements, the question of climate change is of increasing concern to land managers. They need to understand how climate change has already affected natural resources and whether other changes may be looming. Without this understanding it is increasingly difficult to judge the effects of management efforts (e.g., stream restoration), evaluate the resilience of existing habitats, or plan future management actions.

Complicating a manager's ability to respond to climate change effects is the common assumption of stationarity—the idea that natural systems fluctuate within an unchanging envelope of variability (Milly et al. 2008). The stationarity assumption is being compromised by major shifts in background environmental conditions. These major shifts may be changing the timing, magnitude, and intensity of critical abiotic elements in this region. In addition, the common assumption that restoration planning can use historical reference conditions as a goal may not be valid if extrinsic drivers display nonstationarity. Consequently, the understanding of trends, variability, and interactions among abiotic drivers is needed to inform restoration strategies, prioritize restoration sites, and implement scenario planning to foster strategic thinking about future conditions and management alternatives.

The National Park Service (NPS), through its Inventory and Monitoring Program, has monitored many abiotic drivers and has conducted several biological surveys. A major concern is the decline of salmon populations in coastal California park streams. In central and northern California, several salmon and steelhead populations have been in decline for years and many are federally listed as threatened or endangered. In 1997, coho salmon *(Oncorhynchus kisutch)* were federally listed as

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Table 1. Gaging stations used in analysis.

Station name	Agency	County	Period of record (water years)	Drainage area (km^2)
Bull Creek near Weott	USGS	Humboldt	1961–2010	72.8
Eel River at Scotia[1]	USGS	Humboldt	1911–2010	8,062.7
Elder Creek near Branscomb	USGS	Mendocino	1968–2009	16.8
Lacks Creek	USGS/NPS	Humboldt	1981–2010	43.8
Little Lost Man Creek near Orick	USGS/NPS	Humboldt	1975–2010	9.0
Little River near Trinidad	USGS	Humboldt	1956–2010	104.9
Mad River near Arcata[2]	USGS	Humboldt	1951–2010	1,256.2
Mattole River near Petrolia	USGS	Humboldt	1951–2010	634.6
North Fork Caspar Creek	USFS[3]	Humboldt	1964–2003	4.7
Olema Creek	NPS	Marin	1998–2010	40.0
Panther Creek near Orick	USGS/NPS	Humboldt	1980–2010	15.7
Prairie Creek above Brown Creek	NPS	Humboldt	1990–2008	10.6
Prairie Creek above May Creek	NPS	Humboldt	1991–2008	32.6
Redwood Creek at O'Kane	USGS	Humboldt	1973–2010	175.3
Redwood Creek at Orick	USGS	Humboldt	1954–2010	720.0
Redwood Creek at GOGA	NPS	Marin	1998–2010	18.1
San Geronimo Creek	MMWD[4]	Marin	1980–2009	24.3
Smith River at Crescent City	USGS	Del Norte	1932–2010	1,590.3
South Fork Caspar Creek	USFS	Humboldt	1964–2003	4.2
South Fork Eel River at Leggett	USGS	Mendocino	1966-2009	642.3
Van Duzen River at Bridgeville	USGS	Humboldt	1951–2008	575.0

[1]Flow regulated by Lake Pillsbury and diversion through Potter Valley power plant at drainage area of 883 km^2
[2]Flow regulated by Ruth Reservoir at drainage area of 313 km^2
[3]U.S. Forest Service Pacific Southwest Research Station
[4]Marin Municipal Water District

threatened in the Southern Oregon/Northern California Evolutionary Significant Unit, including Redwood National Park, and as endangered in the Central California Coast.

Factors affecting coho populations include elevated stream temperatures (Welsh et al. 2001), water withdrawals, dams, loss of spawning and rearing habitat, and extreme hydrologic events (Carlisle et al. 2009). Millions of dollars are being spent in coastal parks on watershed and stream restoration projects. As the NPS plans salmon restoration activities in coastal watersheds, it is critical to understand the abiotic factors and interactions that affect salmonid populations (MacCall and Wainwright 2003, Battin et al. 2007).

The objectives of this study are to identify trends in climate drivers and hydrologic regimes that may be having local and regional effects on salmon populations in north coastal California streams, with an emphasis on four national park units, Redwood National Park (REDW), Golden Gate National Recreation Area (GOGA), Point Reyes National Seashore (PORE), and Muir Woods National Monument (MUWO).

Study Area

The climate and streamflow monitoring stations used in this study are located along the north coast of California from the San Francisco Bay area (37°N) to the Oregon border (42°N). The region has a Mediterranean climate with mild, rainy winters and generally cool, dry summers. Coastal fog is a common occurrence in the summer months. Mean daily minimum temperatures in January range from 0.4 to 7°C and mean daily maxima in July range from 14 to 30°C. Mean annual precipitation is moderate to high, generally more than 100 cm, but as low as 70 cm in the southern part of the region and more than 200 cm in the northern mountains. The region is dominated by the Coast Range, which separates the coastal area from the hotter Central Valley farther inland.

Table 2. P-values for trends in air temperature and precipitation at four national park units.

	REDW[1]	GOGA	PORE	MUWO
Average annual precipitation	0.512	0.906	0.772	0.451
Average annual temperature				
Maximum	0.269	**<0.001**[2]	**<0.001**	**<0.001**
Mean	0.810	**<0.001**	**<0.001**	**<0.001**
Minimum	0.414	**<0.001**	**<0.001**	**<0.001**
Average monthly temperature (maximum)				
January	**0.095**	**<0.001**	**0.002**	**<0.001**
February	0.700	**0.007**	**0.019**	**0.002**
March	0.692	**0.010**	**0.034**	**0.004**
April	**0.027**	0.152	0.123	**0.079**
May	0.960	**0.002**	**0.002**	**0.002**
June	0.257	0.923	0.626	0.993
July	0.854	0.538	0.406	0.287
August	0.214	**<0.001**	**<0.001**	**0.080**
September	0.578	**0.005**	**0.006**	**0.032**
October	0.243	**0.001**	**0.003**	**0.003**
November	**0.022**	0.443	0.762	0.414
December	0.843	**0.042**	0.187	**0.016**

[1]REDW, Redwood National Park; GOGA, Golden Gate National Recreation Area; PORE, Point Reyes National Seashore; MUWO, Muir Woods National Monument
[2]Values in bold show statistically significant increasing trends at the 90% level.

Methods

The PRISM (Parameter-elevation Regressions on Independent Slopes Model) Climate group at Oregon State University provided air temperature and precipitation data. The PRISM group incorporated climate data from specific points (climate stations) and extrapolated those data to describe air temperature and rainfall in selected national park units (http://prismmap.nacse.org/klamath). Long-term trends in maximum, mean, and minimum temperatures and precipitation were examined through linear regression. Streamflow records were obtained for gaging stations operated by the U.S. Geological Survey (USGS), REDW, GOGA, and PORE. Most datasets consisted of mean daily streamflow and peak flows. Streamflow records from 67 gaging stations located along north coastal California were examined. Many of the rivers have altered hydrologic regimes because of impoundments or diversions, and others had short (<10 years) periods of record or large data gaps. Nineteen stations with adequate records and unimpaired flow regimes were retained for more in-depth analysis, as well as two rivers with dams in their headwaters (Table 1). Drainage areas ranged from 4.2 to 8,063 km^2. Some stations also included stream temperature records from in situ temperature data loggers with sampling intervals that ranged from 15 to 60 minutes. Daily maximum, mean, and minimum water temperatures were calculated from these datasets. Following the approach of Gu et al. (1999), we used air temperature and streamflow as independent variables with stream temperature as the dependent variable in a multiple regression analysis. Because salmon migration, spawning, and rearing success are dependent on magnitude and timing of flow, we analyzed the 7-day low flow and the dates of percentiles of cumulative runoff volumes. Cumulative runoff curves were constructed by accumulating mean daily discharge for each water year (WY; October 1–September 30). Various percentiles of runoff could then be calculated. For example, the timing of the center of mass of flow (the date by which 50 percent of the runoff has occurred, or CT) was calculated for each station.

Results

The PRISM results cover the period of 1895 to 2007. Figure 1 shows an example of PRISM output for GOGA. Park units in the San Francisco Bay area (GOGA, MUWO, and PORE) showed a statistically significant increase in annual maximum, mean, and minimum air temperatures of about 1°C during the last 100 years (Table 2). On a monthly basis, significant increases were detected in January, February, March, May, August, September, and October. Farther north, in REDW, air temperature did not increase significantly. No trend in annual precipitation was detected, but all park units showed statistically significant decrease (at the 90-percent level) in September rainfall during the last century, a decrease of 10 mm in the south and 30 mm in the north.

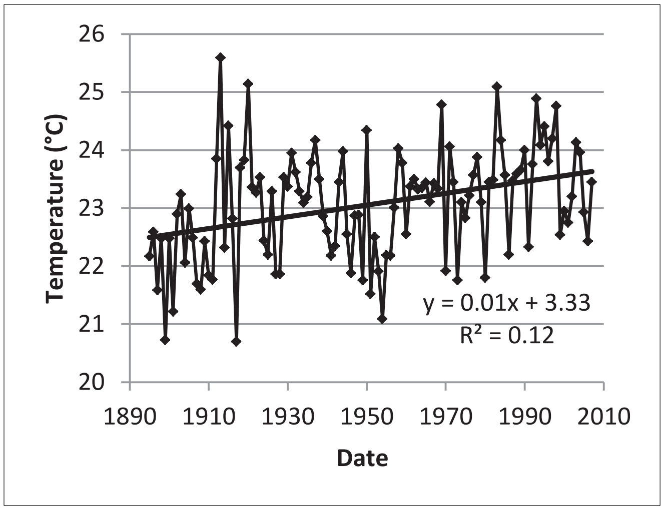

Figure 1. Average daily maximum air temperature for August calculated for GOGA by PRISM.

Timing of runoff was assessed at 21 gaging stations. CT ranged from December to April but commonly occurred in early February (Figure 2). CT was fairly synchronous among watersheds from the San Francisco Bay area north to the Oregon border (Figure 2). Since the late 1940s, CT has shifted towards earlier in the water year in many mountainous rivers of western North America (Stewart et al. 2004). In contrast to trends documented in those mountainous watersheds with large snowmelt components, none of the lower elevation coastal streams in this study showed a significant trend at the 0.10 level of earlier or later runoff based on CT values.

Another streamflow characteristic of concern for fish is summer low flow. The 7-day low flow decreased at 10 of the stations over the period of record. Redwood Creek at Orick is an example of this decrease (Figure 3). Other stations that exhibited a statistically significant decrease at the 90-percent level were Mattole, Van Duzen, and South Fork Eel rivers and Lacks, Olema, North Fork Caspar, Elder, Redwood at O'Kane, and San Geronimo creeks. Statistical significance does not automatically imply a biological significance. Nevertheless, lower flows are associated with less pool volume for summer rearing of salmon or dry river reaches. In addition, solar radiation can heat a smaller mass of water faster than greater water volumes, leading to elevated stream temperatures.

Coho salmon are very sensitive to elevated stream temperatures, and stream temperature tends to peak in late July or early August. The maximum weekly average temperature (MWAT) in the southern streams (Olema Creek in PORE and Redwood Creek in GOGA) ranged from 14.2°C in WY2010 to 19.1°C in WY2006, based on measurements from WY2003 to 2010. MWATs in Prairie Creek above May Creek in REDW were somewhat cooler, ranging from 13.1°C to 15.8°C from WY1997 to 2010. In contrast, MWATs in Redwood Creek at O'Kane were consistently above 16.8°C, the temperature restriction for coho assessed by Welsh et al. (2001). At this station MWATs ranged from 19.8°C to 23.9°C, based on measurements from WY1997 to 2010.

The relationship between air temperature (as a surrogate for solar radiation, which directly affects water temperature) and stream temperature was assessed for Redwood Creek near the O'Kane gaging station, where several years of air and stream temperature data were available (Sparkman, California Department of Fish and Game, personal commun.). For the months of June, July, and August, daily maximum stream temperature was strongly correlated with both daily maximum air temperature and mean daily streamflow at the gaging station ($n = 400$, $r^2 = 81$ percent, p-value <0.0001).

Discussion

Gaging station records have different periods of record, so a full statistical comparison among the stations could not be done. Consequently, the trends noted in this paper should be considered a starting point for more in-depth analyses in specific watersheds. For example, a decrease in summer low flows can be problematic for salmonid habitat, but it cannot simply be ascribed to climate change without assessing other possible influences. Minor diversions for domestic or agricultural use are not quantified, and some illegal water diversions may also influence summer low flows. Sedimentation in river beds may cause surface runoff to go subsurface, decreasing the amount of available surface flow in the summer. Regrowth of trees in formerly logged watersheds can increase the evapotranspiration demand and decrease summer low flow (Reid and Lewis, in press). A decrease in summer

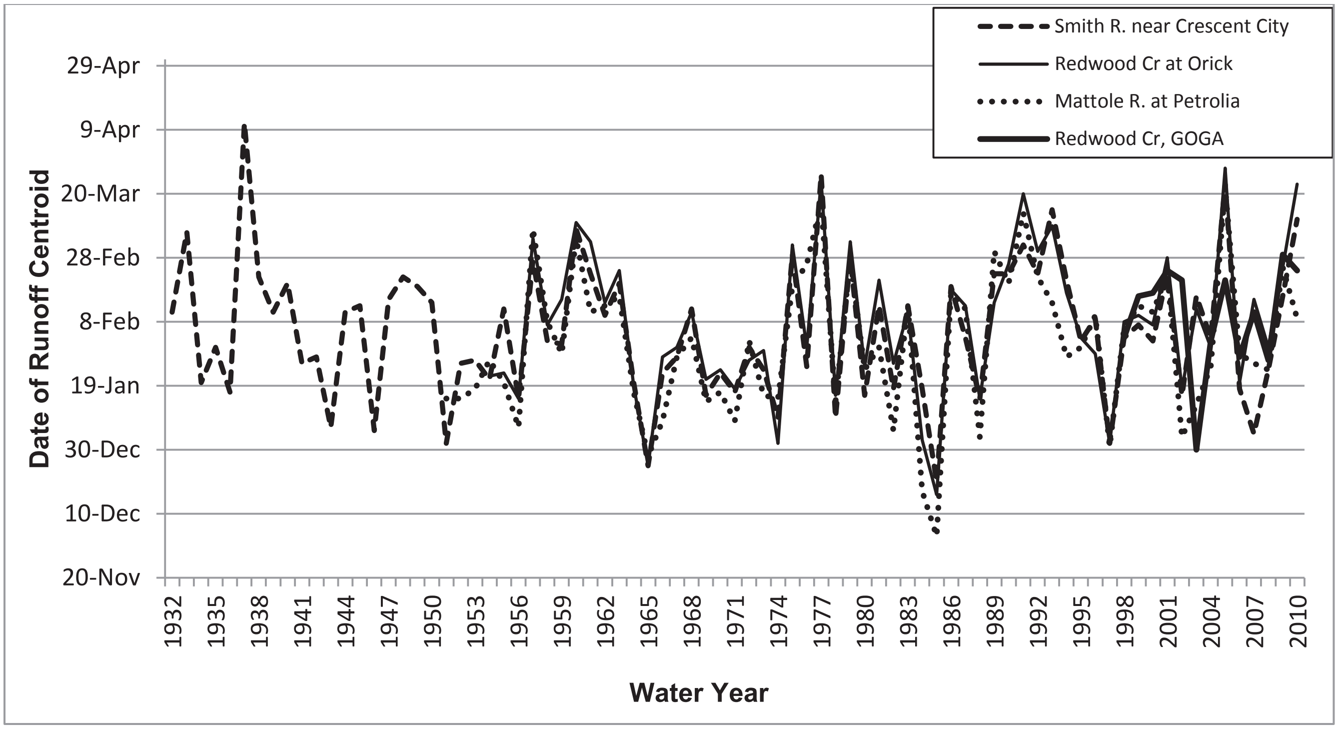

Figure 2. Timing of runoff centroid for four rivers, spanning the coastal region from the Oregon border (Smith River near Crescent City) to the San Francisco Bay area (Redwood Creek at GOGA).

fog can also increase the evapotranspiration demand. Preliminary analysis of fog frequency at two sites within the study area suggests a decrease in fog in recent years (Johnstone and Dawson 2010). Nevertheless, the full spatial and temporal distribution of fog is not known, and further analysis based on remote sensing data of coastal fog is in progress. Future research will also assess changes in vegetative cover during the period of streamflow records.

Interannual variations in climate are linked to the El Niño/Southern Oscillation phenomenon (ENSO). Annual flood peaks in coastal rivers at the north end of our study area are significantly smaller during El Niño conditions; south of 35°N (south of San Francisco) they are significantly larger during an El Niño phase (Andrews and others 2004). Future research will examine trends in other aspects of runoff (in CT, low flows, etc.) with respect to ENSO events.
Elevated stream temperatures in Redwood Creek at O'Kane in 2006 and 2009 led to fish kills (Sparkman, California Department of Fish and Game, personal commun.), and stream temperatures at many locations along Redwood Creek in REDW are above the temperatures preferred by coho (Welsh et al. 2001, Madej et al. 2006). Increases in stream temperature can be exacerbated by the diversion of streamflow for agriculture or other uses, or if reduced rainfall decreases summer baseflow. Consequently, if air temperature increases or summer flow decreases, we can expect warmer summer stream temperatures, adding to stressful conditions for summer-rearing salmonids and for unique runs of adult summer steelhead holding in pools throughout the summer.

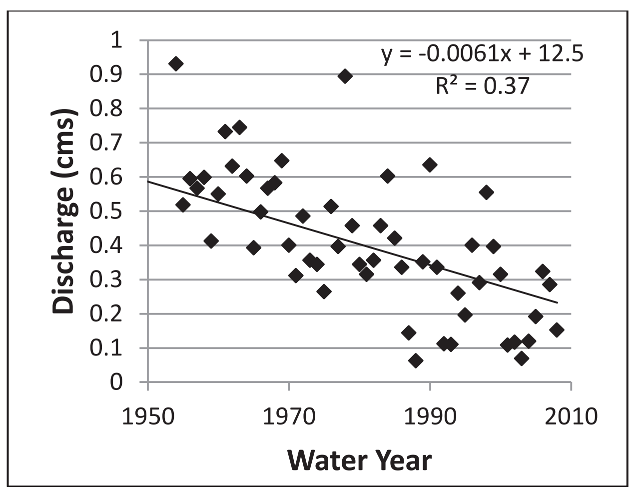

Figure 3. Seven-day minimum flow at Redwood Creek at Orick, USGS Station #11482500.

Conclusions

The PRISM climate data for four national park units in north coastal California show increases in annual maximum, mean, and minimum air temperatures of about 1°C during the last century for the three sites near San Francisco Bay, but not at Redwood National Park farther north. September rainfall has decreased at all four units over this same time period. Summer low flow has decreased in several streams in this region, although the causes for the decrease are not fully understood. Daily maximum stream temperature is correlated with daily maximum air temperature and mean daily streamflow. Maximum weekly average stream temperatures in some of the streams exceeded the value preferred by coho salmon (<16.8°C).

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Evidence of Climate Change in the Streamflow and Water Temperature Record in the Missouri River Basin

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Abstract

The streamflow and water temperature record was examined for evidence of changing climate conditions in the Missouri River Basin (MRB). Observations of 52 years of continuous record (1957–2008) at about 200 U.S. Geological Survey (USGS) gages indicate that streamflow conditions are changing in the MRB. Trends are evident in the annual mean discharge using the non-parametric Kendall Tau test. There is a strong geographic distribution of the trends, which are generally upward in the eastern and downward in the western parts of the basin. For example, most significant trends ($p \geq 0.10$) are upward at stations in Colorado, North Dakota, South Dakota, Iowa, and Missouri, whereas most significant trends ($p \geq 0.10$) are downward at stations in the mountain west (Montana and Wyoming). The reduced runoff in the western basin has resulted in depleted storage in mainstem reservoirs on the Missouri River for most of the last decade. In addition to mean annual streamflow, mean monthly trends have also changed for many stations. A simple shift of flow, however, from one season to another is not apparent, but trends, both upward and downward, occur across many or all months of the year. The amount of water that these trends of recent decades represent, compared to earlier decades, is profound in some cases. The historical record of water temperature for the Missouri River was compiled and examined for evidence of changing conditions. The long-term records examined consisted of three USGS stations—Missouri River at Toston, Mont. (station 06054500), Missouri River below Garrison Dam, N. Dak. (stations 06339000 and 06338490), and Missouri River at Nebraska City, Nebr. (station 06807000)—and reservoir outflow records collected by the U.S. Corps of Engineers at Garrison Dam in North Dakota and Oahe and Gavins Point Dams in South Dakota. All available data were examined for evidence of water temperature change for two indicator parameters: (1) long-term change over the respective period of record, and (2) any change in the date of the annual maximum temperature. Some evidence exists that the river is warming over time, especially for the reach downstream from the reservoirs, as indicated by the Missouri River station at Nebraska City.

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Long-Term Climate Change Controls Stream and Riparian Area Response to Disturbance in the Semiarid Great Basin

J.C. Chambers

Abstract

Climate change can have long-term effects on watershed geomorphology and, thus, on stream channel processes and riparian ecosystem dynamics. In the central Great Basin, paleoecological and geomorphic reconstructions show that drought conditions coupled with wildfires from 2,500 to 1,900 YBP (years before present) resulted in hillslope erosion and sediment deposition in valley bottoms and on side-valley alluvial fans. Because the hillslopes were depleted of sediment, the subsequent trend has been towards channel incision during flood events. The rate and magnitude of incision has been accelerated by anthropogenic disturbances like roads in the valley bottoms. At watershed scales, basin lithology and geomorphometry determine the long-term response of the basins to disturbance and, thus, the extent and composition of riparian vegetation. Watersheds dominated by volcanic rocks are rugged, have high stream power and hypsometric integrals, and are dominated by disturbance-tolerant woody riparian vegetation. Those dominated by sedimentary and metasedimentary rocks have low stream power and hypsometric integrals, are strongly influenced by side-valley alluvial fans, and have stepped-valley profiles that support a high proportion of meadow ecosystems. At local or stream reach scales, the hydrogeomorphic setting determines the response of streams and riparian ecosystems to disturbance and is reflected in the degree of incision and the structure and composition of the riparian vegetation and aquatic biota. Many of the watersheds have adjusted to the new sediment regime, but many others are functioning as non-equilibrium systems. Development of effective management strategies for these watersheds requires an understanding of the long-term effects of climate and the processes controlling the current responses of the streams and riparian ecosystems to disturbance.

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Ecohydrology

Predicting the Impact of Glacier Loss on Fish, Birds, Floodplains, and Estuaries in the Arctic National Wildlife Refuge

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Abstract

In this paper we explore the impacts of shrinking glaciers on downstream ecosystems in the Arctic National Wildlife Refuge. Glaciers here are losing mass at an accelerating rate and will largely disappear in the next 50–100 years if current trends continue. We believe this will have a measureable and possibly important impact on the terrestrial and estuarine ecosystems and the associated bird and fish species within these glaciated watersheds.

Keywords: glaciers, birds, fish, shrubs, arctic, climate

Climate-Driven Change of Glaciers and Its Potential Impact on Physical Hydrology

Glaciers throughout the Brooks Range are losing mass at a rate that is accelerating with time, and most will likely disappear in the next 50 years. Research on McCall Glacier in the eastern Brooks Range documents this accelerated ice loss over the past 50 years (Nolan et al. 2005). It is clear that glacial retreat began in the late 1800s in this region, following the strongest advance since the last glacial maximum. From at least the 1500s to the 1800s CE, these glaciers expanded by storing water from the annual precipitation cycle, but now they are losing this mass, discharging more water than current annual precipitation levels. A variety of modeling predicts disappearance of glaciers in the near future (Delcourt et al. 2008), largely driven by a rise in the late-summer snowline, such that in many recent years there is no remaining accumulation of the past winter's snow. On inland valley glaciers like these, the position of this snowline is likely to be 10 times more sensitive to air temperature than to precipitation (Oerlemans 2001), and our local records show greater changes in air temperature than in precipitation over the past 50 years. McCall Glacier is one of the five largest of the over 400 glaciers in the Arctic Refuge, with an area of about 6 km^2 and an average thickness of about 75 m (Pattyn et al. 2009). Average size of glaciers in the region is about 1 km^2 and likely less than 20 m thick. Our measurements here and at many other glaciers in the area indicate area-averaged ablation rates from 0.5 to 1.0 m/a. Thus, even without sophisticated modeling, it is clear the bulk of the glacial ice here will disappear soon, and our photo comparisons indicate that many glaciers already have disappeared in the past 50 years.

Table 1. Glacierized area as of 1956

	Glacier area (km^2)	Watershed area (km^2)	Percent
Jago	126	2,208	5.7
Okpilak	139	1,011	13.8
Hulahula	116	1,841	6.3
Sadlerochit	38	1,698	2.2

Table 1 summarizes glacierization (percentage of land covered by glacier ice) characteristics of the four most heavily glaciated watersheds in Arctic Alaska, all located within the Arctic National Wildlife Refuge. The average glacierization is 6.2 percent over the entire area, about the same as that in the Hulahula River. These percentages have been decreasing over time as glaciers shrink; we recently acquired new digital elevation models and air photos of nearly all of these glaciers, but as yet we do not yet have updated measurements. While these glaciated watersheds are

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not huge by Alaskan standards, they are still large and located within an ecologically-sensitive area that is expected to be especially vulnerable to the effects of climate change, as we hope to demonstrate in this paper.

Stream discharge here in summer is dominated by glacier meltwater, in contrast to the rest of Arctic Alaska. Nonglaciated watersheds in Arctic Alaska such as the Colville, Sagavanirktok, and Kuparuk Rivers (which collectively drain about 80,400 km^2 of the North Slope) typically issue an average of 61, 44, and 52 percent, respectively, of their annual discharge to the Beaufort Sea during two weeks of snowmelt in the spring (McClelland et al., in press). We installed a discharge gauge on the Hulahula River in fall 2010, but do not yet have a full year of measurements. In the meantime, we have compared rates of river contribution averaged over watershed area to determine relative contributions of precipitation versus glacial melt. Annual precipitation in Arctic Alaska is usually less than 30 cm/a, with roughly half falling as snow and melting in early June. Current glacier ablation rates are usually over 80 cm/a averaged over the glacier area, or roughly 5 cm/a averaged over the watershed areas (assuming 6.2 percent glacierization). So, the glacier contribution is in the same order of magnitude as rain and snow contributions, and after the spring freshet glaciers dominate flow compared to the infrequent rains, much of which gets intercepted during overland flow. As glacier ablation rates continue to rise because of increased warming, the fraction of water contributed by glaciers may initially increase. When glacier reserves are depleted, however, their contribution will plummet.

This paper represents the initial attempt by the authors to integrate our individual research projects to contribute to a multidisciplinary understanding of how climate-driven changes in glaciers may affect ecological trajectories over the next 50 years.

Potential Impacts on Riparian Ecosystems

We have some evidence that indicates that the spread of vegetation within the floodplains of these glaciated watersheds is limited by the geomorphological instability related to increasing glacial discharge. We attempted to assess the effect of glaciers on floodplain stability by comparing time series of vertical air photos at locations along a river fed primarily by glaciers (Jago River) to a river fed by nonglacial sources (Kongakut River). Old and new imagery was opportunistically acquired, and comparisons between images were made where spatial overlap occurred. None of the imagery was acquired when water levels were high so that no vegetation would lie undetected under water. In each area where repeat imagery was overlain, we created a single (virtual) transect zig-zagging from one side of the river to the other and placed points every 50 m along the transect. Transects ranged in length from 5,850 m to 22,850 m, totaling 70,650 m and consisting of 1,413 sample points. The placement of the transects was constrained by the available imagery but otherwise could be considered random within the floodplain; points not within the floodplain were excluded from this analysis.

Three of the five Kongakut River transects showed an increase in vegetation in the floodplain since acquisition of the old imagery (old images acquired between 1948 and 1982). Of the two remaining transects, one showed equal number of vegetated points in old and new imagery, and another contained no vegetation in the floodplain in old or new imagery (Ø). Overall, floodplain vegetation along the Kongakut River increased over time ($p < 0.05$). There was an insignificant positive downstream trend in percent change in vegetated floodplain points, which was 0 percent, Ø, +30 percent, +48 percent, and +33 percent. The single long reach of the Jago River assessed using this technique contained equal number of vegetated points in old and new imagery (0 percent change).

The increase in floodplain vegetation along the Kongakut River is similar to the trend observed in North Slope floodplains west of the Canning River (Tape et al. 2006). Possibly, a decrease in discharge or decrease in aufeis volume is causing the increase in vegetated bars along the Kongakut River. The absence of trend in the Jago River floodplain suggests that glaciated watersheds are not following the same trajectories, but our observations thus far are too limited to make conclusive generalizations. Work is currently in progress to acquire high-resolution air photos of both glaciated and nonglaciated rivers in this area to further assess the role of enhanced glacial discharge on vegetative-growth dynamics.

Potential Impact on Fish Ecology

The loss of glaciers and their meltwater may reduce instream connectivity and cause fish habitats to become fragmented, especially in late summer when anadromous Dolly Varden (*Salvelinus malma*) are returning from estuarine/marine areas to reach spawning and overwintering areas in these glaciated watersheds. As an integral part of the aquatic ecology of Alaska's North Slope, these fish are particularly vulnerable to the effects of climate change (Martin et al. 2009) and also support a number of subsistence fisheries (Pedersen and Linn 2005). Anadromous Dolly Varden occur in most of the larger drainages north of the Brooks Range (Viavant 2009), and like other fish in the region, these populations have adapted to habitats and physical conditions that include a short growing season, extensive ice cover, summer water temperatures <10°C, and long periods of darkness (Reist et al. 2006). During the spring freshet, the mature individuals migrate from overwintering areas in rivers to estuarine and marine waters for summer feeding (Viavant 2005). In glacial streams, these fish primarily return to freshwater in August when glacial meltwater provides adequate discharge to allow migration (Martin et al. 2009). Juvenile fish overwinter in their natal streams for 2–3 years before making their initial journey to saltwater (Fechhelm et al. 1997).

The population of anadromous Dolly Varden in the Hulahula River has been the focus of several studies from the 1970s through the 1990s (Viavant 2009), with more recent, complementary work providing detailed information about Dolly Varden abundance and behavior. Helicopter surveys of index areas in the river estimated relative abundance at 9,575 and 3,653 Dolly Varden in mid-September of 2007 and 2008, respectively (Viavant 2009). Another study used sonar to estimate number of Dolly Varden returning to the Hulahula River in fall: 10,412 fish in 2005; 7,471 in 2006 (Osborne and Melegari 2008); 23,158 in 2007; and 12,340 in 2008 (U.S. Fish and Wildlife Service, unpublished data). Subsequently, a radio telemetry study in 2007–08 identified overwintering at four sites (U.S. Fish and Wildlife Service, unpublished data). The telemetry study also showed that a small fraction of the fish overwintering in the Hulahula River in one year overwintered the following year in nearby streams. Genetic studies distinguished the Hulahula River population from other stocks and provided a basis for understanding stock-specific ecology (Crane et al. 2005). Thus, we have some evidence that these fish can explore alternatives when faced with changing river conditions and perhaps have a means to track that.

Several harvest assessments have noted the importance of Dolly Varden from the Hulahula River to the Kaktovik subsistence fishery (Pedersen and Linn 2005). From October 2000 to September 2002, all fishing efforts by village residents in early winter occurred in the Hulahula River, with Dolly Varden being the only species captured (Pedersen and Linn 2005). Users reported that three sites in the river are traditionally used, but two sites were noted as the most productive for winter Dolly Varden fishing. No summer fishing took place in the Hulahula River.

Although the information about Dolly Varden from the Hulahula River is not complete, the existing data provide the most focused and comprehensive set of information available about this species on the North Slope. More information of this kind that could be used to evaluate the importance of seasonal meltwater and the effects that the loss of glacier may have on these fish would guide future management of this resource and provide a foundation for modeling climate change effects on migratory species in other aquatic systems. An integrated, multidisciplinary approach that links Dolly Varden ecology with concurrent assessments of glacier characteristics and stream attributes will be critical for assessing the sustainability of this resource for future users on the North Slope.

Potential Impacts on Shorebird Ecology

In 2010 we investigated shorebird and invertebrate abundance on three deltas, two of which were associated with rivers that received significant inputs of glacial meltwater (Jago and Hulahula Rivers) and one that has little glacier influence (Canning River). Our preliminary analyses suggest the differences between deltas fed by glacial versus nonglacial rivers may influence patterns of shorebird use. Glacially influenced deltas had siltier substrates, and the lagoons around the deltas were less salty during the period when shorebirds used them. Freshwater occurred along two-thirds of the waters' edge of glacial-influenced deltas, but much less freshwater occurred in the delta not glacially influenced. Characteristic conditions found on the glacial deltas and in adjacent waters were likely caused by inputs of silt, clay, and freshwater from melting glaciers in the Brooks Range. The delta with little glacial influence is a branch of the Canning River, which appears to have very little freshwater output during the late summer. Conditions here may illustrate what could happen on some glacially influenced deltas once late-summer meltwater from glaciers is no longer present.

Tens of thousands of shorebirds migrate to coastal habitats of the Arctic National Wildlife Refuge after breeding on the Arctic Coastal Plain of northern Alaska and Canada, and habitat differences apparently affect availability of shorebird food resources. The greatest concentrations of shorebirds are found on mudflats associated with river deltas, which provide important foraging habitat for post-breeding shorebirds (Taylor et al. 2010). Shorebirds likely depend on these delta mudflats for food resources to begin migration. For some species, food requirements are further increased during this period because of the molting of new flight feathers.

Two groups of freshwater invertebrates, Oligochaeta and Chironomidae, were more abundant in the silty habitats of glacial-fed deltas. The nonglacial delta had low invertebrate abundance, presumably due to the absence of freshwater invertebrates in the sandier and saltier habitats found there. Invertebrate abundance remained low on this delta until a storm surge deposited saltwater invertebrates (Amphipoda) on the mudflat. The life histories of freshwater versus saltwater invertebrates differ, with implications for shorebirds. For example, freshwater species spend multiple years in mudflat habitats (Butler 1982, Danks et al. 1994), while occurrence of saltwater invertebrates is unpredictable. Chironomid larvae spend at least three years in mudflats before pupating and turning into adults. Larvae that are present in mudflats for multiple years provide a more predictable and stationary food resource than those of species with a yearly life cycle or species that are mobile. For example, saltwater invertebrates like Amphipoda retreat from mudflats each winter (Evans 1976, Craig et al. 1984) when the mud freezes. In the summer amphipods are generally unavailable to foraging shorebirds until they are washed onto mudflats by storm surges and become stranded in puddles. Because storm surges are unpredictable events, we consider saltwater invertebrates to be a less dependable resource for shorebirds.

In 2010, we sampled triglyceride levels in semipalmated sandpipers (*Calidris pusilla*) early in the post-breeding season before the occurrence of any storm surges. Triglyceride levels provide a measure of fattening rates (Williams et al. 1999, Guglielmo et al. 2002). We found triglyceride levels were higher for birds feeding on the glacially influenced deltas compared to mudflats without glacial influence. We assume the difference was due to the low abundance of invertebrates on the delta without glacial influence. Soon after our sampling a storm surge occurred, coinciding with a pulse in shorebird migration. After the water levels dropped we observed thousands of shorebirds feeding on both saltwater and freshwater invertebrates on all three deltas. It appears that shorebirds utilize saltwater food resources opportunistically, but they rely on freshwater invertebrates as a more consistent resource.

There had been little previous research on the relationships between shorebirds, invertebrates, and delta mudflats along the coast of the Arctic Refuge. Our work suggests that glacially influenced deltas in particular provide an important resource for post-breeding shorebirds. Loss of Brooks Range glaciers will likely lead to decreases in sediment transport and freshwater inflow in rivers that are currently influenced by glaciers. These decreases in turn are likely to result in sandier delta mudflats and saltier lagoons, with potential implications for invertebrates and the birds that feed on them. Our observation that deltas with no glacial influence lack freshwater invertebrate species suggests that these species will disappear in currently glaciated watersheds as their mudflats become sandier and saltier with the loss of glacial meltwater and silt. Therefore, we hypothesize that loss of glaciers will have a negative impact on shorebirds as they prepare for migration, and we plan to investigate this further.

Potential Impacts on Marine Foodwebs

Glacier loss could impact estuarine ecosystems within the Arctic National Wildlife Refuge by altering the quantity, quality, and seasonality of river inputs. For example, differences between river inputs with and without significant glacier influence may be important in determining amounts and pathways of terrestrial carbon and nitrogen movement through coastal food webs. A comparison of stream and river water chemistry among catchments with different percentages of glacier coverage in southeastern Alaska demonstrated that concentrations of dissolved inorganic nitrogen are negatively correlated with glacier coverage whereas concentrations of soluble reactive phosphorus are positively correlated with glacier coverage (Hood and Berner 2009). Bioavailability of dissolved organic matter is also positively correlated with glacier coverage (Hood et al. 2009). If the correlations described above hold true for glacier-fed streams and rivers within the Arctic Refuge, then the glaciers may be a particularly important source of soluble reactive phosphorus and labile dissolved organic matter in mid to late summer that is not available in rivers without significant glacier inputs.

Although concentrations of dissolved inorganic nitrogen were negatively correlated with glacier coverage in the Hood and Berner (2009) study, it should be noted that high nitrate concentrations have been linked to glaciers and (or) proglacial features in some other studies (Apollonio 1973, Hood and Scott 2008). Thus, robust conclusions about the importance of glacier-fed streams and rivers as sources of nutrients and organic matter to coastal waters of northern Alaska will ultimately require focused studies in this region. We can expect glacier loss to be accompanied by a general decrease in mid to late summer export, including a decrease in relatively labile dissolved organic matter associated with microbial activity within and beneath glaciers (Hodson et al. 2005), but specific trajectories of individual waterborne constituents remain uncertain.

While we now recognize that terrestrial organic matter inputs to the Arctic Ocean are larger and more labile than previously thought, many questions remain about the influence of these inputs on food webs. Is terrestrial organic matter a major energy source supporting metazoan consumers, or is most of the energy from terrestrial organic matter lost during microbial processing? Does decomposition of terrestrial organic matter serve as a source of or a sink for inorganic nitrogen in coastal waters? Most of the labile river-supplied organic matter delivered to coastal waters probably enters the microbial food web. Yet, a recent study by Dunton et al. (2006) provides evidence of significant carbon and nitrogen from terrestrial organic matter making it into Arctic cod collected from lagoons along the northern Alaska coast. This finding suggests that either there is a strong link between the microbial and metazoan food webs or there is a direct pathway for terrestrial organic matter into the metazoan food web. In order to effectively predict how productivity in arctic coastal waters may be influenced by future climate change, we need to gain a better understanding of how terrestrial inputs contribute to coastal food webs under current conditions. Present contributions and future losses of glacier inputs may be particularly important within the Arctic National Wildlife Refuge.

Discussion

There are many uncertainties regarding climate change and its impact on Arctic landscapes and ecosystems, but we believe we have identified a straightforward and testable hypothesis linking these together. Glaciers here exist solely at the mercy of climate, unlike tidewater glaciers that have strong nonclimatic influences and major ice sheets that can influence their own climate; a 1–2ºC warming has caused them to enter a trajectory where they will likely disappear in the near future. Even if climate remains constant from this time forward, most glacier ice here will disappear because the late-summer snowline is higher than the elevation of most of the mountains. While direct effects of current climate change on fish and birds may be subtle and difficult to detect, the indirect effects on downstream ecosystems caused by the loss of glacial meltwater and silt may be enormous and predictable. Thus we hypothesize that loss of glaciers in the Arctic National Wildlife Refuge will exert strong influence on downstream ecosystems, affecting fish, birds, shrubs, and marine ecology. In this paper we have attempted to share what we know of these influences and predict future trajectories. We are just beginning to investigate relationships between climate, glaciers, and ecology in this region, and we welcome input from the broader scientific community as we pursue this work.

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Land Use and Salmon Habitat: A Comparison of North Pacific Watershed Parameters

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Abstract

Links between land use and freshwater habitat have been demonstrated in diverse studies, most focusing on one area, one time period, or comparisons among small drainages. This meta-analysis combined varied data sources to examine linkages between land use and the status of salmonid stocks on the North American coast from San Francisco Bay, California, to Bristol Bay, Alaska. To focus on land use only, the sample consisted of 61 midsized, coastal watersheds (approximately 500–60,000 km^2) and excluded basins where manmade dams blocked more than 20 percent of the drainage area. To quantify human influence on the landscape, the parameters percentage of forest cover, road density, percentage of total impervious area, human population density, and the composite Human Footprint Index (from Columbia University's Socioeconomic Data and Applications Center) were used as indicator metrics. To quantify salmon stock health, fisheries assessments of stock status for *Oncorhynchus* populations were evaluated and combined to create an index of salmonid status for each sample watershed. Linear regression showed that the human development metrics correlated. Comparison between the development parameters and the index of salmonid status showed a significant inverse relationship. Because the underlying data lack rigor and the systems are so complex, the relationship is suggestive rather than definitive. Developing better data for such parameters and monitoring them over time could provide useful information for science, fisheries management, and land use planning in the region.

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Evaluating Biodiversity Response to Forecasted Land Use Change: A Case Study in the South Platte River Basin, Colorado

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Abstract

Effects of future land use change on watersheds have important management implications. Seamless, national-scale land-use-change scenarios for developed land were acquired from the U.S. Environmental Protection Agency Integrated Climate and Land Use Scenarios (lCLUS) project and extracted to fit the South Platte River Basin, Colorado, relative to projections of housing density for the period 2000 through 2100. Habitat models developed from the Southwest Regional Gap Analysis Project were invoked to examine changes in wildlife habitat and biodiversity metrics using five ICLUS scenarios. The scenarios represent a U.S. Census base-case and four modifications that were consistent with the different assumptions underlying the A1, A2, B1, B2 Intergovernmental Panel on Climate Change global greenhouse gas emission storylines. Habitat models for terrestrial vertebrate species were used to derive metrics reflecting ecosystem services or biodiversity aspects valued by humans that could be quantified and mapped. Example metrics included richness of species of greatest conservation need, threatened and endangered species, harvestable species (e.g., upland game, big game), and total vertebrate species. Overall, the defined scenarios indicated that housing density and extent of developed lands will increase throughout the century with a resultant decrease in area for all species richness categories. The A2 Scenario in general showed greatest effect on area by species richness category. Areas with low or high species richness were projected to experience the greatest declines. The integration of the land use scenarios with biodiversity metrics derived from deductive habitat models may prove to be an important tool for decisionmakers involved in impact assessments and adaptive planning processes.

Keywords: deductive habitat models, wildlife habitat, biodiversity metrics, ecosystem services, land use scenarios, South Platte River Basin

Introduction

While many direct and indirect stressors can affect biodiversity, land use change is considered to be the most significant (Sala et al. 2000, Mattison and Norris 2005, Swetnam et al. 2010). Land use and land cover change are two processes that have consequences on a global scale and are driven by population trends and urban growth (Bierwagen et al. 2010). The United States population is projected to be between 402 and 616 million in 2090, an increase of 31–55 percent from 2000 (U.S. Environmental Protection Agency 2010).

The U.S. Environmental Protection Agency (EPA) has investigated the future impacts of population growth and urban development in depth. The Integrated

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Climate and Land Use Scenarios (ICLUS) dataset was created by the EPA to address the potential scenarios of population growth and housing development from 2000 to 2100 (U.S. Environmental Protection Agency, Bierwagen et. al 2010).

The U.S. Geological Survey Gap Analysis Program (GAP) has developed datasets for biodiversity conservation purposes in the continental United States (Prior-Magee et al. 2007). The GAP process provides landscape-level assessment for the conservation of biological diversity. GAP maps the distribution of plant communities and predicts the distribution of suitable habitat for terrestrial vertebrate species and compares these distributions with land stewardship to identify biotic elements at potential risk of endangerment. The baseline datasets GAP provides are uniquely suited for use with biodiversity assessments at broad multiple scales. The Southwest Regional Gap Analysis Project (SWReGAP) provides these datasets for the American Southwest states of Arizona, Colorado, Nevada, New Mexico, and Utah (Prior-Magee et al. 2007).

Evaluating the effect of urban encroachment and development on biodiversity is becoming increasingly important. Synthesis and analysis of future land use scenarios using datasets such as the ICLUS and SWReGAP habitat models are valuable to science and the future of conserving biodiversity, especially for informing land managers and decisionmakers about potential consequences and benefits of environmental management choices.

Study Area

The South Platte River Basin ranges from the plains of western Nebraska, eastern Colorado, and Wyoming to the mountains of the Front Range in Colorado (Figure 1). Within the South Platte River Basin are many rapidly growing cities, such as Denver and Fort Collins, Colorado, each with increasing pressures on terrestrial and aquatic environments caused by land use change and water development. This area has a projected population growth exceeding 50 percent by 2050 (U.S. Census Bureau 2005), suggesting continued growth and land use change in the future.

Overall, the South Platte River Basin spans 62,580 km^2 with vegetation ranging from grasslands in the plains to mixed conifer forests in the mountains. The study area comprised the portion of the basin within Colorado due to the availability of spatial data and habitat models from SWReGAP (Figure 1). Of the 49,030 km^2 within Colorado, approximately 28 percent is classified as agriculture, 22 percent as Rocky Mountain Ponderosa Pine, and 7 percent as Western Great Plains Sandhill Shrubland (Prior-Magee et al. 2007).

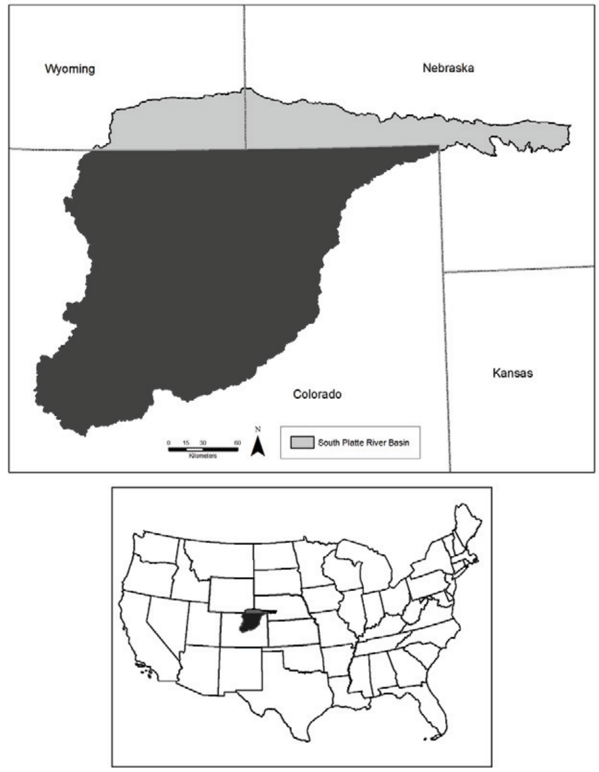


Figure 1. Location and extent of the study area (black) within the South Platte River Basin (black and grey).

Methods

The EPA-ICLUS (Version 1.3.1) dataset was used to assess habitat change and effects on biodiversity metrics. These seamless, national-scale land-use-change scenarios for developed land were acquired from EPA's Office of Research and Development (U.S. Environmental Protection Agency 2010). The data were extracted from the national coverages for the South Platte River Basin. This dataset allowed for analysis of projections of housing density for the period 2000 through 2100 for the five ICLUS scenarios, including a U.S. Census baseline and four modifications consistent with the different assumptions underlying the A1, A2, B1, and B2 Intergovernmental Panel on Climate Change (IPCC) global greenhouse gas emission storylines (Table 1; Bierwagen et al. 2010). The five ICLUS datasets were reclassified to identify urban (1) or nonurban areas (0).

For this analysis we characterized 4 biodiversity metrics of 17 available (Table 2). These were total vertebrate species richness (maximum=239), state designated Species of Greatest Conservation Need (SGCN) richness (maximum=98), federally Threatened and Endangered (T&E) species richness (maximum=12), and all harvestable species (e.g., upland game, big game) richness (maximum=50). The remaining 13 biodiversity metrics will be examined in subsequent study.

Table 1. EPA land-use-change scenarios for the conterminous United States (Bierwagen et al. 2010).

Scenario	Description
Baseline condition (BC)	Represents a level of medium fertility rates, medium domestic migration, and medium international migration.
A1	Represents fast economic growth, low population growth, and high global integration. Fertility is low with high domestic and international migration.
B1	Represents a globally integrated world but with more emphasis on environmentally sustainable economic development. Fertility and domestic migration are low while international migration is high.
A2	Represents continued economic development, with more regional focus and slower economic convergence between regions. Fertility and domestic migration are high and international migration is medium.
B2	Represents a regionally-oriented world of moderate population growth and local solutions to environmental and economic issues. Fertility rates are medium with low domestic migration and medium international migration.

The four biodiversity metrics were derived from 817 terrestrial vertebrate habitat models developed from SWReGAP (Boykin et al. 2007, 2010). We categorized each metric into four equal intervals of species richness (Appendix A).

Using ESRI ArcGIS 10, the current (year 2000) condition was characterized for the four biodiversity metrics. Current condition provides a baseline comparison for subsequent scenarios. The areas of each species richness category for each biodiversity metric were then quantified for nonurban land cover using the five ICLUS future development scenarios. ICLUS classified areas with housing density greater than 0.8 hectares per housing unit as nonurban (EPA 2010). The change in square kilometers and relative change of land classified as urban and nonurban were calculated and compared among the 5 future development scenarios for the year 2100. This change analysis allowed for examination of biodiversity metrics under each scenario.

Table 2. List of 17 available biodiversity metrics for species richness derived from the Southwest Regional Gap Analysis Project (Prior-Magee et al. 2007). Metrics in italics were used in the present study.

Biodiversity metrics
All vertebrate species
Reptiles
Amphibians
Birds
Mammals
Threatened and Endangered species
All Species of Greatest Conservation Need
Reptile Species of Greatest Conservation Need
Amphibian Species of Greatest Conservation Need
Bird Species of Greatest Conservation Need
Mammal Species of Greatest Conservation Need
All harvestable species
Harvestable upland game species
Harvestable big game species
Harvestable furbearer species
Harvestable waterfowl species
Bat Species of Greatest Conservation Need

Results

Ninety-seven percent of the South Platte River Basin study area was classified as nonurban using the Baseline 2000 Scenario (Table 3). This extent decreased by 2100 in all scenarios to 92–94 percent, with scenario A2 decreasing the greatest. The majority of area classified as nonurban within the study area was associated with species richness categories 2 and 3 (Figure 2).

Table 3. Nonurban and urban area (km^2) and percent (%) of total for baseline 2000 and five future land-use-change scenarios (A1, A2, B1, B2, and BC; see text)

	Nonurban		Urban	
	(km2)	(%)	(km2)	(%)
Baseline 2000	47,425	97	1,523	3
A1 2100	45,708	93	3,240	7
A2 2100	44,873	92	4,074	8
B1 2100	46,219	94	2,729	6
B2 2100	45,852	94	3,095	6
Baseline 2100	45,867	94	3,081	6

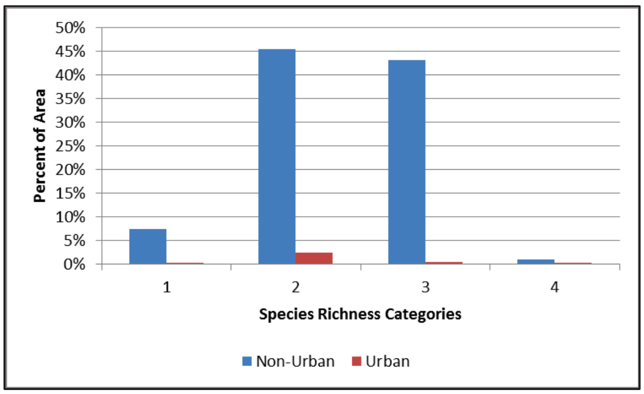


Figure 2. Percent of urban or nonurban area within the South Platte study area by total vertebrate species richness category for the baseline 2000 reference condition. Categories range from 1 (lowest richness) to 4 (highest richness; Appendix A).

A reduction in area based on predicted suitable habitat was identified for all species richness categories for all biodiversity metrics in all scenarios (Figures 3A–D). The decrease in area ranged from 1 to 12 percent. The A2 Scenario resulted in the greatest decreases with 7 of the category-metric comparisons resulting in a 7 percent or greater change. The A2 Scenario represents continued economic growth, high population growth and high domestic and medium international migration.

Species richness categories 1 and 4 had the highest decreases in total species richness, SGCN richness and T&E richness (Figure 3A–D). Thus, the analysis suggests that the most species rich (category 4) and species poor (category 1) areas will be detrimentally affected by urbanization. Each species categories contained equal number of species; however the amount of area for these categories was quite different (Figure 2). Categories 1 and 4 had a very small extent in 2000 (Figure 2), and thus small changes in area lost can have large effects on the percentage of area lost for these categories.

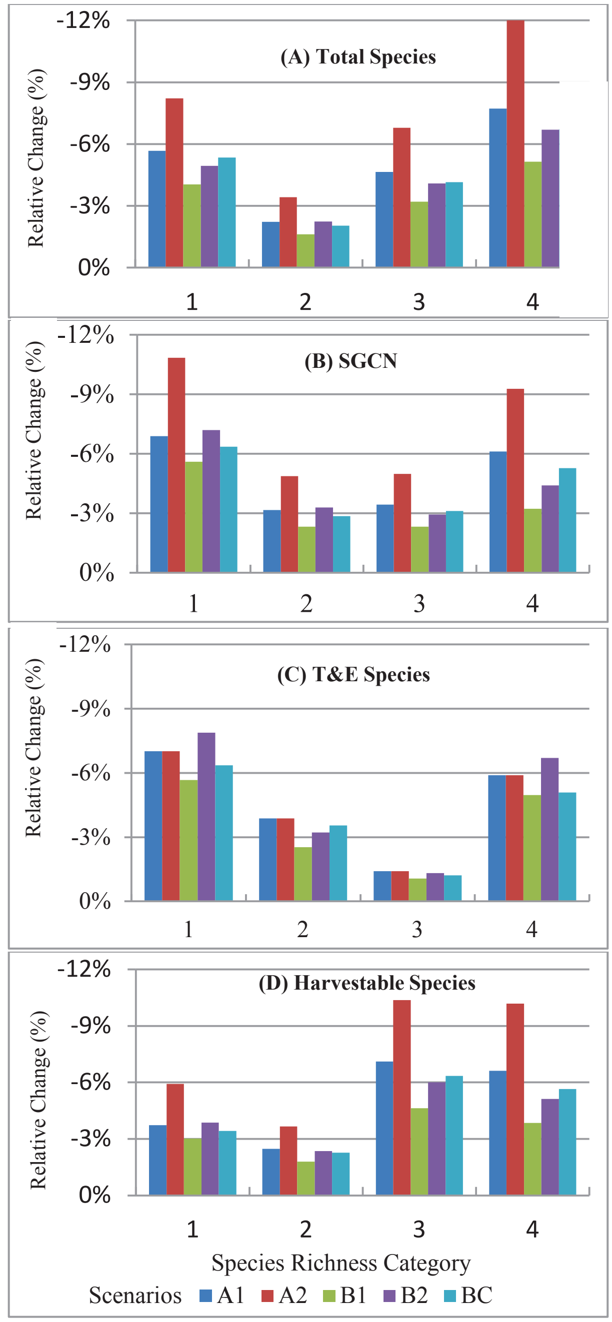
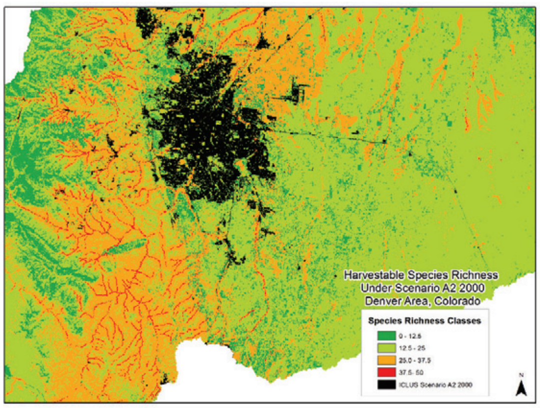


Figure 3. Relative change (%) in extent of nonurban land cover to 2100 across five future land use change scenarios within four species richness categories for (A) total vertebrate species (B) Species of Greatest Conservation Need (SGCN), (C) Threatened and Endangered species, and (D) harvestable species. Future change scenarios were A1, A2, B1, B2, and BC (see text). The four species richness categories range from 1 (lowest richness) to 4 (highest richness; Appendix A).

Harvestable species showed a different pattern for relative change by species richness categories under the 5 scenarios in comparison to the other three species richness metrics. Specifically, richness categories 3 and 4 showed the greatest extents of declines rather than categories 1 and 4 (Figure 3D). This resulted from categories 3 and 4 being well represented in areas of projected urban growth. The spatial pattern of harvestable species (Figure 4) identifies an abundance of categories 3 and 4 occurring near Denver and the cities of the Front Range. Urban growth also affects a large portion of Category 2 south of Denver; however, this is a smaller percentage of this category.

A



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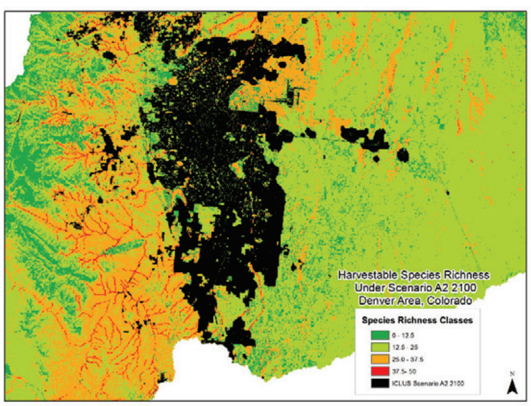

Figure 4. Distribution of harvestable species richness categories for current urban extent from Baseline 2000 (A) and future urban extent from A2 Scenario 2100 (B) surrounding Denver, Colorado within the South Platte River Basin study area.

Conclusions

The analysis indicated declines in nonurban extent over the next century. This change is projected to result in decreases in extent of area for all species richness categories for the four metrics examined: total vertebrate species, Species of Greatest Conservation Need, Threatened and Endangered species, and harvestable species. Among the five climate change scenarios, Scenario A2 presents the greatest increase in urban growth both in percent change and total area. Areas with low or high species richness are projected generally to experience the greatest declines. Areas with suitable habitat for high numbers of harvestable species will be affected by this urban growth.

Our purpose was to integrate available land use scenarios with deductive habitat models to provide an important tool for decisionmakers involved in impact assessments and adaptive planning processes across a variety of environmental management sectors. This initial analysis will be followed by future work on the remaining 13 biodiversity metrics (Table 2) and in different geographies to test transferability of the process.

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Appendix A

Extent and change in land cover types from the baseline scenario in 2000 for five climate change scenarios, four biodiversity metrics, and four categories of species richness (1, low; 4, high). Biodiversity metrics were species richness for total vertebrate species, Species of Greatest Conservation Need (SGCN), Threatened and Endangered (T&E) species, and harvestable species. For nonurban and urban "% of total" refers to percent of total land cover in study area. Relative change (%) refers to area of nonurban land cover in scenario relative to area of nonurban land cover in Baseline Scenario 2000.

	Total vertebrate species				SGCN species				T&E species				Harvestable species			
Species richness category (number of species)	1 (4–59)	2 (60–119)	3 (120–179)	4 (180–239)	1 (1–24)	2 (25–49)	3 (50–74)	4 (75–98)	1 (0–3)	2 (4–6)	3 (7–9)	4 (10–12)	1 (0–12)	2 (13–24)	3 (25–37)	4 (38–50)
Baseline Scenario 2000																
Nonurban (km^2)	3,590	22,229	21,105	501	3,010	12,214	31,555	646	6,276	27,959	13,090	100	5,800	31,405	9,694	526
Urban (km^2)	96	1,184	228	15	351	1,011	154	7	1,283	174	64	2	373	1,030	112	7
Nonurban (% of total)	7%	45%	43%	1%	6%	25%	64%	1%	13%	57%	27%	0%	12%	64%	20%	1%
Urban (% of total)	0%	2%	0%	0%	1%	2%	0%	0%	3%	0%	0%	0%	1%	2%	0%	0%
A1 Scenario 2100																
Nonurban (km^2)	3,387	21,734	20,125	462	2,802	11,829	30,471	607	5,836	26,874	12,904	94	5,584	30,629	9,005	492
Urban (km^2)	302	1,680	1,205	53	560	1,397	1,236	46	1,725	1,257	251	7	590	1,807	801	42
Nonurban (% of total)	7%	44%	41%	1%	6%	24%	62%	1%	12%	55%	26%	0%	11%	63%	18%	1%
Urban (% of total)	1%	3%	2%	0%	1%	3%	3%	0%	4%	3%	1%	0%	1%	4%	2%	0%
Relative change (%)	–6%	–2%	–5%	–8%	–7%	–3%	–3%	–6%	–7%	–4%	–1%	–6%	–4%	–2%	–7%	–7%
A2 Scenario 2100																
Nonurban (km^2)	3,295	21,468	19,670	440	2,683	11,620	29,985	586	5,836	26,874	12,904	94	5,457	30,256	8,688	473
Urban (km^2)	393	1,946	1,660	75	679	1,606	1,723	67	1,725	1,257	251	7	717	2,180	1,117	60
Nonurban (% of total)	7%	44%	40%	1%	5%	24%	61%	1%	12%	55%	26%	0%	11%	62%	18%	1%
Urban (% of total)	1%	4%	3%	0%	1%	3%	4%	0%	4%	3%	1%	0%	1%	4%	2%	0%
Relative change (%)	–8%	–3%	–7%	–12%	–11%	–5%	–5%	–9%	–7%	–4%	–1%	–6%	–6%	–4%	–10%	–10%
B1 Scenario 2100																
Nonurban (km^2)	3,445	21,869	20,430	475	2,841	11,930	30,822	625	5,921	27,253	12,951	95	5,625	30,842	9,245	506
Urban(km^2)	243	1,545	901	40	522	1,295	885	27	1,641	879	204	7	548	1,593	561	27
Nonurban (% of total)	7%	45%	42%	1%	6%	24%	63%	1%	12%	56%	26%	0%	11%	63%	19%	1%
Urban (% of total)	0%	3%	2%	0%	1%	3%	2%	0%	3%	2%	0%	0%	1%	3%	1%	0%
Relative change (%)	–4%	–2%	–3%	–5%	–6%	–2%	–2%	–3%	–6%	–3%	–1%	–5%	–3%	–2%	–5%	–4%
B2 Scenario 2100																
Nonurban (km^2)	3,413	21,730	20,242	467	2,793	11,812	30,630	618	5,782	27,060	12,917	93	5,576	30,666	9,112	499
Urban (km^2)	275	1,684	1,088	48	570	1,413	1,078	35	1,779	1,071	237	8	598	1,770	694	34
Nonurban (% of total)	7%	44%	41%	1%	6%	24%	63%	1%	12%	55%	26%	0%	11%	63%	19%	1%
Urban (% of total)	1%	3%	2%	0%	1%	3%	2%	0%	4%	2%	0%	0%	1%	4%	1%	0%
Relative change (%)	–5%	–2%	–4%	–7%	–7%	–3%	–3%	–4%	–8%	–3%	–1%	–7%	–4%	–2%	–6%	–5%
Baseline Scenario 2100																
Nonurban (km^2)	3,398	21,774	20,228	467	2,818	11,866	30,571	612	5,877	26,966	12,930	95	5,602	30,690	9,079	497
Urban (km^2)	290	1,640	1,103	48	544	1,359	1,136	41	1,684	1,166	225	7	572	1,746	727	37
Nonurban (% of total)	7%	44%	41%	1%	6%	24%	62%	1%	12%	55%	26%	0%	11%	63%	19%	1%
Urban (% of total)	1%	3%	2%	0%	1%	3%	2%	0%	3%	2%	0%	0%	1%	4%	1%	0%
Relative change (%)	–5%	–2%	–4%	–7%	–6%	–3%	–3%	–5%	–6%	–4%	–1%	–5%	–3%	–2%	–6%	–6%

Modeling Impacts of Environmental Change on Ecosystem Services across the Conterminous United States

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Abstract

Climate model projections suggest that there will be considerable increases in temperature and variability in precipitation across the conterminous United States during the next 100 years. These changes in climate coupled with changes in land use and increases in human population will likely have a significant effect on water resources, carbon fluxes, biodiversity, and the services they provide. As society reacts to changing environmental conditions, the adaptation and mitigation strategies for one ecosystem service could come at the expense of another. It is critical that planning tools be developed to evaluate these tradeoffs between ecosystem services so that sound management decisions may be made in the face of climate, economic, and demographic change. This paper presents the Water Supply Stress Index–Carbon & Biodiversity model (WaSSI-CB) and demonstrates its potential for predicting changes in water supply and demand, carbon dynamics, and potential biodiversity under multiple stresses. The core of WaSSI-CB is a water balance model (WaSSI) that is sensitive to land cover and climate and operates on a monthly time step at the 8-digit hydrologic unit code (HUC) watershed scale across the conterminous United States. Annual U.S. Geological Survey water demand estimates are adjusted for population, disaggregated to the monthly scale, and compared to groundwater and surface water supply to assess water supply stress. Gross ecosystem productivity, ecosystem respiration, and net ecosystem carbon exchange are estimated using actual evapotranspiration. Similarly, potential biodiversity of reptiles, birds, amphibians, mammals, vertebrates, and tree distribution and abundance are estimated as a function of evapotranspiration. We show how the model may be used to predict the effects of climate, population, and land cover change on water resources and carbon fluxes in the next 50 years using downscaled monthly future scenarios, population projections, and hypothetical changes in land cover. Finally, the paper explores tradeoffs among management strategies for these ecosystem services.

Keywords: water supply, water demand, carbon sequestration, biodiversity, climate change

Introduction

Increasing water use in the United States has led to widespread hydrologic manipulation and consumptive off-stream water use, practices that alter river flows (Vörösmarty et al. 2004), threaten the sustainability of the resource (Alcamo et al. 2003), and degrade ecosystem function (Carlisle et al. 2010). Future changes in climate will place additional pressure on freshwater supplies (Bates et al. 2008). The effect of these stressors will be highly variable over both time and space, making it difficult to assess effects on water resources into the future.

Like water supply, carbon sequestration and biodiversity are valuable ecosystem services that are vulnerable to the effects of climate change and human activities (Nemani et al. 2003, Beer et al. 2010). Carbon sequestration, or net ecosystem exchange (NEE), is the difference between ecosystem respiration (Re) from autotrophs and heterotrophs and gross ecosystem productivity (GEP), or photosynthetic assimilation of carbon by foliage. When NEE for an ecosystem is negative, the ecosystem is a net carbon sink. When NEE is positive, the ecosystem is a net source of carbon. Ecosystem water use, or evapotranspiration (ET), is tightly coupled with ecosystem productivity (Law et al. 2002, Sun et al. 2011 a) and biodiversity (Currie and Paquin 1987, Currie 1991). As a result, NEE and biodiversity can be predicted based on ET, and the factors that affect ET

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(e.g. climate change, land use change) will also have an effect on NEE and biodiversity. Managing an ecosystem to enhance NEE or biodiversity will result in reduced residual water supply for human use because NEE and biodiversity increase with increasing ET.

Management tools are needed that can evaluate the tradeoffs between these ecosystem services at multiple spatial and temporal scales in the United States. Unfortunately, there are few integrated models of water supply and demand, carbon dynamics, and biodiversity with which to evaluate the effect of climate, land cover, and population change or the tradeoffs between management strategies for these ecosystem services. The U.S. Department of Agriculture Forest Service has developed the Water Supply Stress Index–Carbon & Biodiversity model (WaSSI-CB) that is intended to fill this need. The model can be used to project the effects of global change on water supply stress, carbon sequestration, and potential biodiversity across the conterminous United States at the 8-digit hydrologic unit code (HUC) watershed scale (Sun et al. 2008, Sun et al. 2011 a). In this paper, we apply the WaSSI-CB model to project the effects of population, land cover, and climate change on water supply, carbon sequestration, and potential biodiversity, and we explore tradeoffs among management strategies for these ecosystem services.

Methods

The core of WaSSI-CB is a monthly water balance model (WaSSI) that is sensitive to land cover and climate, computing the water balance for each of eight land cover classes independently in the approximately 2,100 8-digit HUC watershed scale across the conterminous United States. Evapotranspiration (ET), infiltration, soil storage, snow accumulation and melt, surface runoff, and baseflow processes are accounted for within each basin based on spatially explicit 2001 MODIS land cover (Figure 1), and discharge (Q) is conservatively routed through the stream network from upstream to downstream watersheds. ET is estimated with an empirical equation based on multisite eddy covariance ET measurements using MODIS derived monthly leaf area index (LAI), potential ET (PET_{hamon}), and precipitation (PPT) as independent variables (Sun et al. 2011 a, b). Estimation of infiltration, soil storage, and runoff are accomplished through the integration of algorithms from the Sacramento Soil Moisture Accounting Model and STATSGO-based soil parameters (Koren et al. 2003).

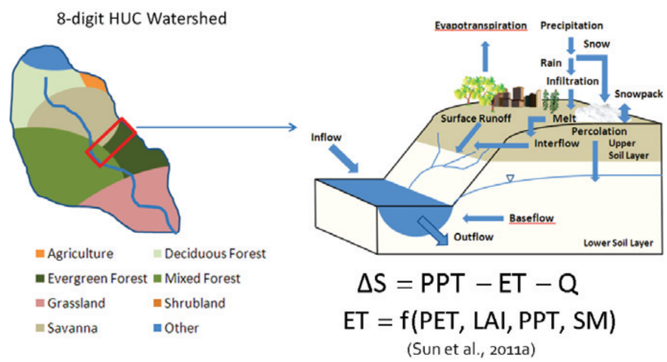

Figure 1. Schematic of the hydrologic processes simulated by the WaSSI-CB model.

Ecosystem GEP, Re, and NEE are estimated using actual evapotranspiration (AET) and water use efficiency parameters (Table 1) that were derived from measured site-level water and carbon fluxes for a variety of land cover types monitored by the FLUXNET (Sun et al. 2011 a).

Similarly, potential biodiversity of reptiles, birds, amphibians, mammals, vertebrates, and tree species richness are estimated as a function of PET and AET (Table 2; Currie and Paquin 1987, Currie et al. 1991).

While WaSSI-CB was designed to make projections regarding the potential diversity of multiple groups of biota, this paper focuses on tree species richness. The effects of development, habitat fragmentation, and forest management were neglected to simplify this hypothetical study, and HUC watersheds where total forest cover (sum of deciduous, evergreen, mixed forest, and savanna) was less than 10 percent of the total watershed area were excluded.

Table 1. Model parameters for estimating GEP as a function of AET, GEP = a*AET [g C/m^2/mo] and Re as a function of GEP, Re = m + n*GEP [g C/m^2/mo], after Sun et al. (2011 a).

Land cover class	a	m	n
Crop	3.13	40.6	0.43
Deciduous	3.2	30.8	0.45
Evergreen	2.46	9.9	0.69
Mixed forest	2.74	24.4	0.62
Grassland	2.12	18.9	0.64
Shrubland	1.35	9.7	0.56
Savanna	1.26	25.2	0.53
Water/urban/barren	1.53	9.7	0.56

Table 2. Model parameters for estimating potential biodiversity as a function of annual PET or AET, after Currie and Paquin (1987) and Currie et al. (1991).

Group	Model
Birds	1.40+0.00159*PET (PET<525 mm)
	2.26–0.0000256*PET (PET≥525 mm)
Mammals	1.12[1.0–exp(–0.00348*PET)]+0.653
Amphibians	0 (PET<200 mm)
	3.07[1.0–exp(–0.00315*PET)]
Reptiles	0 (PET<400 mm)
	5.21[1.0–exp(–0.00249*PET)]–3.347
Vertebrates	1.49[1.0–exp(–0.00186*PET)]+0.746
Trees	185.8/[1.0+exp(3.09–0.00432*AET)]

County-level 2005 annual U.S. Geological Survey (USGS) water demand and groundwater withdrawal estimates by sector (Kenny et al. 2009) were rescaled to the 8-digit HUC watershed scale, adjusted for population, and disaggregated to the monthly scale using regional regression relationships. Return flows by sector were computed using return flow percentages from the 1995 USGS report (Solley et al. 1998). The total water supply in each HUC watershed is the sum of surface water supply at the watershed outlet predicted by WaSSI-CB, total groundwater withdrawals, and the total return flow. Total water demand is the sum of the water use by all sectors in each watershed. The water supply stress index (WaSSI) is computed as the ratio of water demand to water supply (Sun et al. 2008). The WaSSI-CB model currently does not account for water storage in reservoirs or anthropogenic water diversion projects such as interbasin transfers and assumes that all surface water is available for human use.

Intergovernmental Panel on Climate Change (IPCC) AR4 scenarios A1B and B2 were assessed using downscaled CSIRO-Mk2.0, CSIRO-Mk3.5, HADCM3, and MIROC3.2 global circulation models for future scenarios according to the 2010 U.S. Forest Service Resources Planning Act Assessment to account for changes in population (Zarnoch et al. 2010) and climate (Coulson et al. 2007). WaSSI-CB results for all future climate scenarios were averaged to represent the mean (ensemble) response to climate change among these scenarios. Water use for the domestic sector was assumed to vary with watershed population projections according to an empirical per capita water use function. Water use for all other sectors was held constant at the 2005 level. Groundwater withdrawal rates from all sectors were also held constant at the 2005 level.

Results

Water Supply Stress

Total surface water supply for the conterminous United States was predicted to decrease as a result of climate change over the next 60 years from approximately 2.0 trillion m^3/yr in 2000 to 1.6 trillion m^3/yr in 2060 (Figure 2), due in large part to the effects of increasing temperature on ET, but also to decreasing PPT in some parts of the country.

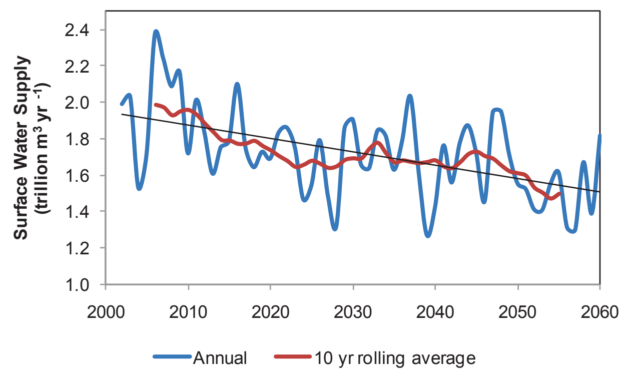

Figure 2. Predicted total U.S. surface water supply through 2060.

Changes in surface water supply will vary considerably across space (Figure 3), with the largest decreases in parts of the Great Plains region and the largest increases in the Southwest. These extreme changes in surface water supply may be misleading because surface water supplies are naturally low in these arid and semiarid environments. As a result, small absolute changes in supply can lead to large percentage changes. Much of the Great Plains region depends on declining groundwater supplies, so despite the lack of dependence on surface water, the Great Plains will likely continue to experience decreases in total water supply in the early part of the 21st century. The large percentage increases in surface water supply in parts of the Southwest are not significant in terms of absolute water supply, so these increases will have minimal effect on water supply in this region.

The WaSSI-CB model predicted that the total water demand in the United States will increase by 6 percent from 2001 to 2060 due to increasing population, with the largest increases in expanding metropolitan areas. The combined effect of decreasing water supply and increasing water demand resulted in increases in the water supply stress index (WaSSI) in most HUC watersheds. A long-term WaSSI value of 0.4 is commonly used as a threshold to identify watersheds experiencing some level of water supply stress (e.g., Alcamo 2000). Using this threshold, the Southwest and southern Great Plains regions were projected to experience water stress in 2051–2060 (Figure 4). Metropolitan areas of the east (e.g., Charlotte, NC; Atlanta, GA; South FL) were also projected to experience water stress.

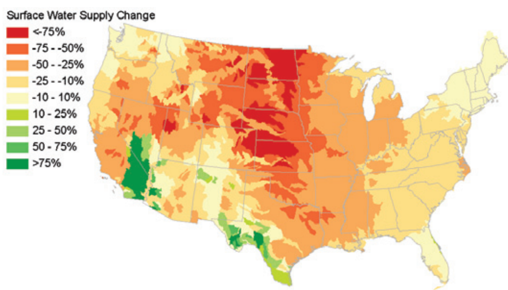

Figure 3. Change in mean annual surface water supply: 2051–2060 vs. 2001–2010.

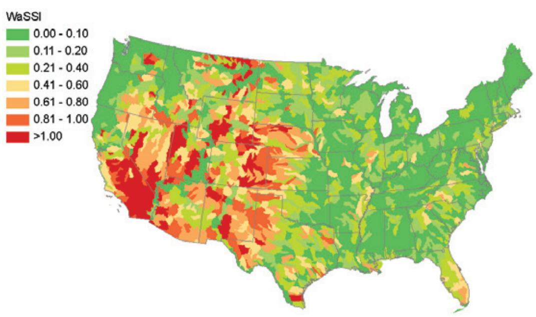

Figure 4. Mean annual Water Supply Stress Index (WaSSI) for 2051–2060.

Carbon Sequestration

Annual WaSSI-CB modeled NEE varied from a carbon source of 145g C/m^2/yr to a strong carbon sink of −1117 g C/m^2/yr (Figure 5). Carbon sequestration was highest in the Southeast, where abundant water and energy were available to drive ET and ecosystem productivity, and lowest in the West (excluding the Pacific Coast), where water was a major limitation. The total net annual carbon sequestration in the United States was 2.68 Pg C/yr during 2001–2010.

Carbon sequestration potential was largely projected to increase (NEE was more negative) across New England, the Upper Midwest, and Pacific Northwest and decrease across most of the Great Plains and Southwest regions (Figure 6) as a result of climate change. Areas that were carbon sources (NEE was positive) either in 2001–2010 or 2051–2060 are shown in gray. Ecosystem NEE is driven by AET, thus carbon sequestration potential will increase in areas with increasing AET and decrease in areas with decreasing AET. Regions where AET is historically energy-limited (i.e., high latitudes) were projected to have the largest increases in NEE as a result of increases in temperature. Regions where AET is historically water-limited (e.g., the Great Plains and Southwest) were projected to experience decreases in NEE due primarily to increases in temperature, but also to decreases in PPT in some areas. The predicted total net annual carbon sequestration in the United States was 2.81 Pg C/yr during 2051–2060, an increase of 4.9 percent from 2001–2010.

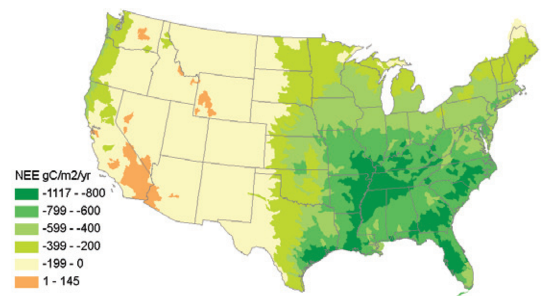

Figure 5. 2001–2010 mean annual net ecosystem carbon exchange (g C/m^2/yr).

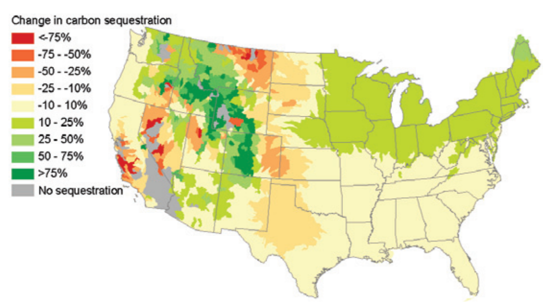

Figure 6. Change in mean annual carbon sequestration: 2051–2060 vs. 2001–2010.

Potential Tree Species Richness

Predicted potential tree species richness, or the number of tree species per unit area, assumes equilibrium conditions. The highest potential tree species richness was predicted for the Southeast, followed by the northern Pacific coast (Figure 7). These trends followed the spatial pattern of predicted AET across the United States. The Southeast, with abundant water and energy, had the highest AET rates and tree species richness. AET and tree species richness were water-limited in the Southwest and energy-limited in the Northeast, upper Midwest, and Pacific Northwest.

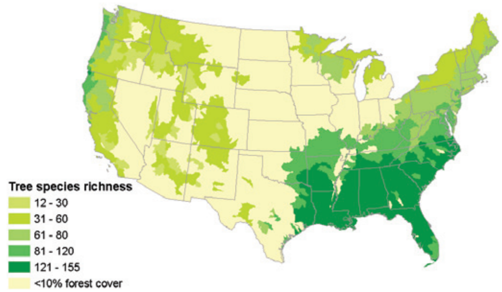

Figure 7. Historic tree species richness.

Tradeoffs Between Water and Carbon

Water yield and carbon sequestration are important societal services forested ecosystems provide. Unfortunately, managing forest resources to maximize one ecosystem service comes with a penalty in the other. To illustrate the tradeoffs between water and carbon, we developed a hypothetical scenario in which 20 percent of all forest land cover in the conterminous United States was converted to shrubland. This scenario may be a potential management option if increasing water supply were a top priority.

Water supply under this scenario had modest increases (up to 15 percent) in HUC watersheds dominated by forest land cover, particularly where the watersheds are in a “headwater” landscape position receiving minimal flow from upstream watersheds (Figure 8). This is partly because many of the “headwater” watersheds are dominated by forest cover, but also because the effects of this management strategy diminish in downstream watersheds as surface water supply was affected by nonforest land covers.

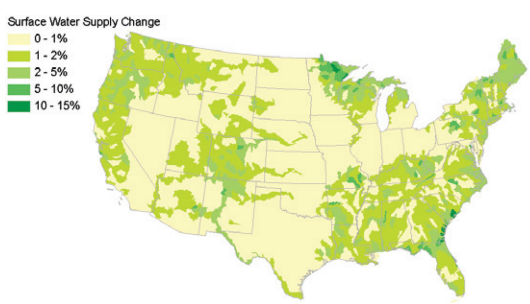

Figure 8. Change in 2001–2010 mean annual surface water supply due to a 20 percent forest conversion to shrubland.

While reducing forest cover by 20 percent increased water supply in some watersheds, this management option led to decreases in carbon sequestration potential over much of the East, Rocky Mountains, and Pacific Northwest (Figure 9) primarily because forest was the dominant land cover in these watersheds. The total net annual carbon sequestration in the United States under this scenario was 2.57 Pg C/yr during 2051–2060, a decrease of 4.1 percent from the 2001–2010 baseline case.

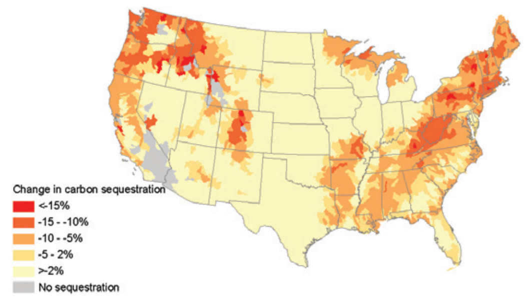

Figure 9. Change in 2001–2010 mean annual carbon sequestration due to a 20 percent forest conversion to shrubland.

On a regional basis, decreases in carbon sequestration (1–9 percent) as a result of this management action were greater than increases in surface water supply (0.4–1.6 percent) (Figure 10). The greatest effect occurred in regions with substantial forest cover and high AET (Northeast, Southeast, Northwest), and the least effect occurred in regions with minimal forest cover and (or) low AET (Midwest, Great Plains, and Southwest).

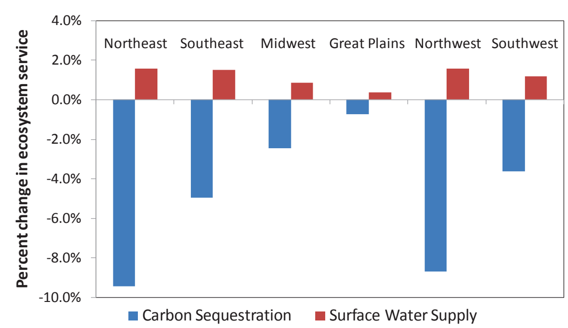

Figure 10. Change regional in 2001–2010 total annual surface water supply and total annual carbon sequestration due to a 20 percent forest conversion to shrubland.

Conclusions

In this paper, we showed how the WaSSI-CB model may be used to predict biodiversity and the effects of climate, population, and land cover change on water resources and carbon fluxes in the next 50 years, and we explored tradeoffs between water and carbon for a hypothetical management scenario where 20 percent of forest cover was converted to shrubland. Model projections indicated that surface water supply will decrease in much of the conterminous United States by 2060, and with water demand likely to increase as a result of population growth, water supply stress was projected to increase. Carbon sequestration potential was largely projected to increase across New England, the Upper Midwest, and Pacific Northwest, and decrease across most of the Great Plains and Southwest regions. Converting 20 percent of forest cover to shrubland led to modest increases in surface water supply and larger decreases in carbon sequestration as one might expect, but the change in water supply and carbon sequestration was highly sensitive to location and dominant land cover type.

The WaSSI-CB model is a work in progress, and several areas are currently under development: (1) reservoir storage; (2) interbasin transfer; (3) limitations on water withdrawal due to aquatic ecosystem needs; and (4) the effect of both climate change and land use change on water quality.

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The Urban Fishery: An Application of System Robustness

Meagan B. Krupa

Abstract

The conceptual framework of robustness applied to a case study of a common pool resource—the Lower Ship Creek Fishery in Anchorage, AK. I apply the robustness framework rather than resilience theory to address the management of this fishery because engineered systems, such as a hatchery fishery, operate independently of some ecological variables within the system. There is a need to distinguish between the socioeconomic and ecological components of the system and use interdisciplinary methods to study their interrelationships because of the unintended effects of engineered components that are relatively insensitive to ecological feedbacks. For example, engineered hatchery fish continue to thrive despite declining stream conditions. I explore the interrelationship of socioeconomic and ecological systems and then use Ostrom's design principles to define, assess, and suggest opportunities for increasing the robustness of an urban fishery.

Keywords: common pool resource, salmon, fishery, hatchery, robustness, management

Introduction

Every summer residents and visitors gather for a unique experience on a creek in downtown Anchorage, AK. Surrounded by industrial yards, the State's railroad, interlocking road systems, and the city's port, pulses of salmon carried by Cook Inlet's world record tides enter Ship Creek. The anglers enter the creek by descending down specially designed staircases built to withstand tides and ice flow. Standing shoulder to shoulder on the banks of Ship Creek, anglers fish for salmon. Undercover patrols move up and down the creek to ensure safety and regulatory compliance. Anchorage's children line up on a child-sized fishing platform to cast their first line into salmon-filled waters. Benches, garbage cans, fish cleaning stations, and restrooms are conveniently located and regularly maintained.

To visitors from cities like Seattle and Baltimore who take buses to see the salmon fishery, Ship Creek appears to have defied the odds. Here, in The Last Frontier, the salmon have seemingly prevailed over the effects of urbanization. Or have they? What many visitors and residents do not realize is that the two salmon species are the product of a carefully engineered hatchery fishery, fueled by a complex network of inter- and intra-institutional arrangements and cost structures.

The above description is what a robust Lower Ship Creek Fishery might look like if it was supported by the appropriate social and economic frameworks. Today's fishery has all the people and fish with few of the amenities needed to sustain them. Declining water quality and quantity, erosion, and barriers to fish passage have substantially altered the creek. Lower Ship Creek is a semiengineered system sitting at the crossroads between wilderness and concrete. In light of this position, the question that managers face is how to create a robust urban fishery when some of the components are engineered and others are natural. Managers need to be able to identify the characteristics that decrease the robustness of this social-ecological system (SES)* and then understand the ecological and socioeconomic context that produces these characteristics.

An SES is an ecological system linked to and affected by one or more social systems. It is defined as the subset of social systems in which some of the interdependent relationships among humans are mediated through interacting biophysical and nonhuman biological units (Anderies et al. 2004).

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* For convenience, a list of acronyms is given at the end of the paper.

Although this urban fishery SES is singular, its challenges are not unique. Increasing urbanization in the lower Pacific Northwest states of Washington, Oregon, Idaho, and California have pushed wild salmon populations to the brink of extinction (Netboy 1980, Nehlsen et al. 1991, Cone and Ridlington 1996, Huntington et al. 1996, National Research Council 1996, Gresh et al. 2000). Widespread public support has leveraged millions of restoration dollars to prevent the loss of salmon populations, and still they are disappearing (Lee 1993; McGinnis 1994, 1995). While the reasons for the failure of salmon restoration in the lower Pacific Northwest are complicated, one of the main drivers may have been the allocation of funds to address the biophysical symptoms, rather than the socioeconomic causes of these symptoms.

What methodological approach can best help managers delineate the socioeconomic causes of biophysical degradation within an urban SES, and can this approach be used to better achieve the goals identified by users and public infrastructure providers that contribute to robustness? I propose that the robustness framework and Ostrom's (1990) institutional design principles can help sport fishery managers contextualize the biophysical problems associated with the management of sport fisheries by evaluating the institutional robustness of this urban SES.

The complex interactions between the components of SESs have been studied in commercial fisheries (McHugh 1975, Finlayson and McCay 1998, Acheson 2003), but urban sport fisheries have received little attention. I define hatchery fish as fish produced from brood stock by artificial spawning in a hatchery environment. Conversely, wild fish are produced by natural spawning in natural fish habitat by parents that were spawned and reared in natural fish habitat.

The Lower Ship Creek Fishery SES is examined because it contains clearly identifiable interactions between the biological and social systems. The social interactions then can be studied to determine how and why biophysical symptoms are produced. When anglers (resource users) fish Lower Ship Creek, they interact not only with each other and the fish, but also with public infrastructure providers, who interact with each other as well. Public infrastructure providers are the agencies, businesses, and organizations that directly or indirectly contribute to the operation and maintenance of the fishery.

Theoretical Background

I examine the SES's response to the engineered component (the hatchery fishery) through the robustness framework. I chose to apply a robustness framework rather than resilience theory because robustness encompasses the unique attributes of this SES, which has relatively weak feedbacks among its designed and self-organized components. Although robustness is a more appropriate analytical framework than resilience for this SES, this study acknowledges that the robustness of semiengineered systems contributes to the overall resilience of communities. For example, the robustness of the Lower Ship Creek sport fishery contributes to the resilience of Anchorage, AK, by increasing local food and recreation options and supporting a diverse set of businesses.

Numerous studies have explored traditional and modern management in social-ecological systems (Berkes and Folke 1998, Gunderson and Holling 2002, Gunderson and Pritchard 2002, Berkes et al. 2003, Dasgupta and Mäler 2004, Folke 2004, Walker et al. 2006, Prediger et al. 2011). Most of these studies have focused on identifying the ecological and social sources of resilience that would enable the system to persist in its current state or management techniques that might increase the system's resilience. While the goals of this study are similar, the semiengineered characteristics of an urban fishery SES are different.

Holling (1996) distinguished between two types of resilience: engineering resilience and ecological resilience. Engineering resilience assumes that ecological systems exist close to a stable steady-state and measures the ability of a system to return to this steady-state following a perturbation (Pimm 1984). An example of engineering resilience is a bridge, which one would prefer to be close to its stable steady-state. When wind causes the bridge to oscillate and leads to its destruction, an undesirable steady-state is reached.

The concept of ecological resilience addresses the amount of change or disruption that a system can sustain before changing to an alternative state characterized by a different set of critical processes, structures, and interactions (Walker et al. 2004). Although this "tipping point" approach is conceptually consistent with robustness and appropriate to many natural resource issues, it may not provide answers for managers of semiengineered systems, where structures and interactions among components are more tightly constrained by human design.

Robustness first emerged in the field of engineering. The robust design methods, or the Taguchi Methods, make companies more competitive through more efficient development processes. Taguchi et al. (2000) define robustness as the state where the technology, product, or process is minimally sensitive to factors causing variability (either in the manufacturing or user's environment) and aging at the lowest unit manufacturing cost. The Taguchi Methods greatly improve engineering productivity by consciously considering the noise factors (environmental variation during the product's usage, manufacturing variation, and component deterioration) and the cost of failure in the field (Phadke 1989). Companies such as Ford, Minolta, NASA, and Xerox have all successfully used these methods (Taguchi et al. 2000).

Similar to what occurs manually in engineering, biological systems naturally develop responses to survive variable conditions. Developmental biology uses the concept of developmental robustness to describe the ability of an organism to continue to grow despite encounters with disturbances (Keller 2002, Felix and Wagner 2008). Robustness is also used in the field of community ecology (MacArthur and Wilson 1967, Tilman et al. 1996).

The field of social science uses the concept of robustness in the study of the institutional governance of common pool resources. Shepsle (1989) stated that social systems were considered robust if they were long-lived and governed by operational rules that had been devised and modified over time according to a set of collective choice rules. Because of the diverse range of operational and collective choice rules found in different social systems, it became apparent that more general design principles were needed to characterize common-pool resource institutions.

When Ostrom (1990) derived a set of design principles from studies of small-scale, long-enduring institutions for governing common-pool natural resources, she did not initially connect them with the concept of robustness. These principles were based on years of field work and case studies of simple and self-contained to complex and linked systems and have been well tested over the last two decades (De Moor et al. 2002, Kaijser 2002, Dietz et al. 2003). Ostrom (1990) eventually paired the concept of robustness with the design principles by stating that a social-ecological system is likely to be robust if it meets many (but perhaps not all) of these principles (Ostrom 1999, 2002, 2005; Ostrom et al. 2003).

Since SESs contain both engineered and biological components, they also experience variability and develop responses to disturbance. As applied to social-ecological systems, robustness is defined as "the maintenance of some desired system characteristics despite fluctuations in the behavior of its component parts or its environment" (Carlson and Doyle, 2002, p. 2539). Levin and Sugihara (2007, p. 27) clarify the difference between the use of robustness in engineering and ecology by stating, "complex adaptive systems are systems in which whatever robustness exists has to emerge from the collective properties of the individual units that make up the system; there is no one planner or manager whose decisions completely control the system."

An SES that is subjected to a particular type and degree of variability may become highly optimized to tolerate that variability and become more sensitive to new disturbances (this characteristic of adaptive systems is referred to as highly optimized tolerance, or HOT) (Carlson and Doyle 2002). Therefore, robustness emphasizes the cost-benefit tradeoffs associated with systems designed to cope with uncertainty (Anderies et al. 2004, Janssen and Anderies 2007). This emphasis is especially relevant to an urban fishery SES, where the engineered components often generate a tradeoff through the replacement of wild fish by hatchery fish. Perceived environmental problems, social conflicts, and economic fluctuations all produce challenges, but with the proper infrastructure, no single shock is likely to bring ruin to a robust system.

The National Research Council (1999, 2002), the Millennium Ecosystem Assessment (2003), and the Consortium for Sustainable Development (International Council for Science, Initiative on Science and Technology for Sustainability, and Third World Academy of Science; Walker et al. 2004) all have focused increasing attention on the concepts of robustness, vulnerability, and risk. More recently, Janssen et al. (2007) examined the robustness of SESs to spatial and temporal variability to determine why some long-lived SESs persist in the face of change and others do not. Anderies et al. (2007) applied the robustness framework to sustainability science to extract broader themes for the management of resources under uncertainty. Levin and Lubchenco (2008) have applied robustness to the management of marine ecosystems.

Both ecological resilience and robustness denote the ability of a system to maintain its macroscopic functional features (e.g., species diversity) rather than the unattainable possibility of constancy (Webb and Levin 2005). The functional robustness or resilience of an ecosystem can be maintained despite some species extinction under conditions where other functionally similar species maintain the same ecosystem properties.

Although ecological resilience and robustness are frequently used interchangeably (Adger et al. 2005, Levin and Lubchenco 2008), there are important differences. Ecologically resilient systems, for example, are generally characterized as evolved systems that demonstrate high diversity, ecological variability, modularity, slow variables stabilized by tight feedbacks, social capital, innovation, overlap in governance, and sustained ecosystem services (Walker and Salt 2006). The characteristics of ecologically resilient systems are often poorly developed in human-designed and human-operated systems.

Unlike the ecological resilience perspective, which often considers human activities as perturbations of an ecological system, robustness considers SESs where humans develop institutional feedback loops to respond to perturbations (Janssen and Anderies 2007). Robust systems are generally characterized as partly designed systems with both self-organized and designed components (Anderies et al. 2003). Crafted institutional arrangements aim to stimulate and support a particular performance of an SES, just as engineers design systems to meet certain design criteria (Janssen and Anderies 2007). Since urban hatchery fisheries are partly designed systems that contain both engineered (i.e., hatchery fish) and biological (i.e., nutrient cycling) components, robustness is a fitting framework for this particular case study.

The timeframe of analysis differs between resilience and robustness as well. Robustness focuses on the ability of an SES to maintain its social and (or) ecological domain of attraction within a specified time frame (Anderies et al. 2003). A system may be robust during one time period and not in another; such is not the case with resilience, which seeks to attain resilience without lapses over a long time period.

SES robustness depends largely upon the ability of its public infrastructure providers to respond to coinciding occurrences of economic, social, and ecological changes (Anderies et al. 2004). When one resource collapses, managers have the ability to achieve the desired outcome through the substitution of another valued good. Management decisions rely upon feedbacks between both slow (e.g., evolution, long-lived institutions) and fast variables (e.g., pollution event, organizational collapse) (Carpenter and Gunderson 2001). Managers are able to make predictions based on slow variables, but the self-organizing properties of ecological and social systems cause increased uncertainty over time (Levin 2000). It is therefore important to examine both self-organized and engineered components when determining robustness.

The hatcheries, which provide public infrastructure, are an engineered component within the SES that lacks many of the characteristics thought to characterize a resilient system. The hatcheries eliminate the diversity and number of species through several mechanisms. They produce larger numbers of targeted sport fish species than the lower creek could naturally support (Alaska Department of Fish and Game 2007). They remove ecological variability by artificially controlling population levels and restricting genetic diversity. Hatchery fish are immune to natural sources of population variability because they are raised in a controlled environment until they are large and strong enough to be released and therefore are not as susceptible to the effects of instream scouring, temperature changes, predation, or pollution. Hatchery fish, therefore, also are unlikely to experience stress related to the high number of pollution events within Lower Ship Creek. For example, the Ship Creek watershed and Cook Inlet experienced 11 spills of petroleum and other organic compounds from October 1995 to July 2002 (U.S. Environmental Protection Agency 2002).

Despite the cumulative effects of urbanization, the hatcheries do not respond to these feedbacks and continue to produce large numbers of salmon because they are produced in a controlled environment. Wild fish populations, which spend their growth phase in the stream, generally decline in response to urbanization (Klein 1979, Steedman 1988, Limburg and Schmidt 1990, Wang et al. 1997, Yoder et al. 1999).

The hatcheries are managed under a single agency (Alaska Department of Fish and Game 2007) rather than under overlapping governance among agencies. The Alaska Department of Fish and Game (ADFG) does not seek innovation in managing fish stocks (Alaska Department of Fish and Game 2003 a). Most importantly, the hatcheries do not address all of the ecosystem services affected by its production. Since little feedback on the social and ecological effects of the fishery influence the ADFG decisions that drive the fishery, resilience theory fails to adequately address the complexity of this and other engineered systems.

Although biologists know approximately how many fish will annually return to Ship Creek, the effect of this component on the greater SES is not known. Management challenges increase when engineered components interact with natural components because the artificial optimization of one system can produce negative effects on other components of the system. One agency may reap the benefits from a well engineered system, while others pay the costs.

Robustness and Ostrom's (1990) design principles may allow managers to better understand the character of and interactions between the components of this semiengineered, urban SES to reduce effects that decrease the robustness of this SES.

Methods

In order to help managers better address the causes of biophysical degradation, I (1) identify and describe the relevant socioeconomic and ecological systems; (2) outline the desired system characteristics, as formally defined by resource users and public infrastructure providers; (3) discuss the interactions within and between these systems, using the concepts of strategic interaction established by Anderies et al. (2004); and (4) use Ostrom's design principles to identify opportunities for increased SES robustness (Anderies et al. 2004).

The SES, its relevant components, and the interactions between the social and ecological systems were defined and analyzed using Anderies et al.'s (2004) framework. The ecological and social components include the components that most directly influence the fishery. For example, the ecological components of water quality and water quantity can affect hatchery production, which controls the fishery. Lower Ship Creek's public infrastructure providers and recreational and subsistence users are the social components that most directly affect the fishery.

The next step is to identify the desires of the public infrastructure providers and users in order to determine whether common goals can be established for the SES and identify potential sources of conflict. The desired social-ecological components of this SES are formally defined by the mandates and missions of public infrastructure providers, including municipal, State and Federal agencies, nonprofit organizations, and local businesses, and by the results of a standardized questionnaire that was sent to each public infrastructure provider. The desired social-ecological components of both subsistence and recreational anglers are inferred from user surveys conducted by the Anchorage Waterways Council (AWC; Figure 1), as well as from the general interests and activities of the two groups.

Once the ecological and social components are identified, the interactions within and between these systems will be discussed using Anderies et al.'s (2004) concepts of strategic interaction. I assessed the fit of this SES to Ostrom's (1990) design principles by

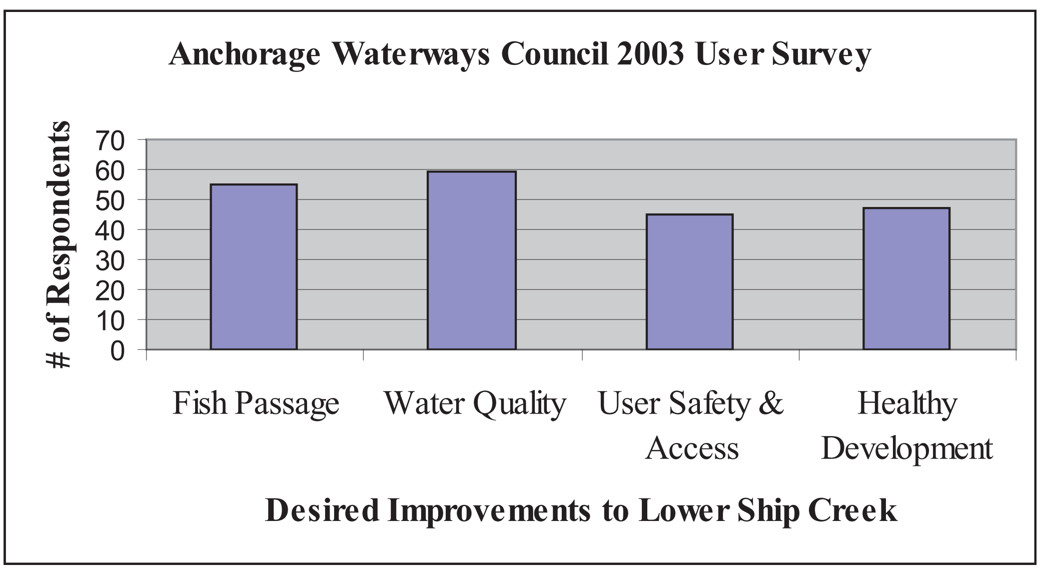

Figure 1. User survey, 2003

applying each of the principles to the Lower Ship Creek Fishery SES to determine which principles this SES failed to meet. I then analyzed the failed principles to identify opportunities to increase the overall robustness of this SES.

Components of the Lower Ship Creek Fishery SES

This SES is located within the Municipality of Anchorage (MOA) in downtown Anchorage, AK. The SES encompasses the last 1.45 km of Ship Creek and extends from the Knik Arm Power Plant Dam (KAPP) to the mouth of the creek at Cook Inlet (Note: KAPP Dam is also known as Chugach Power Plant).

The Ecological System

I define the Lower Ship Creek Fishery SES in terms of its ecological system (this section) and its social system (next section) (Figure 2, Tables 1, 2). The fishery under examination includes hatchery-produced chinook salmon (*Oncorhynchus tshawytscha*) and coho salmon (*Oncorhynchus kisutch*). The ecological components that most directly influence the fishery are the quantity and quality of water available for use by the hatcheries. Lower Ship Creek experiences a tidal range of 11.3 meters, which is the second highest range in North America (National Marine Fisheries Service 2002). The strength and height of these tides pose engineering challenges for the construction and maintenance of public infrastructure and streambank stabilization projects.

Historically, Ship Creek supported wild runs of all five Pacific salmon species—chinook, coho, pink (*Oncorhynchus gorbuscha*), chum (*Oncorhynchus keta*), and sockeye (*Oncorhynchus nerka*)—as well as Dolly Varden (*Salvelinus malma*), rainbow trout (*Salmo gairdneri*), and stickleback (*Gasterosteus aculeatus*) (Alaska Department of Fish and Game 2007). The run sizes of the original five salmon populations are unknown, but it is known that current hatchery-supported runs greatly exceed historical numbers (Alaska Department of Fish and Game 2007).

Two State-run hatcheries located on military bases now annually stock Ship Creek's popular fishery. The Fort Richardson Hatchery was built in 1958 and expanded in 1984. The Elmendorf Hatchery was built in 1965 and expanded in 1976. Ship Creek was first stocked with chinook salmon smolts in 1966 and coho smolts in 1968 (Alaska Department of Fish and Game 2007). A limited chinook salmon fishery first opened in 1970 (Alaska Department of Fish and Game 2007).

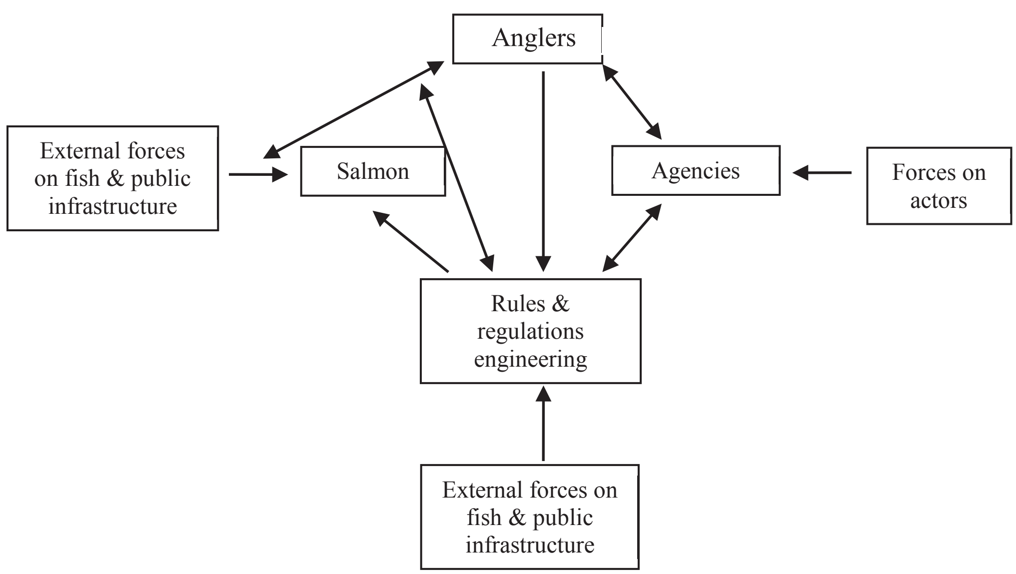

Figure 2. A conceptual model of the Ship Creek Fishery SES. Adapted from Anderies et al. (2004). Linkages are defined in Table 1.

Table 1. Lower Ship Creek Fishery SES linkages. Adapted from Anderies et al. (2004).

Linkages	Potential problems
Between fish and anglers	"Endless" availability of fish, free riding
Between users, businesses, the military, and agencies	Conflicting political agendas, free riding, inadequate inter/intra-agency information and communication, refusal to pay associated maintenance costs
Between public infrastructure and agencies	Unequal investments into the fishery, partitioning of responsibilities in ways that ignore interactions
Between public infrastructure and fish	Ineffective implementation of regulations, poor engineering and inappropriate construction
Between public infrastructure and fish dynamics	Unintended consequences
Between anglers and public infrastructure	Free riding
External forces on fish and public infrastructure	Destroyed fishery (via disease), collapsed public infrastructure (due to funding shortages)
Between forces on social actors	Increased demand, conflict

At present, the hatcheries continue to stock large runs of chinook and coho salmon in this SES. Declining wild runs of chum and pink salmon and Dolly Varden still spawn in the creek, but their numbers are undocumented. Sport fishing for salmon is permitted within the last 1.45 km of the creek, from the KAPP Dam to the mouth (Alaska Department of Fish and Game 2007).

The hatcheries had an agreement with Elmendorf Air Force Base (EAFB) and Fort Richardson Army Base (FRAB) that allowed the hatcheries to take excess heated water from the base power plants and combine it with surface water from Ship Creek. This agreement enabled the hatcheries to maintain a year-round rearing program. With the addition of heated water, the hatchery was able to produce ocean-ready smolts in one year instead of two years (Alaska Department of Fish and Game 2011 a). The closure of both military plants has resulted in considerable declines in the State's salmon stocking programs (Alaska Department of Fish and Game 2011 a).

The hatcheries, which now utilize Ship Creek surface water for their operations, are also concerned with the creek's water quantity and quality. Water quantity (instream flow) is a concern because it is a scarce resource that may be over-allocated. Instream flow is defined as the quantity of water that flows past a given point in a stream channel during one second. The lack of hydrologic data in Alaska is perhaps the most limiting factor in determining instream flow reservations, but other factors include costly and lengthy studies and administrative processes and expensive application fees (Harle and Estes 1993). Under the current adjudication system, permitted water use may exceed supply during peak use times because many water rights applications are still pending (Estes 1998).

According to fecal coliform monitoring data collected by the Municipality of Anchorage from 1989–1994, the water quality criteria for drinking water and contact recreation were exceeded at various times (U.S. Environmental Protection Agency 2004). Since 1990, the Alaska Department of Environmental Conservation (ADEC) has listed Ship Creek from the Glenn Highway bridge to its mouth at Cook Inlet as a 303(d) Impaired Waterbody due to the presence of petroleum hydrocarbons, oil, grease, fecal coliform bacteria, and biological community alteration from urban runoff and industrial spills (U.S. Environmental Protection Agency 2002, 2004).

In 2007, the presence of disease (*Myxobolus cerebralis)* in Ship Creek (Arsan 2006) forced ADFG's Elmendorf Hatchery to limit the introduction of hatchery fish to land-locked systems. To prevent further losses in production related to changes in water quantity or quality, the State has constructed the $96 million Jack Hernandez State Fish Hatchery facility that will implement well-water reuse systems on the banks of Ship Creek (Alaska Department of Fish and Game 2011 a).

Other exogenous controls, such as oceanic and climatic cycles and predation, also influence the survival rate of both wild and hatchery salmon populations (Carpenter et al. 1992). Major climatic and oceanic shifts have significantly altered salmon survival in the Pacific Northwest (Anderson 2000). Predators, such as marine mammals and birds, are often identified as additional factors contributing to salmon decline (Smith et al. 1998). While the effects of these variables are difficult to quantify, they should be considered because they could play an important role in the long-term robustness of the system.

The Social System

The social components that most directly influence the fishery are the public infrastructure providers and resource users. The public infrastructure providers directly or indirectly support the operation and maintenance of the fishery by providing services such as fish production or trash removal services. Resource users consume the production of the fish and contribute to the public infrastructure providers via annual fees.

The public infrastructure providers and the resource users interact within a complex network of private land ownerships and Federal and State jurisdictions. Resource users that purchase a fishing license and follow regulations are allowed to participate in the fishery. Ownership of the land surrounding Lower Ship Creek is complicated. The State of Alaska owns and has jurisdiction over the streambed of Ship Creek. The MOA owns and has jurisdiction over a 30-foot setback on either side of the creek and is responsible for the maintenance of the infrastructure, such as trails, benches, and lighting, that exists within this setback. The MOA also owns the newly constructed bridge near the mouth of the creek. The Alaska Railroad Corporation (ARRC) owns the land adjacent to the last 1.4 km of Lower Ship Creek. Although much of the ARRC land is long-term leased to local businesses, the ARRC has ultimate jurisdiction over these lands. The ARRC also owns a railroad bridge that crosses over the creek within the fishery.

The State of Alaska's Sport Fish Division of the ADFG has management authority under Title 16 and Title 41 in the State of Alaska's statues, makes all decisions regarding the sport fishery on Ship Creek, and is responsible for maintaining garbage cans and portable toilets during the fishery openings. The Habitat Division of the ADFG has jurisdiction over the quantity of water in Lower Ship Creek because it has obtained a water right that established a minimum instream flow. Another State agency, the Alaska Department of Environmental Conservation (ADEC), has jurisdiction over Ship Creek's water quality and can impose sanctions if water quality standards are not met under the Coastal Zone Management Act. The U.S. Environmental Protection Agency (EPA) also has jurisdiction over water quality and can impose sanctions under the National Environmental Protection Act (NEPA). The EPA and U.S. Army Corps of Engineers (USACE) can impose sanctions under Sections 401 and 404 of the Clean Water Act. The USACE can also impose sanctions under Section 10 of the Rivers and Harbors Act. The Alaska State Troopers have legal jurisdiction and the ability to impose sanctions under Title 11 in the Alaska Statutes. The National Marine Fisheries Service (NMFS) has regulatory authority under the Magnuson-Stevens Fishery Conservation and Management Act and the Marine Mammal Protection Act. The U.S. Fish and Wildlife Service (USFWS) has the ability to provide comments on any development actions within Lower Ship Creek but is not a regulatory authority. Both NMFS and USFWS have regulatory authority under the Endangered Species Act. The U.S. Geological Survey (USGS) collects hydrologic data on Ship Creek but does not have regulatory authority and rarely comments on development actions.

Several of these agencies are working to improve the aesthetic appeal and environmental quality of Lower Ship Creek (see "Public Infrastructure Providers" below for more details). The SES is valued by residents because of the creek's unique history and accessibility. The creek once supplied Alaska's Native residents, the Dena'ina, with abundant salmon runs and is Anchorage's original town site. Most recently, Mayor Mark Begich identified the Ship Creek Revitalization Project as one of the top priorities of his administration (Municipality of Anchorage 2007). Many current residents learned how to fish on Ship Creek and are now teaching their children how to catch a salmon in downtown Anchorage. Local businesses recognize the economic potential of the SES and are interested in drawing more people to Ship Creek.

The Lower Ship Creek Fishery SES provides the highest economic benefit to the state of any hatchery program, contributing an advertised \$7.3 million to the economy (King 2004). An annual average (1996–2005) of 47,000 angler days of effort produce an average catch of 8,900 chinook salmon and 16,500 coho salmon (Alaska Department of Fish and Game 2007). This SES also benefits Grace Alaska's Downtown Soup Kitchen, which organizes two salmon derbies each summer (Alaska Department of Fish and Game 2007). Ship Creek's hatcheries represent two of the three State-run hatcheries and supply fish for local creeks throughout Alaska, including Upper Cook Inlet, Resurrection Bay, and Prince William Sound (Alaska Department of Fish and Game 2003, Loopstra and Hansen 2005).

Although this easily accessed fishery provides large socioeconomic benefits, it also imposes the external costs commonly associated with common pool resources (Hardin 1968, Ostrom 1990, Ostrom and Field 1999, Dietz et al. 2003, Schlüter and Post-Wostl 2007). As the ADFG has increased the release of hatchery fish over the years (Figure 3), no provisions have been made for the fishery's infrastructure. The lack of public infrastructure, such as bathrooms, fish cleaning stations, and garbage cans, and an increase in trespassing, illegal fishing, angler conflicts, and erosion create annual problems within the fishery (Alaska Railroad Corporation 1999, Anchorage Waterways Council 2011 a). The MOA, ARRC, local law enforcement entities, NMFS, AWC, ADFG, and other resource agencies have all spent money to mitigate pollution by updating and constructing infrastructure (National Marine Fisheries Service 2002, National Oceanic and Atmospheric Administration 2005, Alaska Railroad Corporation 2006, Anchorage Waterways Council 2011 a, Municipality of Anchorage 2007).

Interaction of Ecological and Social Subsystems

The interaction of the ecological and socioeconomic components of the SES determines the system's robustness. As demand for one ecological component increases, the socioeconomic components can react by limiting or compensating for that increase. For example, during the 1970s and early 1980s when groundwater extracted from aquifers near Ship Creek was the principal source of the MOA water supply, areawide declines in groundwater levels resulted in near-record low streamflows in Ship Creek (Moran and Galloway 2006). Because of variable flows and water quality of Ship Creek, its use as a water supply was minimized to maintain flows for aquatic and riparian habitat and to mitigate fecal coliform contamination (Alaska Department of Environmental Conservation 2004). The MOA now receives most of its water supply from Eklutna Lake.

The pressures that have been exerted on Ship Creek's ecosystem processes have created a management need for the maintenance of this fishery's socioeconomic components. As more users come to Ship Creek in search of salmon, public infrastructure providers will be pressured to maintain and (or) expand services to deal with trespassing and safety issues. Since agencies work within existing and sometimes opposing mandates and users have different needs, it is beneficial to carefully examine the formally defined interests of both users and public infrastructure providers.

Desired Ecological and Socioeconomic Components

The major components of this SES can be divided into the categories of essential and desirable. The essential components include the minimum ecological components needed to maintain a robust fishery. A robust urban fishery will include (1) efficient hatcheries, (2) public infrastructure, and (3) sufficient water quality and quantity to sustain hatchery production. The desired components include the characteristics desired by stakeholders within the SES and will be discussed below (Table 2). If either the

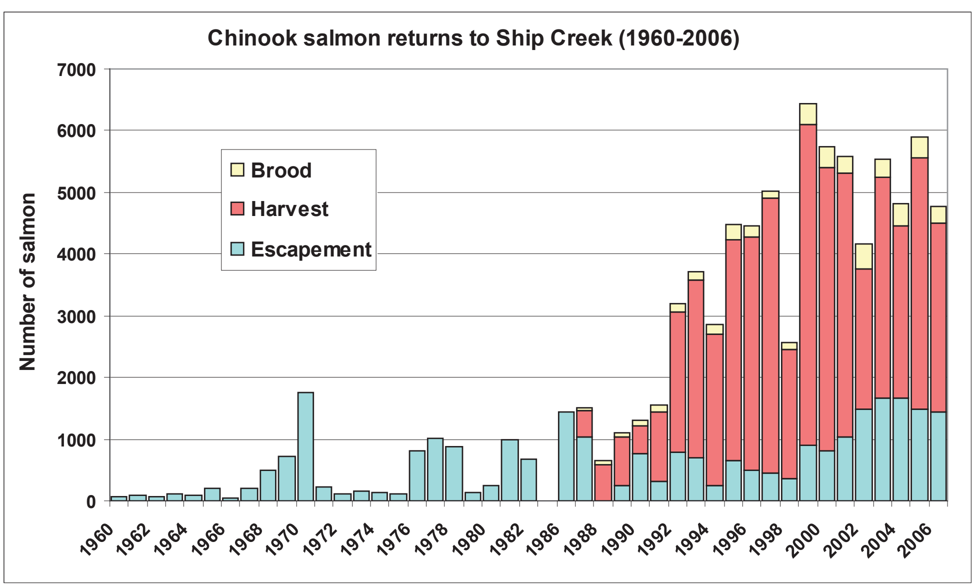

Figure 3. Ship Creek Chinook Salmon returns for 1960–2006. (Alaska Department of Fish and Game 2007)

social (public infrastructure) or ecological (water quality and quantity) components collapse, then this SES would lose its robustness (see “Assessing Robustness” below).

Table 2. Public infrastructure providers’ desired social and ecological components of the Lower Ship Creek Fishery.

Desired social and ecological components	Infrastructure providers
Improved water quality, contaminant removal	EPA, AWC, ARRC
Restored fish passage and habitat	USFWS, ADFG, NMFS, AWC
Increased stream/riparian function	USFWS, AWC
Angling opportunity	USFWS, MOA, ADFG, AWC
Decreased erosion	ARRC, AWC
Decreased trespassing, safety issues	ARRC
Construction of new hatchery and visitor center	ADFG, MOA
Maximized harvest, minimized maintenance costs	ADFG
Increased economic activities in district	MOA

Public Infrastructure Providers

Removing Ship Creek from the 303(d) List of Impaired Waterways is a high priority for both the ADEC and the EPA, which both possess regulatory authority within Lower Ship Creek (U.S. Environmental Protection Agency 2006). The mission of both organizations includes the protection of human health and the environment (U.S. Environmental Protection Agency 2011 a, Alaska Department of Environmental Conservation 2008). In May 2004, the EPA approved the ADEC’s total maximum daily load (TMDL) plan to impose controls on Ship Creek that will improve water quality by reducing fecal coliform bacteria (U.S. Environmental Protection Agency 2004). A TMDL is a calculation of the maximum amount of a pollutant that a waterbody can receive and still meet water quality standards and an allocation of that amount to the pollutant’s sources (U.S. Environmental Protection Agency 2011 b). The EPA is working with the ARRC to monitor Ship Creek’s water quality for petroleum (U.S. Environmental Protection Agency 2006). This monitoring will enable the ADEC to determine the best recovery actions for Ship Creek, which may involve the development of a TMDL or similar recovery plan for petroleum (U.S. Environmental Protection Agency 2006).

The mission of the U.S. Fish and Wildlife Service (USFWS) is “to work with others to conserve, protect and enhance fish, wildlife, and plants and their habitats for the continuing benefit of the American people” (U.S. Fish and Wildlife Service 2011). The USFWS has the ability to provide comments on actions taken within Lower Ship Creek, but it does not have any regulatory authority. The primary short-term goal for USFWS on Ship Creek is to remove barriers to anadromous and resident fish passage through partial or complete dam removal or fish-way improvement so that the creek is largely barrier free by 2012 (M. Roy, written response to author’s Ship Creek questionnaire, 2007). The long-term goal of the USFWS for Ship Creek is to create a barrier free, urban system that achieves a socially accepted balance of augmented and natural fish runs providing ample angling opportunity, relatively natural stream function, and substantially improved riparian function (M. Roy, written commun., 2007).

In accordance with its mission of “creating development opportunities for the highest public benefit, using innovation, partnerships, sound planning, and incentives,” the MOA is interested in sustaining the Ship Creek’s unique urban fishery, natural values, and economic activities (Municipality of Anchorage 2007, 2008). The MOA would also like to see a new hatchery and visitor center built on EAFB (Municipality of Anchorage 2007). As a land owner, the MOA has to grant permission for projects within the 30-foot setback on either side of the creek (see earlier discussion).

The primary mission of the ADFG is “to protect, maintain, and improve the fish, game, and aquatic plant resources of the state and manage their use and development in the best interest of the economy and the well-being of the people of the state, consistent with the sustained yield principle” (Alaska Department of Fish and Game 2011 b). The ADFG lists Ship Creek as anadromous in its “Catalog of Waters Important for the Spawning, Rearing or Migration of Anadromous Fishes” (Alaska Department of Fish and Game 2008). It is an important designation because Ship Creek is technically afforded protection from any activities that would harm the habitat of anadromous fish under Alaska Statute 41.14.870 (Alaska Department of Fish and Game 2007).

The goals of the ADFG's Ship Creek hatcheries include (1) generating at least 50,000 angler days of opportunity directed at stocked chinook and coho salmon, (2) meeting the brood stock goals of 500 chinook salmon and 1,000 coho salmon, (3) maximizing the harvest of surplus hatchery salmon, (4) improving existing hatchery operations, and (5) accommodating future plans for a new fish hatchery and (or) visitor facility adjacent to the creek (Alaska Department of Fish and Game 2007). The ADFG is also interested in restoring fish passage to upper Ship Creek and reducing or minimizing operation and maintenance requirements caused by debris, sedimentation, and icing on Ship Creek (Alaska Department of Fish and Game 2007).

The mission of the ARRC is to be profitable while delivering safe, high quality service to their freight, passenger, and real estate customers and to foster the development of Alaska's economy by integrating railroad and rail-belt community development plans (Alaska Railroad Corporation 2011). As the landowner of most of the property and the entire streambed within the SES, the ARRC is concerned about trespassing and safety issues associated with the fishery and pedestrian traffic and the effect that these issues may have on their leaseholders (Alaska Railroad Corporation 2006). The ARRC is currently working with the EPA to search for and identify possible contaminants and devise strategies for either eliminating or mitigating risks according to the Comprehensive Environmental Response, Compensation, and Liability Act regulatory guidelines (U.S. Environmental Protection Agency 2002, Alaska Railroad Corporation 2007).

The NMFS provides for the stewardship of living marine resources through science-based conservation and management and the promotion of healthy ecosystems (National Marine Fisheries Service 2011). The agency defines Knik Arm, including the Ship Creek estuary, as essential fish habitat (EFH) under the Magnuson-Stevens Fishery Conservation and Management Act for natural runs of migrating and (or) rearing chinook salmon, coho salmon, pink salmon, and chum salmon (Mecum 2006). In the past five years, the NMFS has contributed a considerable amount of money through federal grants for fish passage and habitat restoration projects to benefit natural runs of salmon adjacent to and on Ship Creek (National Marine Fisheries Service 2002, National Oceanic and Atmospheric Administration 2005, Municipality of Anchorage 2007).

The mission of the nonprofit organization Anchorage Waterways Council (AWC) is "to protect, restore, and enhance the waterways, wetlands, and associated uplands of Anchorage" (Anchorage Waterways Council 2011 b). The AWC would like to see the removal of the lower three dams on Ship Creek, improved water quality, the restoration of wild runs in addition to the hatchery runs, and the construction of angler infrastructure (Anchorage Waterways Council 2011 a).

There are fundamental differences between the missions of the ARRC and the other public infrastructure providers, such as the AWC and State and Federal agencies. Any efforts to improve angler opportunities, fish and wildlife habitat, or water quality fall outside of the ARRC's primary mission to provide safe transportation and improve Alaska's economy.

While each of the public infrastructure entities works under different mission statements, most of their goals are related. The Federal natural resource agencies (USFWS, NMFS) and the nonprofit organization (AWC) are all concerned with the restoration of fish and wildlife habitat, including fish passage, and stream and riparian functions. The EPA and ARRC are concerned with improving water quality. In addition to these goals, the Federal agencies and nonprofit organization are also interested in maintaining angler access and opportunity. The AWC, MOA, and ARRC are all concerned with preventing erosion. Since most of the goals of the public infrastructure providers are compatible, a broadly desirable outcome is possible without having to address the major tradeoffs often associated with common-pool resources.

Resource Users

All resource users fish Lower Ship Creek, but there is varying interest in fish and wildlife restoration, fish passage, and water quality among resource users. According to a user survey conducted by the Anchorage Waterways Council (AWC) in 2003 (Figure 1), users are primarily interested in improving the water quality of the creek. Other concerns documented by the AWC survey in order of importance to users are fish passage, user safety and access, and "healthy" development. "Healthy" development is defined as non-industrialized development, such as bait shops, that supports the Lower Ship Creek Fishery. While the survey did show that both residents and visitors participate in the fishery, the exact ratio of these user types is unknown. Based on the observations of volunteers conducting the surveys, the fishery's resource users can be separated into two categories, as fishing for (1) recreation or (2) subsistence, as determined by the type of fishing gear used. Recreational users tended to use more expensive and elaborate fishing gear than subsistence users.

Subsistence users rely upon Ship Creek's salmon as a food source and spend a considerable amount of time fishing the creek. They are concerned about any effect that could decrease their ability to catch fish. They want high returns of fish populations and catch limits and low licensing fees. An increase in licensing fees could prevent their participation in the fishery and (or) lead to an increase in illegal fishing efforts since many of the subsistence users have low incomes. They do not support dam removal because the dam currently impedes fish movement and traps fish within the Lower Ship Creek fishing area. Its removal would therefore decrease their ability to catch fish.

Recreational users tend to spend less time on the creek and do not rely upon Ship Creek salmon as a food source. They want an aesthetically pleasing and safe environment for their sport fishing experience. They generally support the restoration of fish passage through dam removal because it would make catching fish more difficult and enhance their fishing experience. Increased licensing fees do not curtail their involvement in the fishery.

Subsistence and recreational users do have common interests as well. All users benefit from using public infrastructure to safely access the creek to fish. Several accidents over the years have affected both recreational and subsistence users' abilities to safely participate in the fishery. In 2005, three failing culverts at the mouth of Ship Creek were removed to improve recreational and subsistence user safety. Most users are also interested in the construction of a new hatchery because it would increase the fishery's robustness. Currently, there is widespread concern that the outdated hatchery facility will be unable to sustain current fish release levels because of inefficient production methods and the lack of an uncontaminated water supply (Alaska Department of Fish and Game 2011 a).

Results: Assessing Robustness

An SES can broadly be considered as robust if it "prevents the ecological systems upon which it relies from moving into a new domain of attraction that cannot support a human population, or induces a transition that causes long-term human suffering" (Anderies et al. 2004, p. 7). Since the examination of cost-benefit tradeoffs is inherent to the robustness framework (Anderies et al. 2004, 2006), it is beneficial to conceptualize the strengths and weaknesses of societies and ecosystems.

Using Ostrom's (1990) design principles derived from studies of long-enduring institutions for governing resources, the robustness of this SES can be assessed based on the ability of the public infrastructure providers to create a flexible yet inclusive management structure that allows the SES to adapt to changes in angler numbers, stream conditions, and development pressures (Table 3).

Clearly Defined Boundaries

The Ship Creek fishery has clearly defined boundaries (Table 3). The ADFG defines the salmon fishery 1.45 km from 15 meters below the KAPP Dam to the mouth of the creek at Cook Inlet. Anyone who has purchased a sport fishing license from ADFG and abides by the fishing regulations has a right to fish the creek.

Graduated Sanctions

Graduated sanctions do exist within the Lower Ship Creek Fishery. The bail schedule for sport fish violations takes into account the severity of the violation. A single violation only receives one penalty, with increasing numbers of violations carrying different penalties. However, if violators harvest too many fish, they are fined a species-dependent set amount for each fish they have taken over the legal bag limit. For example, in the winter of 2008 an individual was caught with more than 100 fish over his limit on another Anchorage creek. His fine amounted to more than $7,000 (D. Bosch, Alaska Department of Fish and Game, Sport Fish Division, personal commun., 2008).

Conflict Resolution Mechanisms

Ship Creek anglers and officials have access to low-cost local arenas to resolve conflict. The most obvious conflicts occur between anglers during peak fishing times when space is limited and the fish are running. Safety concerns are usually quickly addressed by the troopers or the ARRC police because of the creek's easy access. Plain-clothed troopers and the ARRC police patrol the creek on a daily basis. Citizens may contact the ADFG or the Alaska State Troopers with their concerns.

Minimal Recognition of Rights to Organize

The rights to organize are present within this system. If users wanted to create their own institution, they could do so and claim rights to participate in management decisions. However, the necessary institutional framework and social networks are currently lacking, and the diverse and scattered populations of users and their negative opinion of the agencies that govern their actions make organization highly unlikely. It is unclear whether the users could form a group that adequately represents all interests and work cooperatively with governing organizations.

Proportional Equivalence between Benefits and Costs

There is a disproportionate relationship between the benefits and costs of this SES (Krupa and Valcic 2011). The costs of maintaining this fishery are currently not accounted for while the benefits are routinely advertised as producing $7.3 million annually to the State of Alaska (King 2004). The SES benefits include revenues associated with the purchase of sport fishing permits (ADFG), tourism (local businesses, MOA), annual salmon derby entrance fees (nonprofit organization), and outfitting (local businesses). The SES costs include all facilities, services, and programs supporting the fishery as well as the costs paid to mitigate the problems created by the fishery (externalities). These costs are primarily borne by the MOA, ARRC, State of Alaska, ADFG, EAFB, FRAB, local businesses, and nonprofit organizations (Alaska Railroad Corporation 2006, Alaska Department of Fish and Game 2007, Anchorage Waterways Council 2011 a, Municipality of Anchorage 2007). The ADFG has not responded to these mounting costs by limiting the total allowable catch (TAC), which reduces user traffic. In fact, the number of angler user days has increased each year (Figure 3).

Collective-Choice Arrangements

There is no effective collective-choice arrangement on Ship Creek between the resource users and the ADFG. All rules regarding the fishery's TAC and openings are determined by the Sport Fish Division of the ADFG with survey input from Ship Creek anglers. However, there is no forum for dialogue with anglers or any mechanism for input from other public infrastructure providers. Since the other public infrastructure providers are directly affected by changes to the SES, their inclusion in the decisionmaking processes would likely increase this SES's robustness.

Table 3. Design principles derived from studies of long-enduring institutions for governing resources. Adapted to the Lower Ship Creek Fishery from Ostrom (1990).

Principles that characterize Ship Creek	Principles that do not characterize Ship Creek
<u>Clearly defined boundaries</u>	<u>Proportional equivalence between benefits and costs</u>
The physical boundaries of the resource system (Lower Ship Creek Fishery) and the anglers with rights to harvest salmon are clearly defined.	Rules do *not* allocate costs and benefits proportionately among infrastructure providers.
<u>Graduated sanctions</u>	<u>Collective-choice arrangements</u>
Users who violate fishing regulations receive graduated sanctions (depending on the seriousness and context of the offense) from officials accountable to publicly elected officials.	Anglers that harvest salmon are *not* included in the group who can modify harvest and protection rules.
<u>Conflict resolution mechanisms</u>	<u>Monitoring</u>
Anglers and enforcement officials have rapid access to low-cost local arenas to resolve conflict among users or between users and officials.	Monitors do *not* adequately audit biophysical conditions and user behavior, so infrastructure providers have no strong basis to manage adaptively for robustness.
<u>Minimal rights to organize</u>	
Anglers' rights to organize are not challenged by external governmental authorities, and users have long-term tenure rights to utilize the fishery.	

The ADFG is a State agency governed by the Alaska Board of Fisheries with several departments that provide different services and goals. Differing agendas can produce intra-agency tension between different departments, but there are opportunities for cooperation. One of the goals identified by the entire ADFG as a State agency is to optimize public participation in fish and wildlife pursuits (Alaska Department of Fish and Game 2011 b). The mission of the Habitat Division of ADFG is to preserve the State's fish and wildlife resources by protecting the areas they need to complete their life cycles (Alaska Department of Fish and Game 2011 c). This effort includes maintaining fish passage and instream flow. The ADFG therefore has incentive to work with public infrastructure providers to reduce erosion and protect fish and wildlife habitat within the fishery.

One potential challenge to including the public infrastructure providers in decisionmaking processes is their shifting roles within different scenarios. For example, the MOA's involvement in projects on Ship Creek has drastically increased with the election of Mayor Mark Begich. Another challenge is that the ADFG personnel who decide TAC are often politically appointed by the Governor and therefore are subjected to public scrutiny and influence.

Monitoring

The lack of user and biophysical monitoring restricts the system's robustness. Although the ADFG, Alaska State Troopers, and ARRC all monitor user licensing and behavior on Ship Creek, enforcement remains a problem in this easily accessed fishery. The salmon fishery below the KAPP Dam is closed nightly from 11 p.m. to 6 a.m. from May 25 to July 13, but the Alaska State Troopers routinely catch people catching fish during this period (Alaska State Troopers 2007). The ARRC also closely monitors user behavior to ensure their safety and prevent trespassing on its railroad tracks and bridges.

The USGS currently monitors the quantity of water in Ship Creek at two gauge stations. The AWC, EPA, and ADEC monitor the water quality of Ship Creek, but other biophysical characteristics, such as fish habitat and morphology, go unmonitored. Due to a lack of biophysical monitoring, the ecological (and resulting social and economic) costs and benefits of restoration projects are largely unknown and therefore are a source of conflict among public infrastructure providers.

Interactions

By evaluating the strengths and weaknesses of the relationships between the public infrastructure providers and resource users, the SES fails to meet three of Ostrom's (1990) seven design principles. The SES does not have (1) a proportional equivalence between benefits and costs, (2) collective-choice agreements, and (3) sufficient user and biophysical monitoring (Table 3).

In lacking the design principles of proportional equivalence between benefits and costs, collective-choice agreements, and sufficient user and biophysical monitoring, SES robustness decreases. The development of a proportional equivalence between benefits and costs and collective-choice agreements would address the problems of free riding and subtractability of use through the creation of rules (Anderies et al. 2004) but would fail to address the problem of enforcing these rules. User and biophysical monitoring would play a vital role in enforcing these rules and increasing SES robustness. If addressed in unison, these three opportunities could increase SES robustness and sustain the resource.

Conclusion

The challenges of regulating an urban engineered combat fishery are real but not insurmountable. Urban managers may be able to utilize the design principles (Ostrom 1990) within the robustness framework to distinguish the socioeconomic and ecological components of engineered systems and use this knowledge to more effectively maintain engineered systems. Future research into the similarities between interacting public infrastructure providers and resource users in other urban engineered SESs as well as the SES's ability to meet the design criteria would provide further insight into the components of robustness.

Urban systems possess great social, economic, and ecological value and can be maintained despite uncertain conditions, but this will require a paradigm shift within the public infrastructure providers. Currently, the six strongest providers sit at opposite ends of the engineered to wild spectrum (Figure 4). The other two providers, the ADEC and EPA, are mainly concerned with water quality and therefore are less concerned with the creek's engineering or wildness (Figure 4).

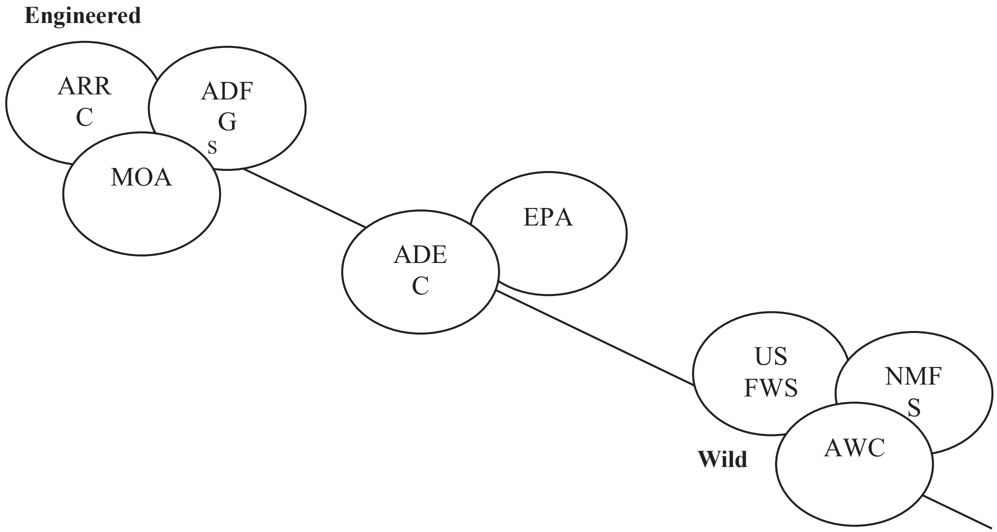

Figure 4. The paradigm divide within public infrastructure providers on Lower Ship Creek. (See the end of the paper for a list of abbreviations.)

Lower Ship Creek is neither engineered nor wild. It is a unique combination of biophysical components reacting with an engineered resource in an urban setting. The current challenges within this SES are the result of the public infrastructure providers' failure to address both of these components in their management efforts.

Currently, the USFWS, NMFS, and AWC are all trying to restore the creek to a more natural condition by improving the creek's overall fish and wildlife habitat and fish passage, but they are largely ignoring the need for the system's engineered and social components. Costly fish and wildlife habitat restoration is extremely difficult if not impossible to maintain within a highly trafficked area such as Lower Ship Creek. The MOA, ARRC, and ADFG are trying to engineer the creek on a reach by reach basis to meet public needs and expectations, but they are generally failing to consider the biophysical components of the creek in their project designs, which has cost agencies a considerable amount of money and has resulted in failed projects.

The good news is that no legal barriers prevent the public infrastructure providers from working together. In fact, many agency mandates and funding sources require the formation of partnerships (AWC, MOA, USFWS, and NMFS). All of the providers have partnered on a variety of reach-scale projects on Ship Creek. The challenge is to get these groups to work together within a more robust institutional framework to maintain Lower Ship Creek's biophysical and engineered components and create a robust fishery (Table 4).

Creating a Proportional Equivalence of Benefits and Costs

Studies of irrigation systems have shown that appropriation and provisions are two major sources of collective-action problems (Tang 1992, Lee 1994). Appropriation problems are time-independent and the result of how a limited resource is allocated (Ostrom 1990). Provision problems are time-dependent and the result of how the responsibility for building, repairing, or maintaining resource systems is assigned, as well as the appropriators' well being (Ostrom 1990). This urban fishery SES experiences problems of ineffective appropriation with provisions.

The high production of hatchery fish and the associated increases in use are causing appropriation problems within this SES. The first step in creating a proportional equivalence of benefits and costs is for the ADFG to address the appropriation problem by decreasing the total allowable catch (TAC) until adequate public infrastructure is in place to prevent further degradation to existing infrastructure and stream conditions or give actors in the system the choice of either reducing TAC or increasing infrastructure (Table 4). Physical (i.e., public) infrastructure is an important component of a robust SES because it determines the degree to which a commons can be exploited, the amount of waste produced by the use of the resource, and the effectiveness of resource and user monitoring (Dietz et al. 2003). For example, the use of relatively inexpensive barbed wire on grazing lands has decreased the cost of enforcing property rights (Krell

Table 4. Opportunities to increase the robustness of the Lower Ship Creek Fishery using three of Ostrom's (1990) design principles that this SES failed to meet.

Creating a proportional equivalence of benefits and costs	**Developing collective-choice agreements**	**Increasing user and biophysical monitoring**
Step One	Step One	Step One
ADFG decreases TAC until adequate public infrastructure is in place to prevent further degradation to existing infrastructure	ADFG creates a formal process for including public infrastructure provider input in their annual hatchery operation plans	ARRC and State Troopers increase patrols of Lower Ship Creek and strictly enforce existing ADFG fishing and ARRC trespassing regulations
Step Two	Step Two	Step Two
Public infrastructure providers identify improvements and maintenance costs needed to support the fishery at future TAC levels	Public infrastructure providers work together to define specific roles in the implementation of improvements and maintenance efforts	EPA and ADEC develop a long-term funding plan to support AWC's water monitoring efforts and USGS's water quantity monitoring at two existing sites
Step Three	Step Three	Step Three
Public infrastructure providers establish a cost-sharing agreement for improvements and maintenance costs needed to support the fishery at future TAC levels	The MOA's Watershed Task Force monitors these agreements and settles disputes through arbitration	Public infrastructure providers include a monitoring component into the design of every improvement and maintenance effort

2002). On Lower Ship Creek, the addition of walkways and staircases would decrease the need for and cost of conducting streambank restoration projects.

Provision problems within this SES exist because of inequities and confusion in the assignment of resource system responsibilities. For example, the ADFG currently benefits from the fishery but pays very few of its infrastructure costs. A more equitable cost sharing framework, such as the one established by a group of irrigators in Japan (Sarker and Itoh 2001), would enable the agencies to share project costs.

Therefore, the next step is for public infrastructure providers to work with the ADFG to identify improvements and maintenance costs needed to support the fishery at future TAC levels. These costs include both improvements (i.e., bathrooms and walkways) and ongoing maintenance (i.e., garbage removal and infrastructure repair) efforts. The public infrastructure providers can then establish a formal cost-sharing agreement for improvements and maintenance costs needed to support the fishery. Past projects, such as the removal of three failing culverts at the mouth of the creek, have been delayed because of disputes over who pays what costs. A formal cost-sharing agreement would reduce future cost disputes and animosity among providers.

Developing Collective-Choice Agreements

Effective governance requires the collection and communication of factual information about socioeconomic and ecological conditions so that managers can make appropriate decisions. Dialogue between the public infrastructure providers and users allows for the correct use of information, building of social capital, and the ability to change and deal with inevitable conflicts (Dietz et al. 2003). Sarker and Itoh (2001) state that sound coordination between social and physical capital has significantly contributed to the success of Japanese irrigation management.

Currently, there is a gap between the public infrastructure providers and users. This gap could lead to the construction of infrastructure that does not match the needs of the users. The creation of a linkage between public infrastructure providers and users has proven to be an important component of robust SESs (Levine 1977, Moore 1989, Lam 1996). When the bureaucrats from the Indonesian government introduced new rules and infrastructure into a rice production system and ignored the indigenous rules of the users, water shortages and pest outbreaks ensued (Lansing 1991). Although the individual characteristics of long-lasting common-pool resource SESs differ greatly, they all have resource users linked to public infrastructure providers (Coward 1979, Siy 1982, Martin and Yoder 1983, Laitos 1986, Maass and Anderson 1986, Blomquist 1992).

The ADFG is in a good position to bridge the existing gap between public infrastructure providers and users. The inclusion of public infrastructure providers into the annual hatchery planning process would enable the development of collective-choice agreements that would define specific roles in the implementation of relevant improvements and maintenance efforts (Table 4). The existing Mayor's Watershed Task Force would then monitor these agreements and settle disputes through mitigation.

Increasing User and Biophysical Monitoring

Increasing user and biophysical monitoring would protect the investment of improvements and maintenance costs on Lower Ship Creek. An increase of patrols would increase user safety through the strict enforcement of existing ADFG fishing and ARRC trespassing regulations. The AWC and USGS currently monitor water quality and quantity, respectively. Both of these organizations have experienced funding shortages that cut monitoring efforts in the past. To prevent future monitoring gaps, the EPA and ADEC could develop a long-term funding plan to support the AWC's water monitoring efforts and the USGS's water gauging at their two existing sites (Table 4). Another way to support user and physical monitoring efforts is for public infrastructure providers to fund and include a monitoring component in the design of every improvement and maintenance effort.

With the presence of night time violations and nonpoint source pollution within this SES, managers should be aware that monitoring and enforcement efforts may become economically inefficient (Colby 1995, Berkes and Folke 1998, Heal 1998). Combining user education and outreach with monitoring and enforcement may prove to be a more effective solution.

Coordination and Implementation

The Mayor's Watershed Task Force may be in a good position to bring the public infrastructure providers together to discuss the issue of robustness in its entirety and specifically address the implementation of each of the above steps (Table 4). Currently, the Task Force is a multiagency advisory team that provides information and advice on the prioritization of restoration projects in Anchorage. In this capacity, it would be difficult for the team to implement steps toward increased robustness. The good news is that the Task Force is seeking to upgrade its status to a municipal board. If the Task Force formalized its existence as a board within the municipal structure, it could assume an increased role in watershed management and create more opportunities for multiagency involvement in decisionmaking processes.

As a municipal board, the team could establish a broad vision for Ship Creek as well as other creeks within the Municipality and use this vision to work toward increased robustness. The specific steps within the general goals of creating a proportional equivalence of benefits and costs, developing collective-choice agreements, and increasing user and biophysical monitoring could become milestones on the way to a more robust Ship Creek (Figure 2).

The popular Lower Ship Creek Fishery can demonstrate the robust management of an engineered fishery. A robust fishery has the ability to take pressure off other wild stocks while creating a sense of ownership within the greater community. Anchorage managers have a great opportunity to save time and money by robustly managing this engineered urban fishery for the thousands of people who wander down to the banks of Lower Ship Creek each summer. It is hoped that other managers will learn from the opportunities derived from this case study to increase the robustness of other creeks throughout Alaska and the world.

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List of Abbreviations

ADEC	Alaska Department of Environmental Conservation
ADFG	Alaska Department of Fish and Game
ARRC	Alaska Railroad Corporation
AWC	Anchorage Waterways Council
EAFB	Elmendorf Air Force Base
EFH	essential fish habitat
EPA	U.S. Environmental Protection Agency
FRAB	Fort Richardson Army Base
HOT	highly optimized tolerance
KAPP	Knik Arm Power Plant Dam
MOA	Municipality of Anchorage
NEPA	National Environmental Protection Act

NMFS	National Marine Fisheries Service
SES	social-ecological system
TAC	total allowable catch
TMDL	total maximum daily load
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service
USGS	U.S. Geological Survey

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Carbon, Nitrogen, and Agriculture

Improved Nitrogen Management Utilizing Ground-Penetrating-Radar: A Nine-Year Investigation

Timothy Gish, Craig Daughtry, Andy Russ, Lynn McKee, John Prueger

Abstract

Water availability and efficient use of nitrogen are critical components of a sustainable and profitable agricultural system. Since nitrogen is typically excessively applied, considerable amounts of nitrogen may leach to and move through the subsurface to surface water resources. Our hypothesis is that knowledge of the subsurface hydrology can be utilized to reduce nitrogen applications by identifying where pathways serve as a subsurface irrigation system. This research was conducted at the U.S. Department of Agriculture–Agricultural Research Service's Optimizing Production Inputs for Economic Enhancement site in Beltsville, MD. In this study, two corn production fields about 4 ha each were studied over 9 years to evaluate nitrogen use efficiency with and without a knowledge of the subsurface water flow pathways determined using primarily ground-penetrating radar (GPR) and digital surface elevation maps. Since the depth to the GPR-identified subsurface water flow pathways typically varied along the pathway, the site has both capillary and lateral flow components. Field B received uniform applications of nitrogen with 34 kg N/ha applied at planting and then about 134 kg N/ha as sidedressing when the corn was about 60–80 cm high. The second field, field D, was under precision management receiving 34 kg N/ha at planting and then 0–134 kg N/ha as variable-rate sidedressing (same time as field B). The amount of N applied in field D, for each 8 × 8 m unit area, at sidedressing depended primarily upon the location and characteristics of the subsurface water flow pathways. Knowing where the subsurface flow channels existed allowed us to apply less N downslope along the subsurface flow pathways where they approached the surface. As a result, the precision N site generally received about 34 percent less nitrogen than the uniform N application site, yet there was no significant reduction in yields. This work demonstrates that knowledge of the subsurface hydrology can improve nitrogen use efficiency and thereby increase farm sustainability.

Keywords: hydrology, crop yield, precision farming

Introduction

Sustainable agriculture is typically defined as an integrated system of animal management and crop production practices that in the long term enhance environmental quality and sustain natural resources while maintaining productivity. Although nitrogen is frequently overapplied, Keeney and Deluca (1993) showed that considerable N loss was occurring on agricultural land well before the widespread use of inorganic fertilizers. As a result, some N loss will occur regardless of agricultural management strategies. However, by improving water quality and productivity as goals, the concept of "sustainable agriculture" will require innovative solutions

Nitrogen is frequently overapplied as insurance against low yields, and that poses a risk to surface and groundwater quality. Jaynes et al. (2001) observed in tile drains (installed at 1.45 m) that nitrate leaching increased with increasing N application. In the lowest level of N applied (57–67 kg N/ha), nitrate losses in the tile drain were typically above the U.S. Environmental Protection Agency maximum contamination level of 10 mg NO_3-N/L. Unfortunately, at these lower N

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application rates, corn yields suffered and were significantly lower that the medium and high rates of N applied.

Including a nonlegume winter cover crop into the production management system has been found to reduce nitrate leaching since the cover crop reduces percolation to groundwater and utilizes nitrate that would otherwise leach to groundwater (Francis et al. 1998, Shepherd 1999). Unfortunately, when cover crop planting is delayed due to poor weather conditions the cover crop may not become established well enough to absorb significant quantities of soil water and N (Shepherd 1999). Furthermore, growth of the succeeding crop may be reduced if the cover crop has a high C/N ratio (Francis et al. 1998).

A growing crop management strategy to reduce agrichemical pollution is site-specific or precision farming. Historically, farmers have treated a field as a single unit applying the same nutrient practice to the entire field. However, since soils are inherently variable, a field can typically be divided into discrete areas, with each area managed according to its needs rather than by field averages. Birrell et al. (1996) found that much of the crop yield variability could be explained by the depth to the claypan layer. Depth to the claypan was readily measured using electromagnetic techniques (Sudduth et al. 1995), and nutrient management plans developed on the basis of claypan depth since this feature strongly influences soil water relationships for the crop (Kitchen et al. 1997).

To develop an accurate nutrient management plan, it would be useful to have reliable estimates of water and chemical movement through and along subsurface layers. Traditional methods for assessing the spatial nature of hydraulic properties include the collection of soil core and well log data (Sudicky 1986, Ritzi et al. 1994). These methods are of limited benefit for large fields as only a fraction of the field can be reasonably sampled. Additionally, it is virtually impossible to ascertain the spatial behavior of water and chemical movement using point data because the sampling density of soil core and well log data is considerably below the inherent spatial variability of soil hydraulic properties. As a result, uncertainty associated with estimating water movement where no samples were acquired can be significant.

Soil layers can significantly affect water movement and chemical transport because abrupt changes in texture or density across the boundary of two adjacent layers causes a discontinuity of soil pores. Research has shown that this mismatch of pore entry value and soil hydraulic conductivity can trigger funnel flow (Kung 1990 a and b, 1993; Ju and Kung 1993). Under this condition, uniform matrix flow could converge and form discrete subsurface preferential flow pathways, especially when these soil layers are inclined. Accordingly, Gish et al. (2002) identified subsurface restricting layers (typically a clay lens) using ground-penetrating radar (GPR). The depths to these restricting layers were evaluated and subsurface flow pathways identified. The subsurface flow pathways were then used to show that during a drought year, yields decreased with distance from the GPR-identified subsurface flow pathway (Gish et al., 2005). These studies indicate that with a knowledge of the subsurface stratigraphy a nutrient management plan could be developed that could reduce N inputs without a significant reduction in productivity.

The objective of this research was to determine if a multiyear nutrient management plan can be developed using knowledge of the subsurface hydrology constructed primarily with GPR data.

Methods

Site Description

The research site is a 21-ha agricultural production farm located at the U.S. Department of Agriculture Henry A. Wallace Beltsville Agricultural Research Center in Beltsville, MD (near 39°01'44" N., 76°50'46" W.). A variety of data including general soil properties, crop parameters, and geophysical, meteorological, and remotely sensed data are acquired annually on this site, which is identified as OPE3: the Optimizing Production Inputs for Economic and Environmental Enhancement site. One of the principal objectives of OPE3 is to determine field and catchment-scale fluxes of agricultural inputs. The site contains four fields that range from 3.6 to 4.2 ha, each draining into a 1st-order stream and riparian wetland and each delimited with earthen berms. The soils are variable but mostly sandy, with the majority being Typic Hapludults, coarse-loamy, siliceous, mesic, and are well drained. The surface soil textures range from sandy loam to loamy sand, have an average organic matter content of <3 percent.

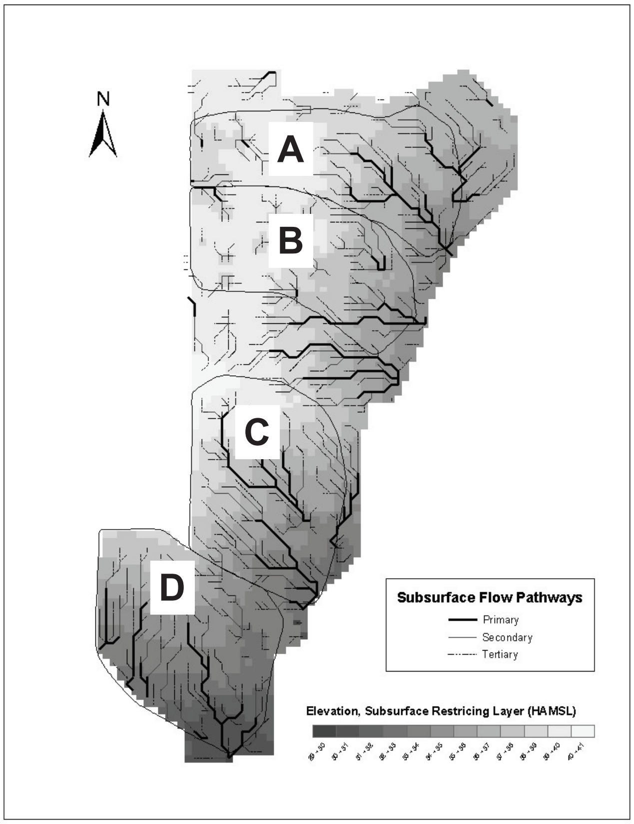

Figure 1. Depth to the subsurface restricting layers and location of GPR-identified subsurface flow pathways.

Subsurface Flow Pathways

OPE3 field boundaries, subsurface restricting layer elevations, and the corresponding GPR-identified subsurface flow pathways are shown in Figure 1. Specifically, a subsurface interface radar system-2, with a 150-MHz antenna (Geophysical Survey Systems Inc., North Salem, NH) was used to identify subsurface reflections that could represent the depth of soil layers restricting vertical water movement (i.e., clay lenses). Over 40 km of ground-penetrating radar data were acquired for the OPE3 site, and a digital trace of the subsurface reflections were produced using RADAN Software (1999, Geophysical Survey Systems Inc.). The spatial autocorrelation of these subsurface reflections that restrict water movement for the entire research site were determined using GEO-EAS (1991, U.S. Environmental Protection Agency) and GS$^+$ (2001, Gamma Design Software) geostatistical software packages. To determine the elevation of the subsurface restricting layer, the depth of these subsurface reflections was averaged over each 8×8-m cell and was subsequently subtracted from a digital elevation map (DEM) averaged over the corresponding 8×8-m cell. The DEM was developed by analyzing real-time kinematic global positioning system (GPS) data acquired on a 5-m grid over the entire research site (Trimble, Sunnyvale, CA).

The subsurface restricting layers that have been identified with GPR reside between 1 and 4 m below the soil surface (Gish et. al. 2002). However, groundwater above these soil restricting layers can be much shallower. Thus, although the average restricting layer depth at this site is at a depth of 1.5 m, the water table may be well within 1 m of the soil surface. Although Gish et al. (2005) demonstrated that averaged corn grain yields decreased with increasing distance from the subsurface flow pathways (during a drought year) they also showed that there were areas where the restricting soil layers (and water above them) were too deep to influence soil water contents and crop yield. Additionally, since the subsurface flow pathways are three-dimensional, the depth to the restricting layer varies along the length of the GPR-identified subsurface flow pathway. Depressions along the GPR-identified subsurface flow pathway are common, and these depressions form cascading pools of water when the pathways are actively flowing (Figure 2).

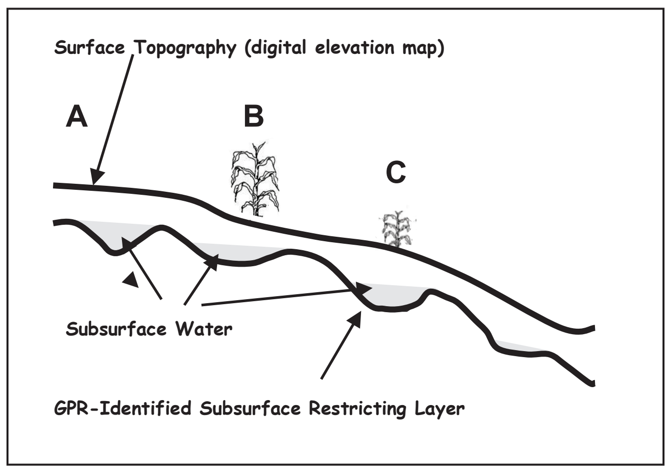

Figure 2. Schematic of a GPR-identified subsurface flow pathway with typical variations in elevation and formation of localized pools.

If there is no flow along the subsurface pathways are then water that has accumulated previously within these localized pools will behave as a local perched water table. As a result, the GPR-identified subsurface flow pathways have both lateral flow and perched water table components.

Figure 2 is a schematic that depicts the effect of the subsurface on corn growth during a drought year. Region A is at the top of the field were the subsurface flow pathways are initiated. Region B represents areas within the field were the subsurface restricting layers have formed pools of subsurface water that may provide water and nutrients to the crop root zone. Region C denotes areas within the field that are located near a subsurface flow pathway but where the pathways are too deep to affect plant growth and yield. Additionally, as more water is likely to drain into a specific GPR-identified pool, less N would need to be applied as side dressing`. In this study, The ARC-GIS (2002, ESRI) hydrologic modeling tools FLOWDIRECTION and FLOWACCUMULATION programs were applied to the raster grid of elevation corrected subsurface topography to determine the amount of water draining into each GPR-identified pool within the field.

Nitrogen Applications

For each of the 9 years, 34 kg N/ha were banded in fields B and D during planting of corn (*Zea mays* L.). The amount of N applied during side dressing varied from year to year. During the first two years (1998 and 1999), fields B and D received the same N treatment with 134 kg N/ha applied at side dressing (see Table 1 for total N inputs). Beginning in 2000, fields B and D received different N treatments with B representing uniform applications of N and field D the precision N treatment. Side dressing of N generally occurred 4–5 weeks after planting. No data is shown for 2003 because the crop was destroyed during Hurricane Isabelle.

Table 1. Nitrogen application rates for each year.

Year	Total N applied (kg N/ha)		Sidedressing N (kg N/ha)	
	Field B	Field D	Field B	Field D
1998	168	168	134	134
1999	168	168	134	134
2000	168	140	134	107
2001	140	98	106	98
2002*	103	80	69	46
2004	172	109	138	75
2005	164	121	131	87
2006	164	121	131	87
2007	164	121	131	87

*No data for 2003 are shown as the crop was totally destroyed by Hurricane Isabelle.

The pre-side dress nitrate test (PSNT) was used from 2000 to 2004 in field B. As a result, field B received 34 kg N/ha at planting and the PSNT determined the side dressing N rate (Meisinger et al. 1992). Starting in 2005, however, PSNT values were no longer acquired and a constant side dressing rate of 131 kg N/ha was applied in field B.

Nitrogen side dressing on field B occurred on the same dates as field D for all years. From 2000 to 2004, field D received a side dress N prescription based on the subsurface hydrology and PSNT values. For example, in areas corresponding to regions A and C in Figure 2, the N rate prescribed by the PSNT values were applied. However, in regions within the field that correspond to region B in Figure 1, no nitrogen was applied at side dressing, regardless of PSNT values. Beginning in 2005, PSNT values were no longer acquired, so the N side dressing rate was determined by subsurface hydrology alone. Regarding the subsurface hydrology, the amount of N applied at sidedress was determined in an algorithm that accounted for several factors: (1) the proximity of the nearest GPR-identified subsurface flow pathways; (2) the depth to these subsurface flow pathways; and (3) the amount of land draining into the GPR-identified pools (depicted in Figure 2). Briefly, using Figure 2 as an example, areas within the field that correspond to region A, where the subsurface flow pathway initiated (no convergence of subsurface flow), or region C, where the pathways were located too deep (>3 m), the highest rate of about 134 kg/ha was applied at side dressing. In general, as the regions within the field approached a subsurface flow pathway (vertically or laterally), N application amounts were reduced linearly until the within-field region was within 1 m of the subsurface flow pathway, indicating that no N would be applied at side dressing.

Yield Monitoring

Corn grain yields for all 8 years were acquired with a yield monitor (AgLeader 2000, Roswell, GA) interfaced with a differential GPS. Yield data were processed to eliminate measurement errors resulting from harvester detours around field instrumentation and other obstacles. The spatial autocorrelation of corn grain yields were determined using GEO-EAS packages. To make direct comparison of yield data to the GPR-identified flow pathways, the GEO-EAS (1991, U.S. Environmental Protection Agency) and GS^+ (2001, Gamma Design Software) geostatistical software yield data was then kriged at 8×8 m. In general, corn grain yield values were collected every 1.4 m in the row direction.

Results

Field Comparisons

Fields B and D have nearly identical soil textures in the top 0.3 m with an average sand content ranging between 61 and 63 percent. Surface slopes are about 1 percent greater in field D than field B, but the depth of the restricting layer (orthogonal to the soil surface) is similar, about 1.5 m.

For the first two years of this study, fields B and D received the same tillage and agrichemical treatment and as such generated similar yield responses. In 1998, both fields generated a corn grain yield of 3.8 Mt/ha. Although 1998 had precipitation below normal, 1999 was a severe drought (Table 2). As a result, field B generated a corn grain yield of 1.3 Mt/ha while field D generated a yield of 1.5 Mt/ha. Since soil properties, depth to the subsurface restricting layers, and even yields over two years are nearly identical, a direct comparison of the subsequent yield data should be appropriate.

Weather conditions varied a great deal for the next eight years, with total rainfall amounts (from planting to plant senescence) ranging from 0.6 to 3.3 m. During this time, yields in the uniform N treatment varied from 3.4 to 7 Mt/ha. Meanwhile, corn grain yield varied from 3.6 to 7.4 Mt/ha for the precision N field. Figure 3 compares the corn grain yield in both fields for all years except 2003, which was never harvested due to being destroyed during Hurricane Isabelle.

Statistical analysis revealed that there was no significant difference in corn grain yields between the two fields even though the precision N field (D) received much less N at sidedressing. Although both field received 34 kg N/ha at planting, averaged sidedressing on field B was 120 kg N/ha compared to only 79 kg N/ha on field D. As a consequence, field D received about 41 kg/ha less N with no significant reduction in yield, a reduction in sidedress N of over 34 percent.

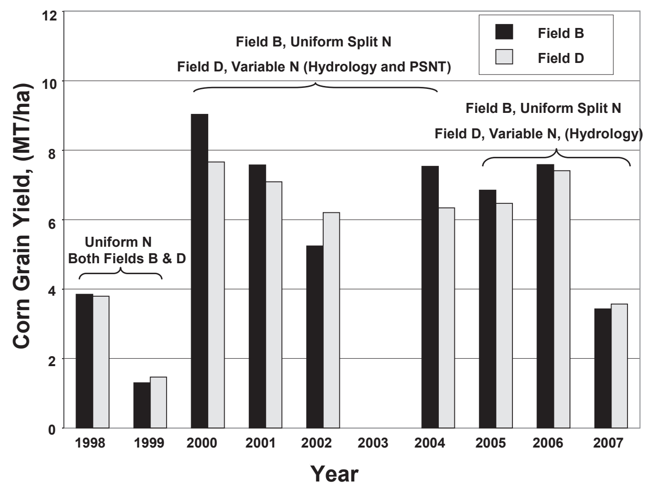

Figure 3. Yield comparison between uniform side dressed N (field B) and precision side dressed N (field D).

Table 2. Rainfall and yields.

Year	Rainfall* (m)	Corn grain yield (Mt/ha)	
		Field B	Field D
1998	1.1	3.8	3.8
1999	0.6	1.3	1.5
2000	2.9	9.0	7.7
2001	3.1	7.6	7.1
2002†	1.6	5.2	6.2
2004	3.3	7.5	6.3
2005	2.6	6.8	6.5
2006	3.4	7.6	7.4
2007	1.0	3.4	3.6

*Rainfall reported here was measured from the day of planting until plant senescence.

†No data for 2003 are shown as the crop was totally destroyed by Hurricane Isabelle.

Conclusions

Corn grain yields from two nitrogen application treatments were compared for 9 years at the OPE3 site in Beltsville, MD. The uniform N treatment received 34 kg N/ha starter N at planting and the bulk of the N at sidedressing, 4–5 weeks after planting. The precision N treatment also received 34 kg N/ha starter N at planting, with the bulk of the N applied at sidedressing. However, in the precision treatment, the sidedressed N was determined primarily as a function of subsurface hydrology. Corn grain yields varied significantly over the nine years, in part because of the amount of rainfall received.

Over the nine years there was no difference in corn grain yields between the two treatments even though the precision N treatment received 34 percent less N at sidedressing. This research indicates that knowledge of the subsurface hydrology determined with ground-penetrating radar can be useful in reducing N inputs without having a detrimental effect on yields.

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Water, Energy, and Carbon Flux Observations from Agricultural Research Service Watersheds and Agro-Ecosystem Experimental Sites

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Abstract

Several Agricultural Research Service watershed locations and long-term experimental/monitoring sites have been measuring water, energy, and carbon fluxes using eddy covariance techniques. Several sites have been collecting flux data for the last 5–10 years, while other locations have only recently started a monitoring program. The measurement sites from east to west include corn in Beltsville, Maryland, at the OPE3 watershed; pasture and switchgrass fields near State College, Pennsylvania; corn and soybean rotation in the Walnut Creek and South Fork watersheds near Ames, Iowa; corn and soybean rotation near St. Paul, Minnesota; grassland near Mandan, North Dakota; grassland and shrubland sites at the Jornada Experimental Range near Las Cruces, New Mexico; riparian, grassland, and shrubland in Walnut Gulch watershed and San Pedro river basin near Tombstone, Arizona; grassland and savanna sites in the Santa Rita Experimental Range near Tucson, Arizona; a riparian site near Reno, Nevada; and high elevation shrubland and forest sites in Reynolds Creek watershed near Boise, Idaho. This presentation provides an overview of the measurements conducted at these sites and comparisons of energy flux partitioning, water use, and net carbon exchange during the growing season. In addition, there will be a discussion of possible future multi-location research projects and inter-comparison studies involving the eddy flux measurements and ancillary data.

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Integrating Watershed- and Farm-Scale Models to Target Critical Source Areas While Maintaining Farm Economic Viability

Lula T. Ghebremichael, Tamie L. Veith

Abstract

Nonpoint source pollution from agriculture and the effects best management practices for mitigation are commonly evaluated based on hydrologic boundaries using watershed models. However, management practice effectiveness is affected by which of the feasible practices are actually selected, implemented, and maintained. It is increasingly recognized that alternative management practices to mitigate nutrient losses from agricultural watersheds are applied at the field and farm levels and are usually selected and maintained at the farm level. To be successful, watershed- and farm-scale models must be combined in such a way that environmental concerns, such as identification and mitigation of critical source areas, are addressed while farm production systems are maintained or improved. This study develops a modeling framework for integrating watershed- and farm-scale models that is based on experience from numerous location-specific studies at both scales in the northeastern United States.

Keywords: critical source area, model scale, net profit, phosphorus balance

Introduction

Targeting critical source areas (CSAs) of pollution for best management practices (BMPs) is important in successfully controlling nonpoint source pollution (Walter et al. 2000, McDowell et al. 2001, Weld et al. 2001). Critical source areas are relatively small proportions of a watershed that contribute disproportionately high pollutant loads to nearby streams (Gburek and Sharpley 1998, Pionke et al. 2000). Many studies have demonstrated the potential of watershed-based simulation models and geographic information systems (GIS) in assessing pollutant losses from CSAs and associated BMP effectivenesses (Zollweg et al. 1996, Secchi et al. 2007, Busteed et al. 2009). Watershed-scale models, simulating pollutant transport to water bodies, commonly involve representations of complex watershed systems based on hydrologic boundaries. These models use input data of physical landscape properties taken from geodatabases (e.g., digital elevation models [DEMs], soil maps, property lines, and land cover data). Despite the environmental potentials for watershed-scale tools to aid in targeting, these tools are limited in practical application by conservation specialists for on-farm CSA delineation and BMP targeting. A few of the challenges can be mentioned.

First, information acquired from these tools is not easily transferred into simplified forms suitable for interpretation by conservation specialists involved with practical aspects of pollution control. Transferring CSA-related findings obtained from hydrologic-scale models to individual farms in a multi-farm and multi-owner watershed remains particularly challenging. Second, BMP evaluations made by watershed models are based primarily on environmental performance, without considering economic and environmental feasibility at the farm-system level. Third, most watershed model tools lack a detailed representation of farm system changes (i.e., labor, resources, and animal feed availability) that are core influencers of farm sustainability and water quality conditions.

To use these tools for practical applications in delineating CSAs and targeting on-farm BMPs, a framework is needed that (1) transfers CSA-related findings obtained from complex models into forms that can be applied at a field level and then into farm-level plans, (2) evaluates farm-scale CSAs and associated mitigation measures with regard to their feasibility and

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economic aspects, and (3) incorporates farm systems and farm-level plans into watershed-based tools to assess their effects on the quality of larger water bodies. This study discusses and demonstrates a comprehensive modeling framework that meets these needs by integrating watershed- and farm-level assessments. Integrating the two scales allows environmental concerns to be addressed while the farmers' production systems are maintained or improved. The framework is demonstrated by combining portions of previous studies from the northeastern United States that used the Soil and Water Assessment Tool (SWAT, Neitsch et al. 2002) and the Integrated Farm System Model (IFSM, Rotz et al. 2011).

Modeling Framework

The modeling framework (Figure 1) aids in the design of environmentally targeted BMPs in agricultural watersheds such that they are also economically viable at the farm level. The watershed-level assessments are driven by the environmental quality goal, while the farm-level assessment is primarily driven by the farm sustainability goal. The framework applies a tiered approach in identifying CSAs at a watershed scale and in planning targeted BMPs at both watershed and farm scales. In task 1, CSAs of pollution that should receive higher priority for potential BMPs are identified. Then, in task 2, potential BMPs needed to treat these CSAs are evaluated at a watershed scale with respect to their potential for preventing water pollution. Both tasks use a hydrologic watershed-level water quality model such as SWAT. CSA data obtained from the watershed model are simplified into a form applicable at the field-level and then into farm-level plans. When modeling is performed with known field boundaries and land ownership, CSAs can be specifically located, and key land owners can be encouraged to participate in targeted nonpoint source pollution control programs. Otherwise, CSA characteristics identified from the model can be developed as a reference; then, by performing field surveys or farm-by-farm assessments, farm fields can be checked against the reference for similarity in characteristics. Once these farm fields are identified, task 3 applies a whole-farm model, such as IFSM, to assess farm-specific feasibility and the economic and environmental effects of implementing the CSA BMPs identified in task 2. Task 3 also incorporates a farm-level assessment that includes farmers' inputs in the process of planning effective CSA BMPs. Finally, task 4 goes back to the watershed level to evaluate the collective effects of farm-level BMPs on the water quality of streams and water bodies fed by the watershed. In addition, some farm factors that are important at a local farm scale (such as labor availability and animal rations), but that are not easily represented at the watershed scale where broader hydrologic processes are modeled, are reevaluated for their effect on watershed-level water quality by integrating farm-scale results across farms.

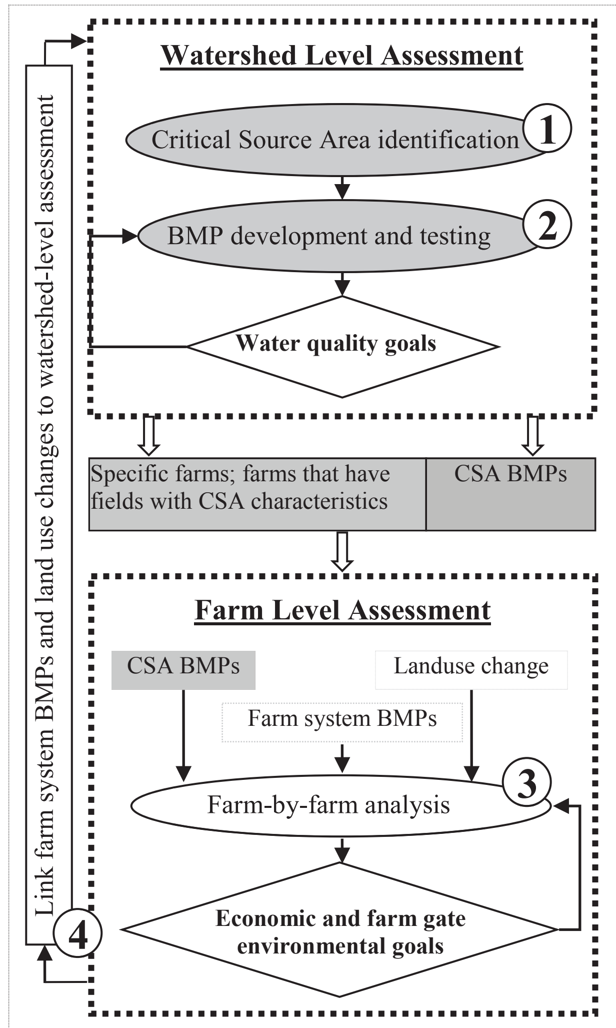

Figure 1. Modeling framework integrating watershed-level (shaded) and farm-level (unshaded) assessments.

Integrating watershed- and farm-level assessments ensures not only that targeted water pollution preventive managements planned at a watershed scale can be linked to farms, but also that farmers' management decisions and BMPs can be tied more directly to downstream pollution of the streams and water bodies. The modeling framework is a comprehensive system approach that incorporates multiple objectives of water quality and farm profitability at appropriate hydrologic and management scales. It can be used to guide implementation of targeted strategies that are both environmentally and economically sustainable.

Typical Application

The modeling framework demonstration draws from previous studies of a 163-ha watershed that encompasses a single 102-cow dairy farm (R-farm). Various watershed- and farm-scale modeling studies have been done in the R-farm watershed to address phosphorus (P) related water quality problems while maintaining the farmer's economic viability.

Task 1: Identify CSAs for Targeting

When using SWAT to model a watershed, the hydrologic response units (HRUs), which are composed of distinct soil, land use, and slope combinations, become the building blocks of the CSAs. HRUs must be transferred to the field level to be used for practical application because the field is the smallest scale recognized by farmers and planners at the ground level. Gitau and Gburek (2005) provide an example of the most direct way to link HRU predictions to specific fields. In SWAT, they used field-distinct land use and detailed field-level management input data to represent the R-farm watershed. With each field uniquely coded and represented by several HRUs, they then calibrated SWAT with respect to streamflow, sediment, and total P (TP) losses and identified variable TP losses by HRU within the fields (Figure 2A).

In this paper, field-by-field based average TP losses were estimated by calculating area-weighted averages of the HRU-based TP losses within each field (Figure 2B). Because these model predictions are field specific and ownership of the fields is known, these predictions can be directly used in planning targeted remedial strategies for this farm. However, when crop fields are in rotation, the rate of TP loss from a particular field may vary from year to year depending on the type of crop and associated management (see graphs for 1993, 1994, 1995 in Figure 2). Therefore, it is important to recognize these year-to-year spatial and temporal variations when interpreting model outputs for practical use.

In many cases, watershed modeling efforts involve land use input data that may not be field specific and (or) management input data that reflect typical practices obtained from extension personnel or other agencies working with farmers. Even when field-specific data are available, the number of fields in larger watersheds may be too large to identify the owners explicitly and represent them in the model. In these cases, it may be necessary to extract important information from modeling outputs regarding specific landscape characteristics, including land use, soil types, and slope, that are likely to result in CSAs. When farm fields and land uses with characteristics similar to these modeled CSAs are identified, they can be selected as priority fields for further analysis and targeting.

Task 2: Evaluate Watershed-Level BMPs

Once priority CSAs for targeting are identified, the next step is to assess and prioritize BMPs needed to remediate each CSA based on the BMPs relative effectiveness toward meeting pollution reduction goals. For example, when a no-till management practice was imposed on all corn and alfalfa fields from 1993 to 1995, SWAT predicted a 15 percent reduction in TP losses compared to the baseline condition (no-till; Figure 3). In addition, the watershed-level SWAT prediction for converting corn to grass (Ghebremichael et al. 2008) reduced TP losses by 9 percent from the baseline (no-corn; Figure 3). This process of BMP analysis continues with as many individual BMPs and combinations as possible until the target water quality goal is achieved.

Task 3: Evaluate Farm-Level CSAs and BMPs

For environmentally-sound measures to be potentially implemented by farms, they have to be feasible for both their practical and economical aspects. Task 3 uses IFSM, a farm-scale model, to assess how BMPs designed at the watershed level affect different factors of the farm production system and its profitability. This analysis should be performed on selective farms identified as critical in Tasks 1 and 2.

Strategies evaluated at a watershed level for their environmental benefits need to be reevaluated at the farm level. For example, a farm-level evaluation of the watershed-level strategies of no-till corn fields and of converting corn land to grass production was developed for the R-farm, which is the only farm within the R-farm watershed. Using IFSM, Ghebremichael et al. (2007) predicted negative economic consequences to the R-farm when corn land was converted to grassland. Although the strategy reduced TP loading at the watershed level, IFSM predicted a \$68/cow/yr decrease in farm profit (Table 1).

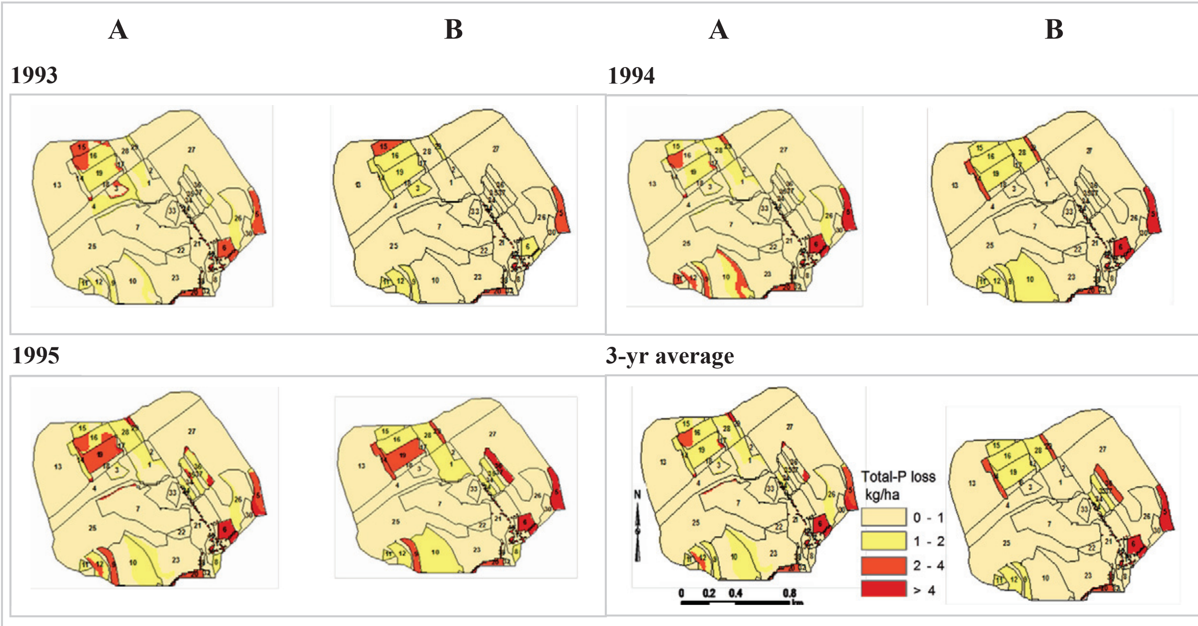

Figure 2. Predicted average total-P losses for the R-farm watershed (A) by hydrologic response units (HRUs) as output by SWAT (Gitau and Gburek 2005), and (B) by fields, calculated as the HRU weighted averages.

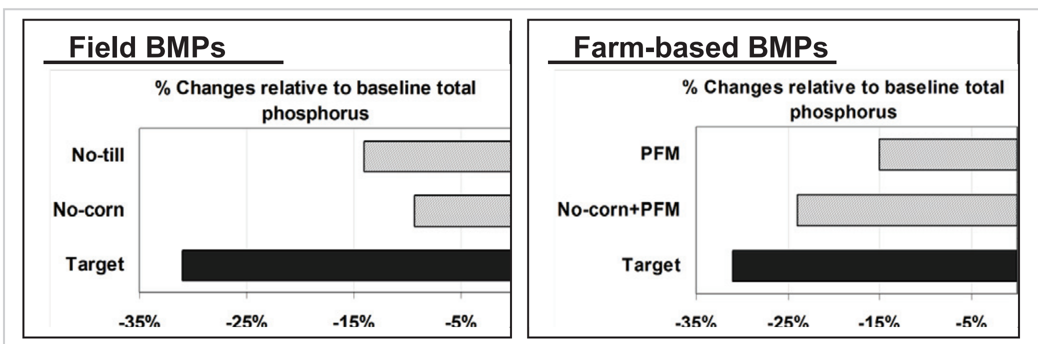

Figure 3. SWAT-predicted effectiveness for two management practices for the R-farm watershed compared with a 31 percent target phosphorus reduction from the baseline. (PFM, or Precision Feed Management, reduces dietary phosphorus and increases forage productivity and utilization.)

Table 1. IFSM-predicted outputs for a baseline scenario and alternative management scenarios for the R-farm.

	Baseline[2]	Change in value[1] as compared to baseline			
		No-corn[2]	No-till[3]	PFM[2]	PFM + No-corn
Environmental aspects of the farm					
P balance, kg/ha	9.6	+1.8	0	–9.6	–9.8
Economic aspects of the farm, $/year/cow					
Milk and animal income	3,318	0	0	0	0
Total production cost	2,880	+68	–43	–237	–253
Cost of production	2,152	–43	–43	+98	+64
Cost of purchased feed	728	+111	0	–335	–318
Farm net return	438	–68	+43	+237	+253

[1]change in value = alternative scenario value – baseline scenario value

[2]data from Ghebremichael et al. (2007); No-corn scenario changed all corn land to grass; PFM (Precision Feed Management) reduced dietary phosphorus and increased forage productivity and utilization

[3]data from Ghebremichael et al. (2009); No-till scenario removed the conventional tillage from the corn land

The predicted reduction in farm profit was due to an increase in purchased animal feed as more feed energy was required to offset reductions in on-farm corn silage production. Also, as more P was brought onto the farm in the increased feed purchases, the predicted P surplus increased slightly.

However, switching from conventional to no-till corn (Ghebremichael et al. 2009) slightly increased the farm's net return compared to the baseline level by lowering production costs, including fuel, tillage equipment, and labor (Table 1). Because Task 2 also predicted that the no-till practice would reduce TP losses from the watershed, the no-till BMP positively addresses both the water quality and farm economic goals. Conversely, some management solutions planned at a watershed level, such as conversion of corn land to grass production, may have negative farm-level consequences. Such negative consequences are likely to hinder successful BMP adoption by farmers unless compensatory changes can be made elsewhere on the farm.

For example, as demonstrated by Ghebremichael et al. (2007), strategies of increasing forage productivity and utilization in animal diet and reducing dietary P can be added to the strategy of converting corn land use to grass production in order to lessen the negative economic effects and address the farm's impending P imbalance. With these strategies combined, Ghebremichael et al. (2007) predicted increased farm net return as the farm utilized more on-farm produced forage and reduced purchased dietary P supplements (Table 1). As the farm used more on-farm produced feeds and less purchased feeds, the P imported through feed also decreased, resulting in a reduced farm P surplus, which is a potential root cause for soil P buildup and subsequent loss in runoff.

Task 4: Reevaluate Farm-Level BMP at the Watershed Scale

Finally, combined effects of farm system and (or) farm-level land use changes on watershed-level water quality should be assessed using watershed-scale tools. Data from farm-level modeling can also be used to supplement inputs to watershed models. For example, as demonstrated by Ghebremichael et al. (2008) through SWAT modeling, farm-level changes that increased forage productivity and decreased dietary P levels were predicted to be environmentally beneficial by reducing P loss at the outlet of the R-farm watershed (PFM in Figure 3).

In the same study, when these strategies were complemented with the strategy of converting corn land use to grass production, SWAT reaffirmed a positive environmental effect by predicting reduced P losses at the watershed outlet (Figure 3). To use SWAT in evaluating farm system changes that are not directly included in the SWAT processes, information resulting from IFSM simulations were used. For example, IFSM was used to determine the change in P content of manure as a result of altering dietary P in cow feed. Then, changes in dietary P in SWAT were modeled by representing the consequential effects in the concentrations of different P forms in the applied manure.

Task 4 of the modeling framework helps assess the expected environmental effects at a watershed level resulting from the implementation of farm-level land use changes and other BMPs. It also helps evaluate collective effects of the farmers' management decisions and BMP implementations on the water quality of downstream rivers and water bodies.

Conclusions

In this paper, a modeling framework integrating appropriate hydrologic and farm scale tools is described and demonstrated for a small watershed in the northeastern United States. The framework provides a guideline for developing and implementing targeted agricultural management strategies that are both environmentally and economically sustainable to the farmers and to the watershed. The modeling framework applies a tiered approach in identifying CSAs at a watershed scale and planning targeted measures at both watershed and farm scales.

Examples are provided for transferring CSA-related findings obtained from complex watershed models into forms that can be applied at a field level and then into farm-level plans. The integration of watershed- and farm-scale models allows an all-inclusive assessment of CSAs and associated measures for both watershed-level strategic planning and farm-level tactical management within an agricultural watershed. The integration of watershed and farm models also allows the transfer of important information across scales.

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Forested Watersheds

Long-term Forest Management and Climate Effects on Streamflow

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Abstract

Long-term watershed studies are a powerful tool for examining interactions among management activities, streamflow, and climatic variability. Understanding these interactions is critical for exploring the potential of forest management to adapt to or mitigate against the effects of climate change. The Coweeta Hydrologic Laboratory, located in North Carolina, USA, is a 2,185-ha basin wherein forest climate monitoring and watershed experimentation began in the early 1930s. Extensive climate and hydrologic networks have facilitated research in the basin and region for over 75 years. Our purpose was (1) to examine long-term trends in climate and streamflow in reference watersheds, and (2) to synthesize recent work that shows that managed watersheds respond differently to variation in extreme precipitation years than reference watersheds. In the basin and in the region, air temperatures have been increasing since the late 1970s. Drought severity and frequency have also increased over time, and the precipitation distribution has become more variable. Reference watersheds indicate that streamflow is more variable, reflecting precipitation variability. Streamflow of extreme wet and dry years show that watershed responses to management differ significantly in all but a forest with coppice management. Converting deciduous hardwood stands to pine altered the streamflow response to extreme precipitation years the most. High evapotranspiration rate and increased soil water storage in the pine stands may be beneficial to reduce flood risk in wet years, but they create conditions that could exacerbate drought. Our results suggest that forest management can mitigate extreme precipitation years associated with climate change; however, offsetting effects suggest the need for spatially-explicit analyses of risk and vulnerability.

Keywords: climate, long-term monitoring, streamflow, forest management, watershed

Introduction

Climate change projections suggest significant changes in temperature (e.g., 2–9°F, or 1–5°C) and precipitation over the next several decades (U.S. Global Change Research Program 2009). Land managers and policy makers are challenged to develop adaptation and mitigation strategies to protect and ensure long-term forest health and sustained ecosystem services. Changing climate is one among many current and potential future threats to the sustainability of forest water resources. Population growth has increased demand for clean water, and pressures from sprawling metropolitan areas, interbasin transfers, and wastewater discharge are all threats to water quality and quantity (Sun et al. 2008). Other threats and stressors include changes in land use, invasive species, and fire. Often these stressors occur simultaneously, making it difficult to distinguish the effects of one single threat on streamflow (Vose et al. 2011). Long-term watershed research can offer valuable insights into the interactions among forest stressors and streamflow, as well as management options that might help forests adapt to or mitigate the effects climate change on water supplies.

Detecting climate change effects in streamflow data is complex, since simultaneous changes in land use (e.g., urbanization and development) can occur, and the signal of the latter can be greater in magnitude than the climate change signal. Long-term data from forested watershed that have undergone little to no change in land use can provide a robust way to detect the climate change signal in streamflow. Paired watershed studies that implement forest management regimes while accounting for climate variation are also a powerful means to investigate the effect of both management prescriptions and climate change on streamflow. Both

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approaches require sufficient length of records to allow effects to be detected. Without long-term data from research watersheds where land use is either constant or well documented, climate and management trends on streamflow may be difficult to identify (Burt 1994).

Streamflow responses to climate change are strongly related to changes in local precipitation, but are less so for temperature; however, the magnitude and timing of response may differ with different forest structure and species. Regions of the United States have shown both increasing precipitation and streamflow (Genta et al. 1998, Kiley 1999, Groisman et al. 2004) and decreasing water yield (National Research Council of the National Academies 2008), requiring further investigation and data collection to confirm regional effects. These large differences may be due to the highly variable precipitation changes. A larger portion of the available research attempts to predict streamflow, water supply, or water resources in relation to precipitation changes from various climate change scenarios (Milly et al. 2005, Moreau 2007, Seager et al. 2009). These models also show variable response to climate change due to a broad range of predictions for future precipitation.

Our objectives for this paper are (1) to present long-term climate trends from the Coweeta Hydrologic Laboratory, (2) to examine streamflow patterns in two control watersheds, varying in elevation, and (3) to discuss management strategies to adapt to or mitigate the effects of climate change on forested watersheds.

Methods

Site Description

Coweeta Hydrologic Laboratory is a U.S. Forest Service Southern Research Station Experimental Forest located in the Nantahala Mountain Range of western North Carolina, USA (Figure 1). Coweeta has been the focus of watershed experimentation since 1934. The Coweeta basin is 1,626 ha; elevations range from 675 to 1,592 m. Historic vegetation patterns have been influenced by human activity, primarily through both clearcut and selective logging, the introduction invasive species (Elliott and Hewitt 1997, Nuckolls et al. 2009), and fire (Hertzler 1936, Douglass and Hoover 1988). Forests on reference watersheds are relatively mature (approx. 85 years old) oak-hickory (at lower elevations) and northern hardwood species (at higher elevations) (Elliott and Swank 2008).

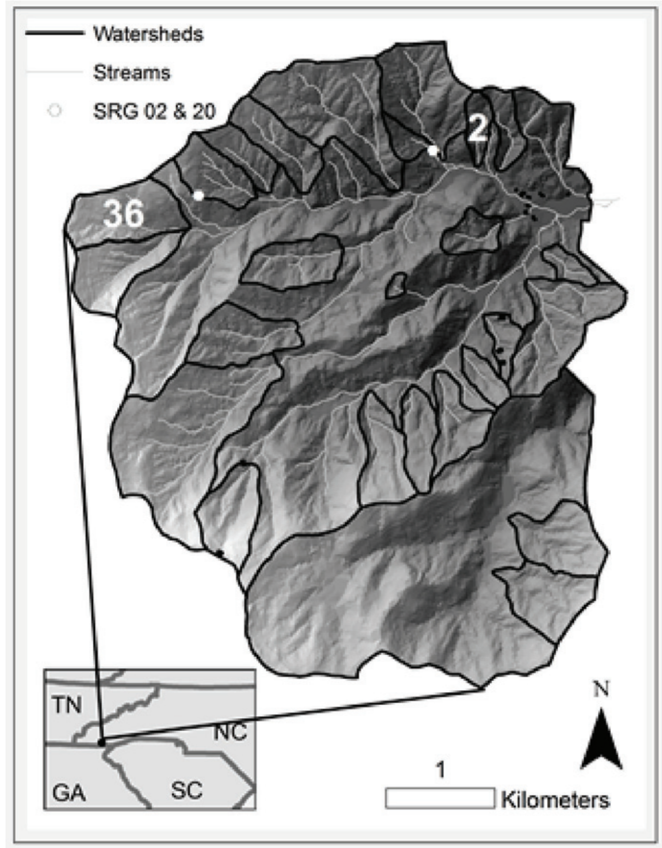

Figure 1. Map of the Coweeta Hydrologic Laboratory showing elevation gradients and main and subwatersheds. Watersheds 2 and 36 are labeled. (SRG, standard rain gauge)

Climate

Daily air temperature and precipitation have been recorded at the Coweeta main climate station (CS01) continuously since 1934. Temperature is recorded daily at 8 a.m. Eastern Standard Time using a National Weather Service (NWS) maximum, minimum, and standard thermometer. Total daily precipitation is collected in an 8-inch standard rain gauge (NWS). Recently, Laseter et al. (in review) presented the long-term trends in climate. We present a subset of those trends herein for comparison purposes.

Streamflow

To assess long-term trends in streamflow related to climate, we analyzed streamflow data from two control watersheds at Coweeta, watersheds 36 and 2 (Figure 1). The watersheds have similar aspects but varying elevations (Table 1). Streamflow data have been collected every 5 min since January 1936 for watershed 2 (WS2) and since May 1943 for watershed 36

(WS36). Both watersheds have remained undisturbed since the late 1920s, with the minor exception of a partial defoliation of WS36 by cankerworm from 1972 to 1979. Both watersheds consist of mixed hardwood forest, though the higher elevation watershed (WS36) contains northern hardwood community species.

Table 1. Characteristics of two control watersheds.

	WS 2	WS 36
Max elevation (m)	1,004	1,542
Elevation at weir (m)	709	1,021
Area (ha)	12	49
Aspect	SSE	SSE
Closest standard rain gauge (SRG)	20	02

Data from the closest standard rain gauge near each watershed were assumed to be representative of rainfall across the watershed. Standard rain gauge (SRG) 02 was used for watershed 36 and SRG 20 was used for watershed 02 (Figure 1).

We analyzed time trends in two ways. First, precipitation was regressed directly against streamflow using simple linear regression. The fit of the relationship was analyzed by looking at residuals and other possible variables of influence, including temperature (not discussed). Secondly, we calculated a runoff coefficient (RO/P) index for the two control watersheds by dividing annual streamflow (RO) by annual precipitation (P). From a simple mass hydrologic balance, streamflow output is the balance of precipitation inputs minus evapotranspiration (ET) losses: RO = P – ET. The ratio of RO to P is thus the fraction of rainfall that appears as streamflow. We used a penalized B-spline curve to analyze any possible trends in the runoff coefficient over time. Spline fit used default and custom settings for SAS 9.2, including 3 degrees, 10 control points (knots), and weight of 0 to characterize the spline curves.

Management

The interaction of management and climate was determined recently by Ford et al. (in press); they analyzed data from six paired treatment and reference watersheds throughout the Coweeta basin. They modeled the responses of streamflow to vegetation and climate. Management included species conversion, clearcuts on high and low elevation, coppice, and old field succession. We present some of the results of that study herein for comparison purposes.

Results and Discussion

Long-Term Climate

Climate data from Coweeta shows that average maximum, annual, and minimum air temperatures have increased significantly relative to the long-term mean, appearing to begin in the late 1970s (Figure 2). The rate of increase is about 0.5°C per decade beginning in the mid 1970s.

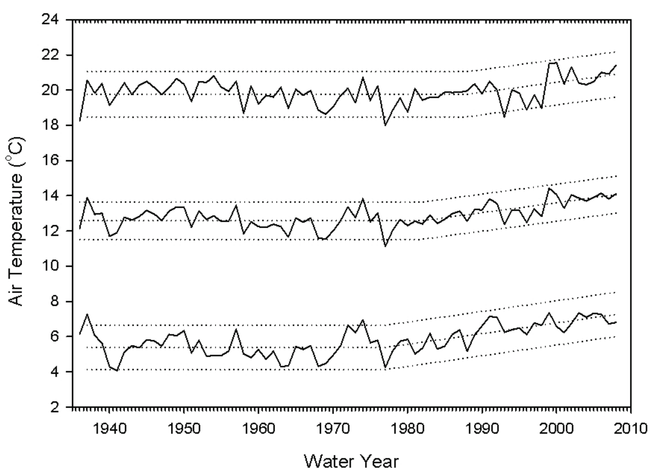

Figure 2. Long-term average maximum (top), annual (middle), and minimum (bottom) air temperatures at Coweeta Hydrologic Laboratory climate station CS01 in Laseter et al. (in review).

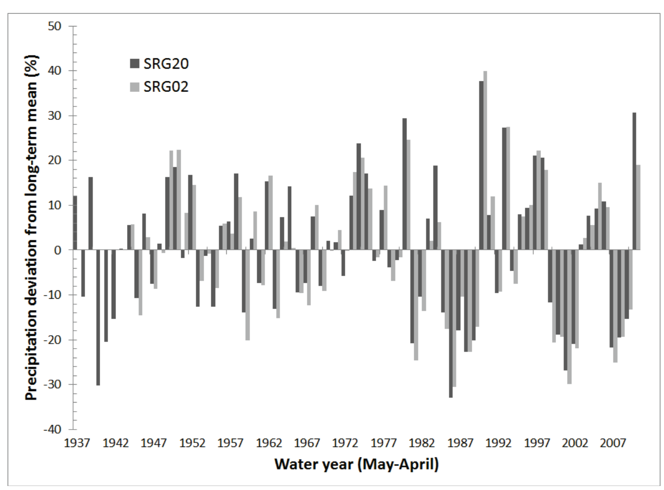

Figure 3. Deviation of annual precipitation totals from the long-term mean recorded at Coweeta Hydrologic Laboratory standard rain gauges 02 and 20.

Coweeta has some of the highest annual precipitation amounts in the eastern United States, averaging 1,794 mm/yr. Analyses of long-term precipitation suggest no significant change in mean precipitation at Coweeta (Ford et al., in press; Laseter et al., in review); however, the variability of precipitation is changing over time. For example, extreme annual precipitation (i.e., low and high rainfall) event years are occurring more frequently with time (Figure 3), which has resulted in increased recent drought severity and frequency (Laseter et al., in review).

Long-Term Streamflow

Streamflow data indicate similar trends to those found in precipitation data, including increased variability since the 1970s, largely due to the strong linear relationships between precipitation and streamflow for the two control watersheds (Figure 4). The higher elevation watershed 36 predicted streamflow or runoff is $RO'_{36} = 0.92P - 204.77$ (R^2=0.68, $p<0.01$), and the lower elevation watershed 02 predicted streamflow is $RO'_{02} = 0.71P - 626.87$ (R^2=0.78, $p<0.01$).

For any given amount of precipitation, annual streamflow on the higher elevation watershed (WS36) is at least 500 mm greater than that for the lower elevation watershed (WS2), and differences become even greater at higher amounts of precipitation. Greater streamflow per unit precipitation at the higher elevation WS36 is related to a combination of factors that reduce ET, including a shorter growing season, differences in species composition, and indirectly steep slopes and shallow soils.

Runoff coefficient analysis shows a clear upward then downward trend over time (Figure 5), which is most well defined in WS02. Simple spline curves were used for graphical display of the trends in data, which will be more completely analyzed in further study. Trend lines for both watersheds suggest decreases in the fraction of precipitation that ends up as streamflow, and hence increases in ET, over time. Our data show an increase up to the mid 1970s, followed by a leveling off or slight decrease thereafter. Decline in the runoff coefficient for WS02 is greater than that for WS36. The declining trend seems to begin in both watersheds around 1980 and may coincide with a drought that occurred at that time. More research is needed to determine the causes of this declining ratio.

Temporal variation in RO/P suggests that either biological or physical factors are changing the rainfall-runoff relationship in both WS2 and WS36. We suggest that most of this variation is due to changes in climatic driving variables and (or) structural and functional attributes that determine ET. For example, data from long-term vegetation plots indicate changes in species composition (Elliott and Vose 2011), with subsequent effects in transpiration (Ford et al., in press). Due to the nature of reference watersheds, the runoff coefficient

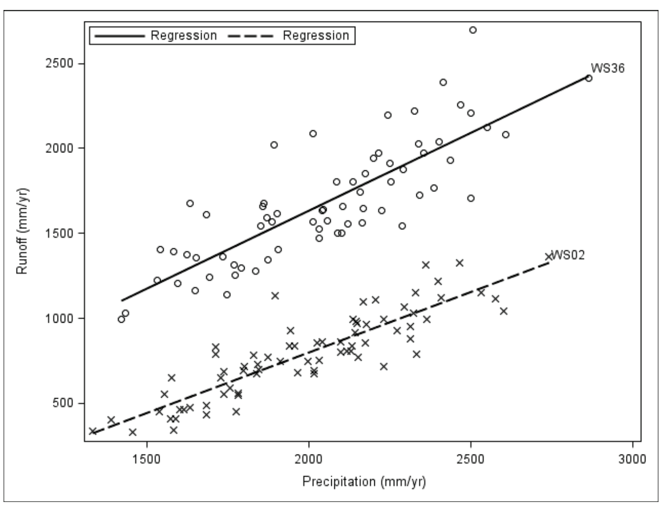

Figure 4. Precipitation versus streamflow for watersheds 36 and 2.

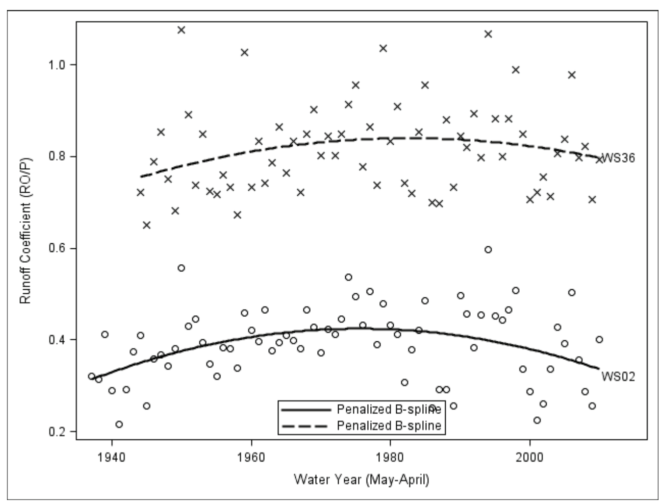

Figure 5. Runoff coefficient (RO/P) over time with penalized B-spline curves to show trends over time. RO/P increases through the first part of the period of record until about 1980, when a decrease is shown for both watersheds.

is a variation of ET adjusted for P. In an altered watershed where roads, compaction, altered flow paths and other interference factor in, the runoff coefficient then represents much more than ET.

Management Implications

In each of the management scenarios, management significantly altered the expected level of streamflow (Figure 6). All watersheds showed significant declines in streamflow excesses over time compared to what would have been expected, with most managed watersheds returning to near expected levels of streamflow within a decade.

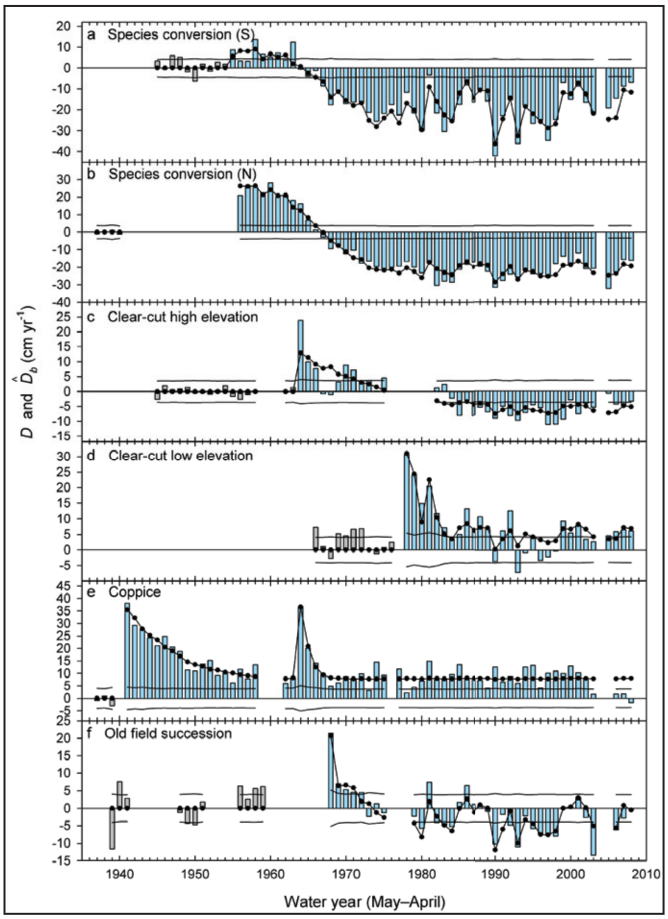

Figure 6. Streamflow response to six management treatments. Bars show observed streamflow excesses (*D*) during pretreatment (grey) and posttreatment (cyan) years. Solid lines show bounds of prediction interval. Filled symbols and lines show modeled streamflow responses ($\hat{D}_b$). From Ford et al. (in press).

Some management options, such as species conversion to pine, created persistently lower levels of streamflow than expected following canopy closure until the end of the record. A coppice forest management strategy allowed for a long-term higher streamflow than the expected levels. Other management strategies eventually returned streamflow to near those expected.

Conclusions

Long-term climatic records indicate warming and increased variability in annual precipitation over the past three decades. The combination of reference and managed watersheds provided a unique opportunity to examine streamflow responses to this variation and examine interactions between management activities and climate.

Precipitation explained a significant portion of the variation in streamflow response for the control watersheds. Runoff coefficients initially increased then declined over time, suggesting corresponding changes in ET over time. The change from an increasing trend to a decreasing trend with time coincided with drought increases in the 1980s and increasing temperature in the late 1970s.

Different forest management strategies could potentially mitigate or exacerbate effects associated with climate change. Forest management affects the vegetation structure and function of the watershed. Streamflow responses depended on the management treatment, and they could be used to mitigate climate change effects. Looking purely at water quantity shows forest management can mitigate for extreme precipitation events in a changing climate. However, these changes should be taken in context with other factors such as carbon sequestration, local climate, and water quality.

Long-term data such as those recorded at Coweeta Hydrologic Laboratory show the trends over time that can sometimes be difficult to resolve in shorter temporal datasets. When managing a forest over the long term, corresponding data collected over the time period of management is key to understanding the full scope of forest development on water resources.

Acknowledgments

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Effects of Forest Cover and Environmental Variables on Snow Accumulation and Melt

Mariana Dobre, William J. Elliot, Joan Q. Wu, Timothy E. Link, Ina S. Miller

Abstract

The goal of this study was to assess the effects of topography and forest cover resulting from different treatments on snow accumulation and melt in small watersheds in the western United States. A paired-watershed study was implemented at the Priest River Experimental Forest, ID, where ten small watersheds with an average area of 4.5 ha were treated by (1) thinning with mastication, (2) burning, (3) prescribed fire with salvage logging, (4) thinning with a prescribed fire, and (5) control (no treatment). At each watershed, a 30-m sampling grid was established for a total of 383 measurement locations. At each location, snow depth was measured between 2004 and 2010 from early February through April to characterize conditions near peak snow and during the midmelt phase. A total of 70 snow density measurements were made at several randomly selected points within each watershed and were used to determine the snow water equivalent. Forest canopy cover at each measurement location following the treatments was measured, and specific topographic variables (elevation, slope, aspect, and curvature) were derived. Correlations between forest cover, snow accumulation, snowmelt, topographic, and meteorological variables were obtained to determine the effects of these variables on snow accumulation and melt.

Keywords: snow accumulation, snowmelt, forest cover, topography, small watersheds

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Introduction

In arid and semiarid landscapes, such as the western United States, mountain snowpack represents a seasonal water storage reservoir that is the primary source of streamflow during the melt season. Both snow accumulation and melt processes are influenced by the interactions among topography, climate, and forest cover. A good understanding of these interactions is a challenge for forest hydrologists and water resource managers. Ideally, we need to manage watersheds to accumulate sufficient snowpack during the winter and slowly melt it when temperatures increase, yielding a long-duration runoff hydrograph with a low peak flow. Most managed forest watersheds receive different treatments. As a result, the snowpack dynamics within the watersheds change. As topography is constant, differences in spring runoff characteristics are mainly due to differences in the climate and forest cover. Climate is not constant and cannot be controlled; therefore, we need to understand the effects of forest management on forest cover and winter processes if we are to optimize streamflows in high-elevation forests in the western United States and elsewhere.

Currently, the processes that control snow accumulation and melt are well known, yet their interactions across varied terrain present modeling and prediction difficulties. Variables like elevation, aspect, slope, temperature, precipitation, solar radiation, relative humidity, wind speed, and canopy are known to influence snow accumulation and melt (Marks 1998, Anderton 2004, Watson 2006, Varhola 2010), but more studies are needed to determine interactions among these variables and how they change for different locations. Changes in elevation, for example, are often associated with changes in climate. Similarly, changes in canopy can change the wind patterns within the watershed and the snow cover energy balance under the canopy (Marks et al. 1998).

Varhola et al. (2010) reviewed 33 different studies from North America and Europe. Using forest cover alone as an independent variable, the authors were able to find significant relations between changes in forest cover and changes in snow accumulation and melt, yet no prediction models could be developed. The authors stated the need for studies that would measure multiple sources of variability as well as the meteorological conditions during the measurements.

In our study we evaluated a total of 33 topographic and climatic variables, including snow depth, snow water equivalent (SWE), snowmelt, and forest cover. Our main objectives were to observe (1) differences in treatment effects on SWE and snowmelt, and (2) correlations of SWE and snowmelt with 31 variables. Our study was conducted in watersheds with an elevation range of 800–1200 m using a multitude of variables in contrast to most of the current studies conducted at higher elevations or alpine regions. We anticipate our findings will contribute much needed information to the knowledge base regarding snow accumulation and melt processes.

Methods

Study Site

This research was located in ten small watersheds at the Priest River Experimental Forest (PREF), ID (48°21′24 N., 116°48′26 W; Figure 1). The watersheds have an average area of 4.5 ha and range in elevation from 843 to 1,236 m. The average slope of the watersheds is 29 percent and increases in elevation from watershed 1 to 10. Within this elevation range, up to around 1,000 m, the soils are shallow Saltese with deeper Jughandle soils only on drainage concave areas. Above 1,000 m the Jughandle soils are predominant and are characterized by a zone of maximum soil water and temperature effectiveness (McConnell 1965). The watersheds were intentionally chosen for their predominant southern aspects, and gneiss is the predominant bedrock type.

Climate

The climate is transitional between northern Pacific maritime and continental types. Average annual maximum and minimum temperatures are 14°C and 0°C, respectively, and precipitation is 805 mm based on the 2004–2010 average values measured at the PREF weather station. Snowfall depths at the same weather station were approximated into snow water equivalent values using a snow density of 100 kg/m^3. Twenty-two percent of the total average precipitation for 2004–2010 was in form of snow. The years with the largest amount of snow were 2008 (47 percent of annual precipitation) and 2007 (33 percent), while the driest years were 2005 (13 percent) and 2010 (13 percent). Year 2005 was discarded from our analysis as the snowpack did not accumulate enough to conduct the field measurements. At the study site snowfall accumulates from December through February and may persist on the ground until late April or even May, depending on the amount of precipitation received during the snow season.

Field Methods

A paired-watershed sampling design was used in this study where each pair received one or two of the following treatments in the years noted: (1) thinned (2007)/mastication (2008), (2) burned (2006), (3) burned (2006)/salvage (2007), (4) thinned (2007)/burned (2008), and (5) controlled. For simplicity, we will refer to the treatments as 1, 2, 3, 4, and 5 hereafter. The treatments were not random; they were prescriptive in that those plots that were unlikely to experience wildfire and did not require thinning were controls, whereas those plots with the greatest fire risk were selected for the burning treatments. The remaining plots were assigned the thinning treatment. The ten watersheds were instrumented in summer of 2003 and monitored for seven years starting in 2004. The simulated wildfire treatments were applied in October 2006, the salvage and thinning in October 2007, and the post-thinning prescribed fire in October 2008.

For each watershed, a sampling grid with a 30-m spacing was applied (Figure 1). The number of points within each watershed ranged between 19 and 62 depending on the size of the watershed. In total, there were 383 sampling locations that were marked with a stake. Between 2004 and 2010, from early February through April or May, snow depth was measured at each sampling location using a 1-m marked iron rod to characterize conditions near peak snow and during the midmelt phase. A total of 70 snow density measurements were made using a 1-m marked tube at randomly selected points within each watershed.

An average snow density was obtained for each watershed on each sampling day and multiplied by the snow depth at each location to estimate SWE.

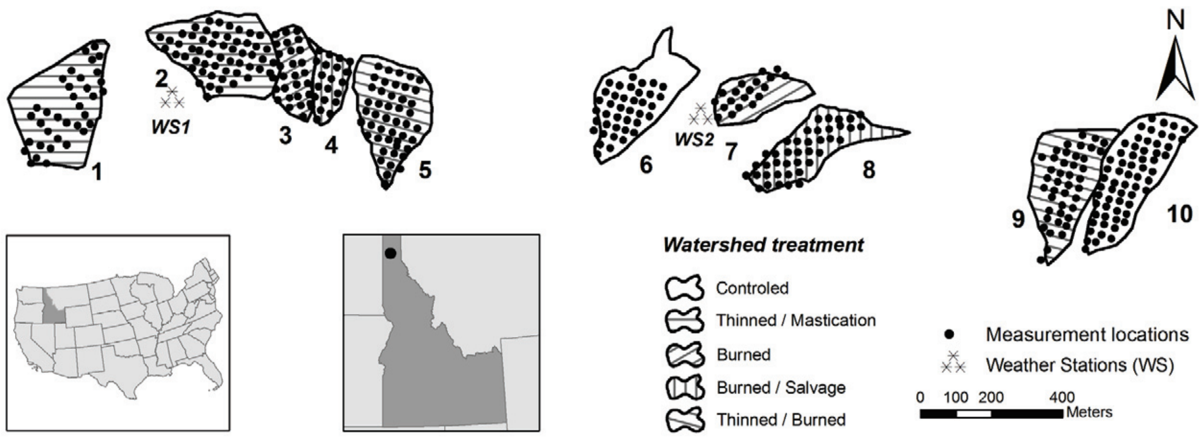

Figure 1. Location of the study site and sampling scheme.

The sampling was initially twice per month, distributed throughout the winter season until the snowpack was completely melted. It took 2–3 days to complete a sampling event. On three occasions half of the watersheds were measured at the end of the month while the other half were measured at the beginning of the next month. In such situations, for analysis, we considered all watersheds as being surveyed at the end of the month.

Snowmelt rates were calculated as the difference in SWE divided by the number of days between two consecutive measurements.

The snowpack melted by the beginning of March in all years except 2008, when it lasted until May. To evaluate the means of SWE and snowmelt for the whole snow season (February–May) in all years, we averaged the SWE and snowmelt values across the four months, considering a value of 0 for the months without snow.
Forest canopy cover at each measurement location following the treatments was measured using airborne laser mapping technology, also referred to as Light Detection And Ranging (LiDAR). Specific topographic variables (elevation, slope, aspect, and curvature) were derived from a 10-m digital elevation model (DEM) for the same locations. Two hydrometeorological stations were installed in the vicinity of watersheds 2 and 7 (Figure 1), and continuous climatic variables were measured for the entire period of the study. Weather station 1 was situated at a lower elevation and the climatic variables at this location were assigned to the first five watersheds, while weather station 2 was at a higher elevation and the climatic variables here were assigned to the last five watersheds.

In total, we used 33 variables: SWE; snowmelt; site; year; month; Julian date; treatment; elevation; aspect; slope; curvature; plan curvature; profile curvature; solar radiation derived from DEM; percent canopy cover; snow depth; average, maximum, and minimum temperatures; average, maximum, and minimum relative humidity; solar radiation measured at the two nearby weather stations; average, maximum, and minimum wind speed; wind direction; average soil water content; precipitation; degree day (number of days with temperature above 0°C between two consecutive measurements); average temperature between two consecutive measurements; actual vapor pressure; and dew point temperature.

Data Analysis

Descriptive Statistics

We were interested in observing the effects of the treatments on SWE both by year and by month. The effects, however, were not obvious as SWE and snowmelt are influenced not only by treatments but also by topographic and climatic variables. Another reason is that vegetation tends to recover in the years following the treatments, and therefore observing the effects of the treatments is difficult in a single study as the weather effects in the first year after treatment will be greater than in the subsequent recovery years.

Proc Mixed Procedure (1999, SAS) was used to assess the least squares means (LS means) and the differences between LS means between treatments across all years and months at a significance level of $\alpha = 0.05$.

Correlation Analyses

The statistical analyses were conducted with SAS 9.1 at $\alpha = 0.05$ (1999, SAS). The Spearman correlation coefficient was used to assess the correlations between SWE and snowmelt with 33 variables. SWE and snowmelt were the response variables while the topographic, climatic, temporal, and canopy cover variables were considered independent variables.

Results and Discussion

Descriptive Statistics

Before any treatments, we observed more snow in the control plots, and this trend continued throughout the study. We observed a greater decrease in the amount of SWE for treatments 1 and 2 compared to the other watersheds for year 2006 (Figure 2). There was no statistically significant difference between the burned watersheds in 2006 and the watersheds in 2007, following the harvesting and thinning treatments.

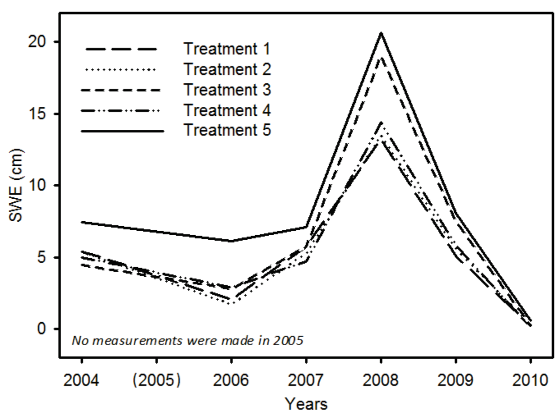

Figure 2. Differences in yearly SWE across all treatments.

Similarly, all the treatments had less SWE compared to the control in year 2008, except treatment 3 which, following the burn in 2006 and the salvage in 2007, had a similar SWE to the control treatment, although they differed statistically.

The difference is smaller in the other years, with generally less amounts of SWE on all treatments. Treatment 3, burn followed by salvage, resulted in a vegetation condition similar to the control watersheds, which, in fact, was the silvicultural goal for the prescription. The other treatments appeared to have little effect on SWE. The difference in SWE is greater between years than among treatments within any given year, likely as a result of variability in weather.

The slope of the curve SWE versus month in Figure 3 reveals the rate of snowmelt. The snowmelt rates changed with time similarly as SWE. Treatments 1, 2, and 4 experienced a similar trend in the snowmelt rate from February until May. For treatments 3 and 5, melt was slower from February to March but faster from March to April. Both treatments 3 and 5 statistically differed from the other treatments in terms of snowmelt rate and from each other.

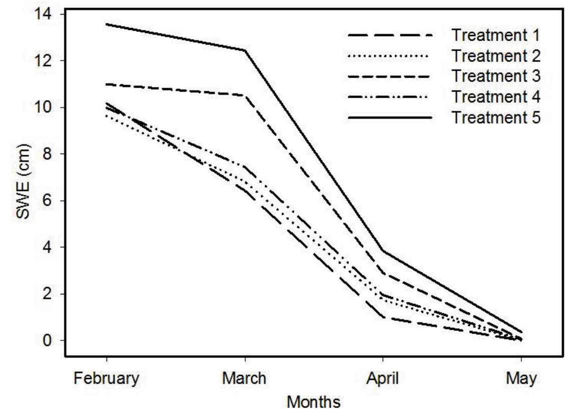

Figure 3. Differences in monthly SWE across all treatments.

Correlation between SWE (snowmelt) and Other Variables

Temporal, weather, and vegetation variables used in this study were correlated with both SWE and snowmelt, but topographic variables were not. We expected topographic variables to have an influence on snow processes. Varhola et al. (2010) submit that slope, together with aspect, should have an influence on snowmelt rates, although Anderton et al. (2004) found no significant terrain control, including slope, in an alpine study. The reason is that the majority of the variables were climatic, representing the main variables influencing both SWE and snowmelt. As shown in other studies (Watson et al. 2006, Jost et al. 2007), elevation and canopy should be strongly related to snow accumulation and melt as we originally expected. Yet we found both variables correlated with SWE for only the years after the treatments were applied, and among the two, only elevation influenced the snowmelt. Our study was focused on ten small watersheds with approximately 400-m difference in elevation, and substantial differences in snow accumulation and melt did not occur during our study. In regard to the canopy cover, most studies assessed effect of this variable under two conditions: open versus forested areas (Jost et al. 2007). In our study, we used a range of canopy amounts. Since the variation in SWE was high, similar to findings in previous studies (Jost et al. 2007), the correlation between SWE and canopy cover was also low.

We further evaluated the correlations separated for each year and month. Among the selected correlations, we present only those that were significantly different from the correlations obtained using the entire dataset. Unless specified otherwise, all the correlations presented were for $p < 0.0001$.

SWE

The strongest correlations were between SWE and month before ($r_s = -0.82$) and after the treatments ($r_s = -0.67$), suggesting that as month increases, SWE decreases. No correlations were observed for SWE versus elevation before treatments and for the year 2010. All the other years had correlation coefficients between 0.05 and 0.12 for 2007 and 2009, and 0.30 and 0.36 for years 2006 and 2008, respectively. The overall correlation after the treatments was 0.14. The higher correlation with elevation in 2008 suggests that elevation was more important for years with larger amount of snowfall.

Canopy cover had a low correlation with SWE for years 2006 ($r_s = 0.06$, $p = 0.0009$) and 2008 ($r_s = 0.08$, $p = 0.0001$). No correlation was found for all the other years, including the year before the treatments.

The correlations between SWE and wind speed or wind direction varied for the different years. All the correlations with maximum wind speed are negative and range from −0.26 for 2006 to −0.40 for 2008; there are no correlations when considering the dataset with all the years after the treatments. There is a positive correlation before the treatments ($r_s = 0.10$). The correlations were negative for all years except for year 2008 when the correlations with average wind speed ($r_s = 0.23$) and wind direction ($r_s = 0.34$) were positive. A negative correlation means that SWE decreases with increasing wind speed, while a positive correlation means that SWE increases with increasing wind speed. As wind speed was shown to have a great influence on the redistribution of SWE (Luce et al. 1999, Anderton et al. 2004), our results are agreeable with these findings.

There was no correlation between SWE and soil water content for the years before treatments, but there was a positive correlation for the years after the treatments ($r_s = 0.41$). The positive correlation is observed for the individual years 2006 ($r_s = 0.42$), 2007 ($r_s = 0.28$), 2009 ($r_s = 0.55$), and 2010 ($r_s = 0.40$); the correlation was negative for year 2008 ($r_s = -0.29$). The positive correlation suggests an increase in soil water content with increasing snow accumulation. This result may be interpreted as a higher SWE and could imply a warmer soil temperature at the beginning of the snow season, causing fresh snow in contact with soil to melt and increasing the soil water content. The negative correlation for 2008 might be due to the late melting of the snow in late April and early May for this year.

Actual vapor pressure was the only variable correlated with SWE after the treatments ($r_s = -0.27$, p = 0.05), and dew-point temperature was negatively correlated with SWE for the years both before ($r_s = -0.08$, $p = 0.03$) and after ($r_s = -0.25$) the treatments.

Snowmelt

The correlation between snowmelt and solar radiation increased from 0.26 before the treatments to 0.42 after the treatments, with a maximum for the year 2008 ($r_s = 0.45$). The stronger correlation after the treatments is possibly due to an increase in the open areas within the forests.

Average, maximum, and minimum temperatures were in general positively correlated with snowmelt after the treatments and for each year. Among the various forms of relative humidity, average relative humidity had the strongest correlation after the treatments ($r_s = -0.58$) compared to before the treatments ($r_s = -0.34$).

There was no correlation between soil water content and snowmelt rates before the treatments, but there was a negative correlation after the treatments ($r_s = -0.16$). The highest correlation between snowmelt and soil water content was for the year 2009 ($r_s = -0.68$), and there is no correlation for 2006. The highest monthly correlations were for February ($r_s = -0.22$) and March ($r_s = -0.42$) and the lowest for April ($r_s = -0.07$).

Conclusions

We were able to observe the effects of the treatments on SWE and snowmelt, although in the third year following the treatments, the effects were less pronounced for normal years, suggesting a high variability of both SWE and snowmelt with years but less with treatments. Greater differences in the rates of snow accumulation and melt among the five treatments were noticed for the more extreme years, such as 2008. With the exception of treatments 3 and 5, there was no clear difference among the means of SWE and snowmelt for the other three treatments. Although the treatments can increase the SWE by creating more open areas within the canopy, this increase can be offset by the increased wind speed and solar radiation, which in turn can result in higher sublimation rates

(Woods et al. 2006, Varhola et al. 2010). No difference was observed in the snowmelt rates in any of the treatments in the two years after the disturbance. As in the case of snow accumulation, the differences in the rates of snowmelt were substantial only in 2008, again underlying the importance of climate in these watersheds.

The large number of correlations in our study is another indicator of the complex interactions among the climatic and topographic variables in forests. At high elevations, as in alpine regions, and in areas with large variations in elevation, the variables responsible for snow accumulation and ablation might be easier to detect. In contrast, these interactions are more pronounced in lower elevation areas or in areas with insignificant variations in elevation.

The strongest correlations for SWE were between month, soil water content, temperature, relative humidity, and actual vapor pressure, and for snowmelt they were between month, relative humidity, temperature, solar radiation, average temperature between two consecutive measurements, and wind speed.

Although our study shows differences in SWE and snowmelt due to treatments, these differences are minor as the changes in canopy structure were not dramatic. This can be seen especially for treatments 1, 2, and 4.

The results presented in this paper are preliminary and exploratory. A more in-depth statistical analysis is needed and will be conducted to better understand the relations of SWE and snowmelt with the other environmental and physical variables as well as years used in this study.

Acknowledgments

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Headwater Variability across the Rain-Snow Transition in California's Sierra Nevada: Stream Discharge, Runoff Timing, and Sediment Yield

C.T. Hunsaker

Abstract

The hydrologic response of eight headwater catchments located at and above the rain-snow transition, 1,500–2,500 m elevation, was investigated over five years (2004–2008) using hourly streamflow, precipitation, snowpack, and weather-station data. The Kings River Experimental Watersheds (KREW) is a watershed-level integrated ecosystem project for long-term research on headwater streams in the southern Sierra Nevada. It is designed to evaluate forest restoration treatments—mechanical thinning and understory burning—and addresses knowledge gaps identified in the Forest Service's adaptive management strategy for the Pacific Southwest Region. KREW also is ideally suited to address climate change because of its location; rain-dominated lower elevation watersheds, that also receive snow, provide a surrogate for how snow-dominated, higher elevation watersheds, could function with climate change.

The annual runoff ratio (discharge divided by precipitation) in eight headwater catchments located across the rain-snow transition increased about 0.1 per 300 m of elevation. Higher elevations have lower vegetation density, coarser soils, and a shorter growing season when compared with lower elevation lands. Average temperature across the 600-m average elevation range was only 1 to 2^{o}C warmer in the lower versus upper elevation catchments, with annual precipitation being 20–50 percent snow at the lowest elevations versus 75–95 percent at the highest. Peak discharge lagged peak snow accumulation on the order of 60 days at the higher elevations and 20 to 30 days at the lower elevations. Snowmelt dominated the diel streamflow cycle over a period of about 30 days in higher elevation catchments, followed by a 15-day transition to evapotranspiration dominating the diel streamflow cycle. Discharge from lower elevation catchments was rainfall dominated in spring, with the transition to evapotranspiration dominance being less distinct. Base flows ranged from <1 to 10 liters per second (L/s), but during spring snowmelt flows of 400 L/s occur for a month or more. Maximum peak flows of more than 1,000 L/s were measured for a single rain event. Annual sediment yield varied from year to year. One of the managed watersheds in the rain-snow transition zone produced 1.8, 15.2, and 18.7 kg/ha/yr for water years 2004, 2005, and 2006, respectively. The increase in sediment accumulation correlated with an increase in annual precipitation. The snow-dominated and managed watersheds and the one undisturbed watershed produced similar, and sometimes higher, sediment loads for these same years. This is an interesting finding since snow-dominated areas are expected to produce less sediment without rain-drop and rapid surface runoff erosion.

Climate warming that results in a longer growing season and a shift from snow to rain would result in earlier runoff and lower water yield. KREW measurements indicate that about one-third less water could flow in the streams from high elevation headwaters in the southern Sierra as temperatures increase 1–2°C. Sustaining or enhancing water yields in these mixed conifer forests will require management of evapotranspiration by reducing vegetation density and using thinning patterns that attenuate snowmelt. KREW hosts the National Science Foundation's Southern Sierra Critical Zone Observatory and is designated for inclusion in their National Ecological Observatory Network.

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Contrasts in Carbon and Nitrogen Ecosystem Budgets in Adjacent Norway Spruce and Appalachian Hardwood Watersheds in the Fernow Experimental Forest, West Virginia

Charlene Kelly, Stephen Schoenholtz, Mary Beth Adams

Abstract

We constructed watershed mass-balance budgets of carbon (C) and nitrogen (N) and measured seasonal net N mineralization in an attempt to account for nearly 40 years of large discrepancies in stream NO_3-N export in two adjacent, gauged watersheds at the U.S. Department of Agriculture Forest Service's Fernow Experimental Forest, WV. These watersheds have similar management histories, varying primarily by vegetation cover, where one watershed is a monoculture of Norway spruce (*Picea abies*) and the other has regenerated to native Appalachian hardwoods. Long-term stream chemistry indicates that the hardwood watershed has approached N-saturation, with relatively high stream export of nitrate-N (15 kg NO_3^--N/ha/yr), whereas the spruce watershed exports virtually no nitrate-N. We estimated the pool size of C and N within the mineral soil, forest floor, litter, above- and below-ground tree biomass, and stream dissolved organic N. We were unable to account for long-term differences in NO_3-N export via streamflow by estimating these pools. Total C and N pools were 28 percent and 35 percent lower in the spruce watershed, respectively. Though historic organic C and N were never measured in the long-term stream chemistry, the discrepancy in C and N budgets between the two watersheds suggests that the spruce watershed may have been subjected to a period of large losses of C and organic N from deeper subsurface soils. Such large losses suggest that species conversion has the potential to significantly alter ecosystem C and N budgets, with implications for long-term productivity, C sequestration, and water quality.

Keywords: Fernow Experimental Forest, Norway spruce, carbon, nitrogen

Introduction

Stream chemistry at watershed outlet weirs integrates ecosystem functions (chemical, biological, and physical) and displays responses of the total watershed to alteration. Studies of nutrient mass-balance budgets have been utilized to account for differences in stream chemistry (e.g., lower stream nitrate export) and to identify effects of management regimes on ecosystem processes influencing such differences (e.g., Triska et al. 1984). Conversion of native vegetation to monocultures of conifer may disrupt biogeochemical cycling of carbon (C) and nitrogen (N) (Guo and Gifford 2002).

Two adjacent watersheds within the U.S. Department of Agriculture (USDA) Forest Service's Fernow Experimental Forest (FEF) in West Virginia provide an excellent opportunity to investigate the specific role of tree species in ecosystem N cycling and retention. Long-term stream chemistry of these watersheds indicates divergent export of stream NO_3-N at the outlets. Mean annual stream NO_3-N exported from an experimental 39-yr-old hardwood stand (watershed 7) is nearly 15 kg/ha, whereas stream NO_3-N exported from a nearby experimental 37-yr-old Norway spruce stand (watershed 6) has been nearly zero for 20 years (mean = 0.18 kg/ha/yr).

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The present work is an attempt to account for nearly 40 years of large discrepancies in stream NO_3-N export in two nearly adjacent, gauged watersheds at the FEF using estimates and comparisons of key components of ecosystem C and N budgets. It was hypothesized that because NO_3-N export has been negligible from the spruce watershed, and because inputs to the two watersheds from atmospheric deposition are equal, then C and N pools will have accumulated to a greater extent in vegetation, forest floor, and soil horizons in the spruce watershed because of slower decomposition of organic material and slower nutrient cycling, resulting in low NO_3-N export to the stream.

Specific objectives were to (1) measure selected pool sizes of C and N within each watershed to ascertain if significant differences in these pools occur after nearly 40 years of influences from contrasting forest vegetation and (2) measure rates of net N mineralization to determine if this measure of current N flux was associated with differences in size of selected C and N pools in the two watersheds.

Methods

Description of the Watersheds

The watersheds used in this study are located within the USDA Forest Service FEF near Parsons, WV (USA). See Kochenderfer (2006) and Kelly (2010) for complete site descriptions. Both watersheds 6 and 7 (WS6 and WS7) were clearcut-logged in sections (1964–1967) and maintained barren with herbicides until 1969. Watershed 6 (22 ha) was planted with Norway spruce in 1973, but WS7 (24 ha) was allowed to regenerate naturally beginning in 1970. After nearly 40 years of growth, the spruce has a closed canopy and dense stand structure with mean basal area of 23 m^2/ha.

Soils in WS6 are mapped as Calvin series (Soil Survey staff, USDA Natural Resources Conservation Service) derived from shale, siltstone, and sandstone parent material. Soils in WS7 are mapped as both Calvin and Dekalb series derived from acidic sandstone parent material. This watershed is dominated by yellow-poplar, red oak, and sugar maple, with mean basal of 17 m^2/ha.

Historic NO_3-N export and specific conductivity data from the spruce and hardwood streams indicate close similarity in ecosystem biogeochemical activity at the time of conversion to a Norway spruce stand (Kelly 2010).

Atmospheric Deposition and Stream Export

Data for wet and dry deposition of NO_3-N were attained from annual records from the National Atmospheric Deposition Program monitoring site WV18 and the CASTnet monitoring site PAR107. Streamflow and weekly NO_3-N concentration data were attained from the USDA Forest Service Timber and Watershed Lab, Parsons, WV. These data were used to calculate total inputs and exports of NO_3-N from the two watersheds for 1973–2009. Total dissolved N was analyzed from 2007–2009 in monthly stream samples from both watersheds in order to determine export values of dissolved organic N (DON) that had not previously been measured in these watersheds.

To determine the potential flux of inorganic N resulting from N mineralization, intact soil cores (0–10 cm) were collected at 12 sampling sites in each watershed. Incubations were performed seasonally for two years to determine temporal differences in net ammonification, nitrification, or immobilization.

Soil C and N Pools

To estimate C and N pool sizes within surface soil horizons and the forest floor, soil samples were collected in July 2007 from 30 sampling sites within each watershed at each horizon. Soil horizons are defined in these watersheds as A (0–10 cm) and B (10–46 cm).

Forest floor samples were collected in October 2007 and 2009 to characterize depth, oven-dry weight, and total C and N at each sampling location. Freshly deposited litter materials were collected monthly from 2008 to 2009.

Biomass C and N Pools

Diameters of trees were measured by the USDA Forest Service in 2003. Each individual tree was converted to biomass (kg) using allometric equations for both above- and below-ground (see Kelly 2010 for description of equations). From total above- and below-ground biomass estimates, tree compartment mass and percent C and N were estimated using values published by Whittaker et al. (1974) for the hardwood watershed and by Feng et al. (2008) for the spruce watershed.

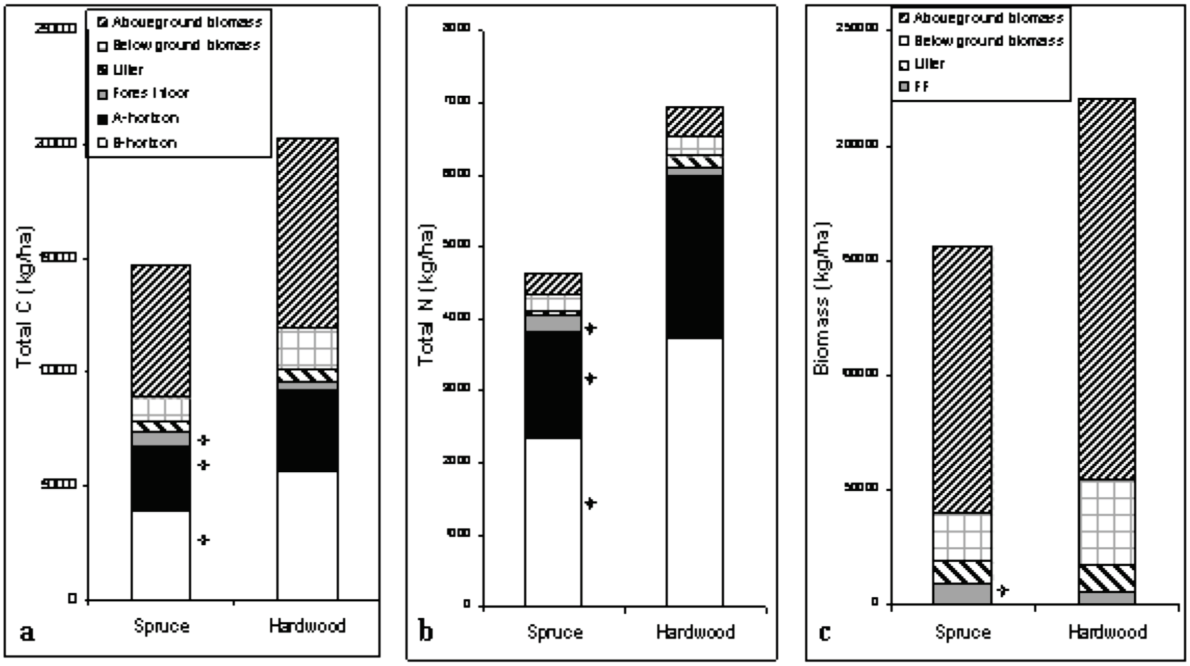

Figure 1. Watershed budgets depicting the mass of (A) carbon, C; (B) nitrogen, N; and (C) biomass contained within each soil and biomass compartment in the spruce (WS6) and hardwood (WS7) watersheds in the FEF. Asteriks denotes significant differences between spruce and hardwood pools at the α=0.05 level.

Data Analysis

One-way analysis of variance (ANOVA) was performed to determine differences in C and N pool size within each compartment by watershed and in in situ net nitrification and net ammonification annually and within each season by watershed.

Results

Inputs and Exports

Since 1973, combined wet and dry atmospheric deposition of NO_3-N is estimated to be 320 kg N/ha for each of the two watersheds. Total stream export of NO_3-N from the spruce watershed since 1973 is 45.93 kg N/ha, which is only 13 percent of the stream export of NO_3-N that occurred during the same time period in the hardwood watershed, which exported 341.07 kg NO_3-N/ha, exceeding the value of atmospheric deposition by approximately 21 kg N/ha. Annual DON export from the spruce watershed was also very low (mean = 0.477 kg DON/ha/yr) during the three years of measurement. In contrast, annual DON export (mean = 10.03 kg DON/ha/yr) from the hardwood watershed was nearly equal to export of NO_3-N (mean = 10.70 kg NO_3-N/ha/yr).

Potential Flux from N Mineralization

Total net N mineralization annual flux was approximately three times greater in the hardwood watershed than in the spruce watershed. Mean annual net N mineralization was approximately 182.50 and 64.0 kg N/ha/yr from the hardwood and spruce soils, respectively.

Soil and Forest Floor C and N Pools

Carbon and N pools within the mineral soil horizons were significantly lower in the spruce watershed relative to the hardwood watershed (Figure 1A). Within the A-horizon, spruce soil contained about 20 percent less C (kg/ha) than the hardwood soil (Figure 1A). Within the B-horizon, spruce soil contained approximately 30 percent less C (kg/ha) than the hardwood soil.

A-horizon soil in the spruce watershed contained nearly 35 percent less N than the hardwood soil (2.03 g N/kg in spruce versus 2.93 g N/kg in hardwood) (Figure 1B). Within the B-horizon pool, spruce soil contained nearly 38 percent less N content (kg/ha) than the hardwood soil (0.73 g N/kg in spruce soil versus 1.17 g N/kg in hardwood soil).

Forest floor C content (kg/ha) was significantly greater in the spruce watershed than the hardwood watershed (3,813 and 2,343 kg C/ha in the spruce and hardwood watersheds, respectively) (Figure 1A). Forest floor N (kg/ha) was significantly greater in the spruce watershed than in the hardwood watershed (132.25 kg N/ha versus 74.68 kg N/ha, respectively) (Figure 1B).

Tree Biomass Pools

Both above- and below-ground biomass in trees, as estimated by allometric equations, were higher in the hardwood watershed (Figure 1C). Above-ground biomass estimates were approximately 30 percent less in the spruce watershed (116,800 kg/ha) relative to the hardwood watershed (166,000 kg/ha). Below-ground biomass estimates were approximately 45 percent less in the spruce watershed (21,000 kg/ha) relative to the hardwood watershed (38,000 kg/ha). This greater biomass in the hardwood watershed equated with larger pool sizes of C and N in the hardwood trees than the spruce (Figure 1A, B).

Discussion

Contrasting Vegetation and C and N Pools

The goal of this study was to quantify selected ecosystem C and N pools in two watersheds that exhibit large differences in long-term stream export of NO_3-N measured since establishment of contrasting forest types in 1973. Results of this study indicate that we were unable to account for these differences in NO_3-N export via streamflow through estimation of the size of C and N pools within the forest floor, mineral soils, above-ground tree biomass, and below-ground tree root biomass in the two watersheds.

Total C and N pools were lower in the spruce watershed in nearly every compartment measured (Figure 1), as was total N mineralization. Total C pools were 28 percent less in the spruce and total N pools were 35 percent less in the spruce relative to the hardwood watershed. The B-horizon soil compartment exhibited the largest difference in both C and N stores (32 percent less C and 38 percent less N in the spruce watershed). These results were contrary to the hypothesis that soil and forest floor C and N stores would be higher in the spruce watershed, thereby accounting for 40 years of relatively high atmospheric N input and very low stream export of NO_3-N from the spruce watershed.

Total N pools in the mineral soil and in tree biomass of two additional watersheds (watersheds 4 and 10) within the FEF are shown in Figure 2. Watersheds 4 and 10 are often used as references because they have been left to natural recovery since being logged in 1905. Comparing the spruce watershed (WS6) to watersheds 4, 7, and 10, which are characterized by native hardwood forests, illustrates that the spruce watershed has considerably less N in the measured pools (Figure 2). It is also noteworthy that the native hardwood watershed of this study (WS7) has similar estimated N pool sizes in soil and biomass as reference watershed 10 (Figure 2).

What Accounts for Differences in C and N Pools?

Three possible mechanisms have been identified that could explain why, after 40 years of much lower NO_3-N export and high atmospheric deposition, the spruce watershed does not exhibit patterns of C and N accumulation comparable to the hardwood watershed.

1. The watersheds are intrinsically different, and the spruce watershed has always had much smaller storage of C and N than the surrounding watersheds;
2. The spruce watershed has been losing N via denitrification at a much greater rate than the hardwood watershed for the past 40 years; and (or)
3. The spruce watershed underwent a phase of organic matter degradation when the hardwood stand was replaced by the Norway spruce stand, causing a large amount of organic forms of N to be leached from the system (e.g., Guggenberger et al. 1994).

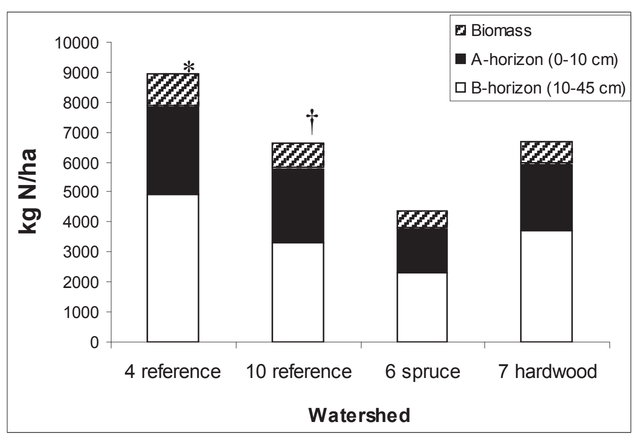

Figure 2. Nitrogen contained within A and B soil horizons and within total biomass (above- and below-ground) for four watersheds within the FEF.
* from Adams et al. (2006); sampled in 2002
† from Christ et al. (2002); sampled in 1997

It is unlikely that the spruce watershed is intrinsically different from the surrounding hardwood watersheds within the FEF to the degree observed in the present study (Figure 2). The soils within all of these watersheds are of the same soil series and have similar historic land use and atmospheric inputs. Historic NO_3-N export and specific conductivity of streamwater indicate close similarity in ecosystem biogeochemical activity at the time of conversion to a Norway spruce stand. When analyzed by decade after conversion, it can be seen that in the first decade following treatment (1971–1980), patterns of stream NO_3-N values were very similar between the watersheds (R^2=0.96). Furthermore, the divergence in specific conductivity did not occur until after the Norway spruce canopy closure occurred (R^2=0.20 after 1981 and R^2=0.0005 after 1991) (Kelly 2010).

Large losses of NO_3-N were not detected in the long-term stream chemistry data for the spruce watershed, suggesting that N might have been lost through fluxes in gaseous phase of N_2O or NO (Reddy and Patrick 1975) and (or) through stream export of DON (Campbell et al. 2000) (also not measured). It is unlikely that denitrification processes can explain the relatively small N pools in the spruce watershed because (1) large fluxes of denitrification usually result in accumulation of C in the organic horizons of soils, which was not observed in the current study, and (2) N losses via denitrification usually account for a small portion of total ecosystem N (Mohn et al. 2000). Additionally, soils in these watersheds are relatively well-drained upland soils with relatively low potential for significant denitrification.

It is more likely that soils within the spruce watershed underwent a phase of organic matter degradation or mass transport of sediment prior to spruce stand establishment, inducing a loss of C and N that was not detected in the long-term stream data that only measured losses of NO_3-N and that did not measure DON or particulate N. This concept of organic matter degradation or mass transport is strongly supported by a chronosequence study of soil C stocks beneath red spruce (*P. rubens*) forests in northeastern North America (Diochon et al. 2009). Soils within these forests exhibited increasingly smaller stocks of C from 1-, 15-, and 45-yr-old stands, reaching a minimum of approximately 76 Mg C/ha in the soil profile at 45 years. Soil C values in the 45-yr-old red spruce stand were very similar to the Norway spruce soils in the present study, which contain 74 Mg C/ha. Carbon loss (decrease in C concentration and content) from young stands in the Diochon et al. study (2009) was reported to occur through enhanced mineralization of organic compounds (verified with stable C isotopic analysis), especially in the deeper soil horizons. Similar declines in soil C were also observed in spruce forests of similar age by other authors (e.g., Parker et al. 2001, Tremblay et al. 2002), indicating that this rate of C loss from spruce soils is a common phenomenon.

Forest clearing may result in decreases in soil C, but C stores generally recover to original levels after several decades, especially if the stand is allowed to regenerate (Harrison et al. 1995). Loss of soil C upon hardwood conversion to conifer can be attributed to both disturbance and changes in amount and composition of plant material returned to the soil via litter and root turnover (Lugo and Brown 1993). Additionally, the presence of ectomycorrhizal fungi introduced upon vegetation conversion have also been documented to induce a 30 percent soil C depletion within 20 years of establishment of an exotic Radiata pine (*Pinus radiata*) plantation (Chapela et al. 2001).

Spruce vegetation has high lignin content in litter materials and shallow rooting architecture. High lignin content results in larger proportions of soil organic N relative to inorganic forms because of slower decomposition and mineralization (Berg and Theander 1984). Shallow rooting architecture may result in leaching of soil C following decomposition of deep roots of the native hardwood that were present prior to conversion, with little subsequent vegetative uptake or stabilization deep in the spruce soil profile. Thus, the spruce features of slowly decomposable organic matter and shallow rooting may help explain the apparent large mass losses of soil C and N relative to the native hardwood in this watershed study.

Conclusions

Results of this study suggest that a significant loss of C and N from ecosystem pools likely occurred following conversion from native hardwoods to a monoculture of Norway spruce in the FEF. Consequently, species selection should be taken into account when managing forests for future C sequestration, for provision of high-quality water, and for effects of high atmospheric inputs of N, especially when relatively short rotation times are implemented between harvests.

Acknowledgments

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Lakes, Wetlands, and Soil Moisture

Classification, Mapping, and Management of Wetlands in Alaskan Watersheds with Rapidly Growing Populations

Mike Gracz

Abstract

Teddy Roosevelt said that we are not so rich that we can afford to preserve everything, but neither are we so poor that we cannot afford to preserve anything. Senator Ted Stevens advocated for waiting until 2 percent of Alaska's wetlands were filled before having section 404 of the U.S. Clean Water Act, regulating placement of fill into wetlands, take effect in Alaska. Increasing development pressure on our aquatic resources might have led us to ask Ted the question: "Which two percent?" Wetland mapping and assessment efforts in the Matanuska-Susitna Valley and on the Kenai Peninsula, including a separate effort by the City of Homer, aim to answer the question we might have posed to Senator Stevens. When wetlands cover less than 1 percent of a State's area, as they do in Nevada or New Mexico, then "we can afford to preserve everything," but when they cover at least 40 percent, as they do in Alaska, some will be filled. Identifying which wetlands can be filled under what conditions is essential to maintain green infrastructure and avoid the costly mistakes made elsewhere. I report on a new wetland mapping classification and on wetland management plan efforts that aim to avoid and minimize disturbance to wetland functions in the face of continued land use change in populous watersheds. Some of the largest successes of these projects have come from unanticipated sources; we should learn from those successes, even though they may seem outside of the normal watershed planning paradigm.

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Will the Arctic Coastal Plain Wetlands Disappear?

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Abstract

Arctic Coastal Plain wetlands in Northern Alaska support a multitude of wildlife and natural resources that depend upon the abundance of water. Observations and climate model simulations show that surface air temperature over the Alaskan arctic coast has risen in recent history. Thus, a growing need exists to explore how the hydrology of these arctic wetlands will respond to the warming climate. In order to answer this question, we conducted a synthesis study combining the observational analysis of an extensive field campaign, which includes direct measurements of all components of the water balance, with a physically-based hydrological model forced by downscaled climate projections. Our studies show that unless the near-surface permafrost was to degrade, forming a more extensive drainage network of troughs, these arctic wetlands would remain temporarily inundated and wet throughout the snow-free period.

Currently, these wetlands exist despite a desert-like annual precipitation regime and a negative net summer water balance. At the primary study site within the Biocomplexity study area on the Barrow Environmental Observatory, shallow ponding of snowmelt water occurs for nearly a month at the vegetated drained thaw lake basin (DTLB). The length and depth of the ponding is only replicated by the hydrological model if the rims of low-centered polygons are represented. Simple model experiments suggest that the polygon type (low- or high-centered) controls runoff, evapotranspiration, and near-surface soil moisture. High-centered polygons increase runoff while reducing near-surface soil moisture and evapotranspiration.

A change in polygon type, from low- to high-centered polygons, has a larger effect on the near-surface soil moisture under the current meteorological forcing than the projected hydrological response to the altered climate regime alone. In the latter, the projected increase in summer precipitation offset the simulated increase in evapotranspiration. The length of the ponding period remained, while its start date became more variable. Nonetheless, the fall water tables for the end of the 21st century were similar to model simulations and field measurements for the present (1999–2009).

The model experiments show that microtopography plays an important role on the hydrologic fluxes and stocks of arctic wetlands. Although no soil drying was projected by the end of the 21st century, differential ground subsidence could potentially dominate the direct effects of climate warming resulting in a drying of the Arctic Coastal Plain wetlands. It is therefore crucial to merge hydrology, permafrost, and geomorphology models and observations at the appropriate spatial scales to elucidate the response of the Arctic Coastal Plain wetlands to a warmer climate.

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Lake Districts of the Koyukuk National Wildlife Refuge

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Abstract

The potential for broad scale habitat changes caused by global climate change has increased interest in the hydrology and dynamics of subarctic lakes and wetlands in recent years. The U.S. Fish and Wildlife Service and other Interior Alaskan land managers charged with the protection of wildlife species, habitat, and water resources seek a better understanding of boreal lake and wetland dynamics in order to anticipate potential changes to important natural resources. Using a geographic information system (GIS) we identified and delineated lake districts within the Koyukuk National Wildlife Refuge, Alaska, as a framework for future inventory and monitoring of Refuge lakes, wetlands and wildlife. We used the U.S. Geological Survey National Hydrography Dataset to identify and map lakes larger than 1 ha occurring within the Refuge. We then used a grid scaled to the geographic area and the number of lakes in the Refuge to identify lake rich areas, defined by (1) a lake surface area (limnetic ratio, in percent) greater than the mean for the Refuge, and (or) (2) a lake density (number of lakes per geographic cell) greater than the mean for the Refuge. The distribution of cells meeting these criteria, as well as information on surficial geology, watershed boundaries, lake shape and type, and field observations, were used as visual guides for manual delineation of lake districts. We identified and delineated 14 lake districts and 4 subdistricts. Combined, the lake districts encompass 11,940 km^2, representing 66 percent of the total land area of the Koyukuk Refuge. The lake districts contain 98 percent of Refuge lakes over 1 ha and 99 percent of lake-cover on the Refuge. Lake abundance on the Refuge was calculated to be 0.54 lakes/km^2 overall and 0.80 lakes/km^2 for the lake districts combined. Limnetic ratio for the entire Refuge was 3.9 percent and the limnetic ratio for the lake districts combined was 5.9 percent. We explored the following qualities of each lake district to identify similarities and differences in the regions: lake characteristics (lake density and limnetic ratio, lake size, shape, type, connectivity and flood potential), elevation (land elevation, hypsometric index, and lake elevation), surficial geology, vegetative cover, and wildlife populations (based on data collected through the Refuge Inventory and Monitoring Plan). We intend to use the lake district analysis for the Koyukuk Refuge as a framework for future field-based studies, habitat analyses, long-term monitoring and wildlife management. We will combine this information with data collected in the field to characterize the important hydrological processes at work within each lake district, to study relations between distinct lake and wetland systems and wildlife populations, and to forecast the potential effects of climate change.

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Water Quality Monitoring

Reconnaissance Investigation of Emerging Contaminants in Effluent from Wastewater Treatment Plant and Stormwater Runoff in the Columbia River Basin

Jennifer Morace

Abstract

In order to efficiently reduce toxic loading to the Columbia River basin, sources and pathways need to be identified. Little is known about the toxic loadings entering the system from wastewater treatment facilities and stormwater runoff. This study provides preliminary data on these sources and pathways throughout the basin. Nine cities were chosen in Oregon and Washington to provide diversity in location, arid eastside and wet westside characteristics, and population densities. Samples were collected from a wastewater treatment facility in each of the cities and analyzed for wastewater-indicator compounds, pharmaceuticals, PCBs, PBDEs, organochlorine and legacy compounds, currently used pesticides, mercury, and estrogenicity. Currently, these treatment facilities are required to sample their effluent to meet their permit requirements, which are very limited. Little is known about the environmental implications of emerging contaminants in these effluents. Results indicate that a majority of these compounds are present in the effluent and some at environmentally relevant concentrations. Although the grab samples were not time-integrated and the effluent is expected to change in nature throughout time, the continuous input of this number of compounds and at these concentrations may have implications on the receiving waters, the foodweb reliant on these waters, and the ecosystem as a whole.

The second component of the sampling effort was directed at characterizing stormwater runoff for a slightly different set of emerging contaminants—PCBs, PBDEs, organochlorine compounds, PAHs, metals, currently used pesticides, and oil and grease. Studies have shown that stormwater, most often untreated before entering the receiving waters, can deliver significant loadings of these compounds. Unlike effluent from wastewater treatment plants, stormwater runoff is sporadic and unpredictable, and the sudden input of these contaminants has implications for the ecosystem. These two pathways are poorly understood in terms of their toxic contribution to the system, yet they act as integrators of human activities and offer an area where changes could be made to reduce harmful human effects on the environment.

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An Upside-Down River: Impoundments and Eutrophication Alter Downstream Predictions of Water Quality in the Klamath River

Allison A. Oliver, Robert G.M. Spencer, Michael L. Deas, Randy A. Dahlgren

Abstract

Large river impoundments are ubiquitous features in many parts of the world. The River Continuum Concept (RCC) and the Serial Discontinuity Concept (SDC) are two examples of widely acknowledged models of river ecosystems that suggest longitudinal shifts in parameters in response to perturbations such as impoundments. While these concepts may have broad utility, they inadequately address how nutrient enrichment (eutrophication) may alter predictions within regulated rivers. The objectives of our study were to investigate these predictions by determining longitudinal patterns of water quality parameters and organic matter composition within the Klamath River in Oregon and California. We collected monthly water samples for one year at nine sites over 130 miles on the Klamath River, beginning at the headwaters and sampling above and below six reservoirs. Samples were analyzed for dissolved oxygen, conductivity, pH, temperature, turbidity, suspended solids, nutrients (TN, TON, NH_4^+, NO_3^-, TP, TDP, SRP, DOC), and organic matter (OM) composition (chlorophyll a, UV-absorbance, fluorescence index, biological oxygen demand [BOD]). Our results indicate that the Klamath River functions as an "upside-down" river in terms of many of the predictions based on river ecosystem concepts such as the RCC and SDC. Conditions in the headwaters were the most degraded, but conditions generally improved below dams and with increasing distance downstream. The highest concentrations (TN= 3.249 mg/L, NH_4^+= 0.124 mg/L, TP= 0.223 mg/L, DOC= 9.67 mg/L), the most labile OM, and the highest BOD (20-day BOD = 42.7 mg/L) were observed in the headwaters during the summer months. SRP generally remained similar throughout the river or increased slightly in the downstream direction, likely as a result of lower nitrogen:phosphorous ratios and reduced SRP uptake. Using a general linear mixed model, we determined a significant effect of river mile, depending upon time of year and TN, TON, suspended solids, BOD, and OM composition. Overall, downstream improvement in water quality likely results from storage and processing of OM in reservoirs and dilution effects from groundwater and tributaries. Four out of six dams on the Klamath River are planned for removal in the next decade, and these results suggest that the removal of downstream reservoirs may affect the transport of nutrients and organic matter, potentially increasing downstream impairment in the summer months. Dam removal should therefore be considered in conjunction with the restoration of upstream conditions.

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Use of Early Agency Coordination to Efficiently Navigate the Permitting Process for Complex Stream- and River-Related Projects

Hans R. Arnett, Sara E. Lindberg, D. Shane Cherry

Abstract

Early coordination with regulatory agencies made it feasible to permit a major relocation of a catalogued salmon stream as part of the recently completed Ketchikan International Airport Runway Safety Area (RSA) project. The project was funded by both the Southeast Region of the Alaska Department of Transportation and Public Facilities and the Alaska Region of the Federal Aviation Administration (FAA) and undertaken to address a congressional mandate to update safety areas at airports nationwide. The project involved shifting the runway 750 feet to the southeast to provide full 1,000-foot-long safety areas off either end of the runway. At the time the project was awarded, it was the FAA Alaska Region's single largest construction project in history.

The project team convened an interdisciplinary team (IDT) that included the project owner, the project team's design and environmental specialists, and technical specialists from key regulatory agencies. The IDT evaluated multiple options for accommodating the RSA expansion by comparing costs, technical feasibility, and environmental effects of each option. The most feasible alternative for accommodating the runway shift involved a 1,300-foot-long relocation of Government Creek—a salmon stream located immediately adjacent to the south end of the runway. The stream relocation provided opportunities to significantly improve habitat compared to the existing stream conditions. The effects to existing stream and estuarine habitat were mitigated by the ecological improvements made to the newly constructed stream channel and estuary. Through early coordination, regulatory agency concerns were addressed in the initial stages of the design process. This collaboration continued through all phases of design, assuring that permits would be issued without delay for this unprecedented stream relocation project.

Risk and uncertainty associated with the stream relocation were managed effectively by implementing a well-developed adaptive management and monitoring plan. Two phases of stream construction allowed lessons learned on the first phase to immediately apply to and improve the second phase, and minor adjustments to the first phase were facilitated during the construction mobilization for the second phase. The adaptive management plan extended agency coordination through construction and into the post-construction phase, providing multiple opportunities for adjustments to the newly constructed stream to assure project success. The Alaska Department of Fish and Game–Division of Habitat[1] was so pleased with the results that they featured the project on their website in an article called "Mitigation Gone Right!" This collaborative approach with regulatory agencies is currently being successfully applied on complex stream and river relocation projects at the Nome and Cordova airports.

Keywords: early agency coordination, permitting, stream relocation, Ketchikan Airport, Government Creek, adaptive management

Introduction

The following sections provide brief discussions of the complexities, issues, and number of regulatory agencies that can be involved in larger river- and stream-related construction projects, especially when projects must

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[1]Formerly the Alaska Department of Natural Resources Office of Habitat Management and Permitting.

comply with the National Environmental Policy Act (NEPA). Federal and State regulations require mitigation for unavoidable effects to aquatic habitat on such complex projects.

The development process of an engineering project begins with a problem statement and culminates with construction and operation. Historically, project owners initiate coordination with regulatory agencies as necessary at different stages of the project, given the regulations that govern the project's environmental effects. However, integrating regulatory issues into the overall engineering constraints and opportunities analysis early on ensures that such issues do not derail the project schedule and budget when introduced later in the design process. As this paper illustrates, such early agency coordination (EAC) with all regulatory agencies involved in the project reduces schedule and cost risk, helps establish regulatory constraints and priorities, and leads to more timely and efficient project completion.

Regulatory agency personnel are experts in the regulatory processes they administer and often have primary training in engineering or science. The idea that agency personnel can be part of an expert team forms a key part of the philosophy behind early coordination. Making these regulatory experts part of the team early-on results in an alignment of project objectives and regulatory objectives, and establishes a collaborative approach while diminishing the chance of falling into an adversarial dialog. This collaborative relationship continues through the design and construction phases.

Early agency coordination follows the normal process of project development with a series of milestones for design and review. The key innovation of EAC, however, is inviting regulatory agency personnel to participate early in and throughout the entire design development process.

Example Project I - Ketchikan Runway Safety Area Expansion Project

The Ketchikan International Airport (KTN) Runway Safety Area (RSA) Expansion Project provides a good example of effectively using EAC to navigate the permitting process on a large and complex project. The project improved safety at the airport by bringing it into compliance with Federal Aviation Administration (FAA) safety standards. Project design efforts began in June 2004 and were completed in November 2006. Construction started in the spring of 2007 and was completed in the fall of 2009. At the time the project was awarded, it was the FAA Alaska Region's single largest construction project in history.

The RSA expansion project extended the runway embankment 1,500 feet, filling the lower 1,200 feet of Government Creek. A new stream channel and floodplain were constructed around the new RSA fill, and Government Creek flows were diverted into the new channel. The new channel ties into the alignment of a small adjacent stream (Boulder Creek) and enters Tongass Narrows within the Boulder Creek estuary. The relocation provided an opportunity to mitigate for lost habitat by improving ecological function of the stream and floodplain and increasing the amount of instream habitat available to adult and juvenile salmonids.

The KTN RSA Expansion project design, including the hydrologic and hydraulic analysis efforts for the Government Creek relocation, was led by USKH Inc. (USKH), under contract to the Southeast Region of the Alaska Department of Transportation and Public Facilities (DOT&PF). USKH was also the design lead for the new valley of the relocated Government Creek. The lead designer of the relocated stream channel was Shane Cherry. The lead designer of the new Government Creek estuary was Jon Houghton, Senior Marine/Fisheries Biologist with Pentec/Hart Crowser. The project was constructed by SECON.

An interagency scoping meeting was held onsite in September 2004. The intent of the meeting was to introduce agency representatives to the five RSA expansion alternatives that were under development, discuss why other alternatives had been eliminated, and provide a forum for agencies and other concerned parties to raise concerns and issues, or to propose other options.

Both ends of the original RSA were constrained by sensitive environmental resources, with the Airport Creek estuary to the northwest and Government Creek to the southeast.

During the September 2004 interagency scoping meeting, the relative merits, economic feasibility, and environmental effects of each alternative were discussed. Resource agency representatives noted that habitat in lower Airport Creek and its estuary were far superior to habitat in lower Government Creek and its estuary. The bed of Airport Creek was low gradient and dominated by gravel. Its estuary was a productive salt marsh with extensive eelgrass beds and healthy populations of clams, forage fish, flatfish, and crabs. Bird use of the estuary was extensive, and grass meadows along the estuary banks provided unique and important black bear habitat. In contrast, the lower channel of Government Creek was essentially a steep bedrock chute with little substrate, with an estuary that was only about one fifth the size of the Airport Creek estuary.

When it became clear that the relocation of Government Creek might be the most feasible alternative for RSA expansion, the DOT&PF Design Group Chief recommended that an interdisciplinary team (IDT) be established to monitor and participate in the stream design process and requested agency participation. The resulting team consisted of representatives from the Alaska Department of Fish and Game (ADF&G), the U.S. Fish and Wildlife Service, the National Marine Fisheries Service, DOT&PF, and members of USKH's design and environmental documentation team, which included Shane Cherry for stream channel design and Jon Houghton for estuary design.

The first IDT meeting was held at KTN in December 2004. During the meeting, relocation objectives were developed with respect to (1) targeted salmon species and habitat types to be created in both the channel and estuary of the relocated Government Creek, and (2) the proper location and configuration of the new estuary. A reference reach including high-value habitat located upstream of the proposed relocation was selected to inform the design of the new channel, should the relocation option go forward. General support was expressed by the IDT for the Government Creek relocation alternative, and a number of required studies and analyses were outlined for presentation at a second IDT meeting to be held in February 2005.

At the February 2005 IDT meeting, the requested additional information was presented with the exception of geotechnical data, which had not yet been possible to acquire. By that time, the engineering and economic feasibility of the stream relocation alternative had also been established. Also, instream habitat and stream geomorphology had been characterized in both the existing channel and in the high-value upstream reach that was to be used as a design target. General relocation design issues were discussed, including whether or not the confluence of the main branch of Government Creek and its north tributary (informally referred to as the North Trib) should be designed to exclude certain species of salmon in order to protect resident trout species.

Two important topics discussed by the IDT in the February 2005 meeting were formalizing the goals of the relocation and developing metrics for measuring success. Goals of fish passage and habitat parity were agreed upon. However, because a relocation of this scale and type was unprecedented in Alaska, it was further agreed that the design target should be for higher-quality habitat than what currently exists. By setting the design target higher than the goal, the goal might still be achieved if the constructed channel failed to meet the design target. Further research efforts were requested, and the desire was expressed for the design to include access for modifications and corrections to the relocated stream channel and floodplain after initial construction.

After completion of the geotechnical and other additional studies, another IDT meeting was convened in April 2005 to discuss the results. The geotechnical data were inconclusive with regard to the geological characteristics of the proposed relocated channel. It was anticipated that the new channel would probably be constructed within both bedrock and glacial till. Different channel design characteristics would be required for the two material types, particularly with respect to the height and spacing of steps and riffles. After some discussion, there was agreement among IDT members that field design and engineering would be required during construction, and that the construction plans and bid documents would need to be developed accordingly. The timing of diversion of flow into the new channel was discussed and tentatively agreed upon by all represented agencies, and design objectives for protecting resident trout populations in tributaries were further clarified. Although there were still some lingering questions about existing conditions in the Government Creek estuary, it was agreed that enough information had been gathered and developed to move ahead with the environmental assessment (EA) for the project, with the alternative that included the relocation of Government Creek being put forth as the Proposed Action.

A draft EA was produced in the fall of 2005. The EA proposed a multiyear post-construction monitoring plan and commitments for adaptive management to make corrections and modifications to the relocated stream and estuary. The ADF&G expressed reservations about the Proposed Action, taking the position that Government Creek was more valuable as a fish stream to the ADF&G than Airport Creek, and formally recommended against the Proposed Action.

A meeting was held with IDT members in November 2005 to discuss the ADF&G's concerns and those of other IDT members and to further refine habitat goals for the relocated stream. The ADF&G reiterated their lack of support for the Proposed Action and voiced concerns that the new channel would be incised, have erosion concerns, and have lower habitat value than the existing channel.

Subsequently, the DOT&PF made firm commitments to develop specific field design measures during construction and to fund and implement a monitoring and adaptive management program to the extent allowed by FAA funding constraints. A "Finding of No Significant Impact" for the EA was signed by the FAA in January 2006.

Interdisciplinary team members were invited to participate in the development of the post-construction monitoring plan in December 2005. The intent of the monitoring plan was to outline evaluation criteria and methods to evaluate the success of the relocation efforts. Three meetings were proposed that would establish the estuary and stream habitat objectives and outline the field methods that would be used to measure success to meet the objectives.

A draft monitoring plan was developed by Pentec/Hart Crowser in March 2006 and modified and refined during the winter and spring of 2006 during a series of IDT meetings. Minor modifications to the monitoring plan occurred in the fall of 2006, with a final revised plan being accepted by IDT members in late September.

The final monitoring plan described the proposed action, its estimated effects, and the proposed mitigation measures. It then laid out the mitigation goals and objectives and the monitoring methods for the new estuary, the relocated main channel of Government Creek, and two affected tributaries.

In the summer of 2006, it became clear that the confluence of the relocated Government Creek channel with Boulder Creek would have a 20-foot elevation differential, which would form a significant fish passage barrier, effectively isolating the upper portion of the drainage from anadromous fish and resulting in a loss of available habitat for juvenile rearing. After consultation with the IDT, the project team decided to mitigate for the lost habitat by constructing side channels within the floodplain of the relocated channel of Government Creek. These channels would provide quiet water habitat for juvenile salmon, allowing them to avoid predators, rest, or wait out periods of high velocity flows. Having a pre-established rapport with the involved regulatory agencies allowed for a quick resolution to this problem.

The mitigation for the fish passage barrier and the inclusion of a heavy vehicle access road were addressed within a reevaluation of the EA that was approved in the fall of 2006. Final permits for the project were received in late 2006 without any significant complications, since regulatory agencies had been fully involved in the design process since the start of the project.

Construction began in the spring of 2007, and construction of the relocated channel of Government Creek was completed in August of that year. Construction of the lower portion of the North Trib was completed the following year.

Adaptive Management and Post-Construction Adjustments

Adaptive management provides a means of gathering additional information during and after construction and of adjusting the project as necessary to improve performance. Adaptive management can be used to manage risk and more efficiently achieve project performance goals. This approach requires clearly stated performance standards and contingency actions that can be deployed to achieve performance if needed. Adaptive management provides flexibility in managing risk.

The relocated portion of Government Creek was deliberately designed to be self-adjusting, consistent with natural channel and habitat formation processes. While the overall benefits of this approach are great, some channel adjustments can have short-term negative effects. Adaptive management was used to correct negative short-term effects of channel adjustment while allowing for the long-term ecological benefits of a naturally functioning dynamic stream and floodplain.

Since the completion of construction in August 2007, Government Creek has experienced a number of significant flows that have activated the floodplain and side channels. The flood-prone nature of the relocated channel is due in part to a combination of lower stream gradient and higher channel roughness resulting from a larger amount of large woody debris (LWD) in the relocated channel compared to the reference reach. The resulting floodplain activity is ecologically beneficial, leading to greater habitat complexity. Adaptive management activities included some minor repositioning of LWD pieces to help stabilize log jams and maintain sufficient water flow in the primary constructed channels.

The DOT&PF originally elected to rely primarily on gradual natural colonization of the floodplain by riparian vegetation, but after the first year of construction, this approach was modified as part of the adaptive management process. Seven vegetation islands were constructed in the floodplain during the summer of 2008 in areas that were thought to receive less frequent flooding. These vegetation islands consisted of large clumps of native vegetation and soil that were placed within protective barriers of logs and boulders.

One of the key performance requirements involved maintaining continuous fish passage through the new alignment. After the first winter season after construction, two bedrock steps had adjusted so that they impaired fish passage during low flows. Adaptive management included physically modifying those two locations to carve intermediate pools and steps into the bedrock to ensure fish passage.

Government Creek appears to be performing as intended several years after construction. All major segments of the project allow fish passage through their respective reaches and provide spawning habitat for substantial numbers of adult salmon. Additionally, all of the constructed reaches provide rearing habitat for both juvenile salmonids and resident fish species, and rearing has been amply demonstrated. Particularly high rearing use by juvenile coho was seen in the constructed middle and upper side channels. Adaptive management helped to ensure the ecological performance of the project and contributed to the ADF&G view that Government Creek represents "Mitigation Gone Right!" The ADF&G's positive appraisal of the constructed project was particularly important given their formal recommendation against the Proposed Action early in the early agency coordination process.

Example Project II - Nome Airport Runway Safety Area Improvements Project

The Nome Airport Runway Safety Area Improvements project is another example of how the EAC process can facilitate efficient resolution of complex issues in order to evaluate alternatives, determine a mutually acceptable path towards permitting a proposed action, and identifying construction sequencing schedules and project timing windows that benefit the resource as well as ensuring the success of the project. The feasibility of relocating the Snake River around future RSA expansions at the Nome Airport has major design and environmental constraints that make it a perfect candidate for the EAC process. Before formal agency scoping could commence and this project could be brought forward to the public, major environmental and design constraints needed to be determined.

The EAC process developed at Ketchikan has been working well for the Nome Airport project. The project leaders convened a series of meetings attended by key regulatory agencies, and design and environmental specialists. Major environmental and construction constraints and data gaps were quickly determined in the first interagency meeting and site visit. During subsequent meetings, the group worked together to identify possible effects on resources and resolve construction constraints in order to minimize those effects. Through that process, a discussion of mitigation of unavoidable effects has naturally evolved. The project is currently developing habitat features to incorporate into the design, effectively addressing concerns brought up in earlier meetings. Future meetings will focus on incorporating adaptive management and construction sequencing plans as the project moves forward into the permitting and NEPA documentation phase.

Example Project III - Cordova Airport Wildlife and Flood Hazard Mitigation Project

The Cordova Airport is situated on highly productive migratory bird habitat between two rivers. A large number of migratory waterfowl are attracted to the open water and edge habitat around the airport. The Cordova Airport Wildlife and Flood Hazard Mitigation Project was created with the EAC process in mind in order to determine a long-term solution to wildlife hazards and flooding safety issues at the airport. The team includes key agency stakeholders, wildlife hazard experts, local airport personnel, and design and environmental specialists to solve issues identified in the Cordova Airport Wildlife Hazard Management Plan.

The interagency team has met twice. The first meeting presented concept designs for reducing open water near the airport, relocating several streams away from runway surfaces, and raising the runway above the elevation of the 100-year flood. The second meeting focused on identifying a plan for developing NEPA documents and acquiring permits for the projects. A comprehensive mitigation plan will incorporate adaptive management and a phased path forward that is flexible enough to be completed over many construction seasons with different funding sources.

Summary

Major benefits of the early agency coordination approach include the ability to address agency and permitting concerns during preliminary design, and to incorporate minimization and mitigation measures into designs from the earliest stages. When mitigation is part of the design, and when regulatory agencies are involved from the beginning of the project, it is easier to determine a list of avoidance, minimization, and mitigation components even before the completion of the environmental document. Attaining agency buy-in early in the process facilitates timely NEPA document approval and moves permit applications quickly through the permitting process. Agreeing to adaptive management plans during the design process allows the project to stay on schedule and avoid project delays during permitting and construction by establishing contingency actions to address changes in conditions or performance. Perhaps the most valuable result of the EAC process is that it instills a sense of trust among the regulatory agencies and allows the design team to use their depth of knowledge to protect the affected resources. This sense of trust fosters collaboration and allows agencies and design teams to work together to meet the Federal mandates and successfully address the purpose and need of the project. Early Agency Coordination integrates environmental success into project success. This alignment builds a stronger team and benefits both the project and the environment.

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Data and Modeling

Hydrologic Response to Climate Change and Habitat Resiliency Illustrated Using Fine-Scale Watershed Modeling

Lorraine E. Flint, Alan L. Flint

Abstract

In the face of rapid climate change, predictions of landscape change are of great interest to land and resource managers who endeavor to develop long-term plans with the goal of maintaining biodiversity and ecosystem services and adapting to extreme changes in the landscape. Climate models, primarily exhibited as increases in air temperature, often support habitat modeling that predicts large-scale migrations, either northward or up in elevation, or extinctions of sensitive species. Current studies rely most dominantly on large spatial scale projections (>10 km) of changes in precipitation and air temperature that neglect the subtleties of topographic shading, geomorphic features of the landscape, and fine-scale differences in soil properties. Climatic parameters downscaled to fine scales has been tested previously with species distribution models and found to improve correlations of vegetation distribution with temperature. For this study, future climate projections were downscaled to 270 m and applied to a hydrologic model to calculate future changes in recharge, runoff, and climatic water deficit for basins draining into the northern San Francisco Bay, CA. Study results provide tools and information for land and resource managers to anticipate potential changes in water availability and landscape stresses for planning purposes.

We generated future watershed hydrology scenarios using a regional coupled climate-hydrology water balance model, Basin Characterization Model (BCM), that predicts water cycle fractions of runoff, recharge, evapotranspiration, streamflow, and changes in soil moisture storage. Primary BCM inputs consist of topography, soil composition and depth, parent geology, and spatially-distributed values (measured or estimated) for air temperature and precipitation. Model calibration is achieved by using historic precipitation and temperature as BCM inputs and comparing model estimates of discharge with streamflow measured at gages. Using estimates of future precipitation and air temperature derived from Global Circulation Models (GCMs) (two models, GFDL and PCM, for two emissions scenarios, A2 and B1) as model input, we describe observed variability over the 20th century and estimate watershed-scale hydrologic response to potential climate change scenarios for the 21st century. Results indicate that by the last 30 years of the 21st century, North Bay scenarios project average minimum temperatures to increase by 1.0°C to 3.1°C and average maximum temperatures to increase by 2.1°C to 3.4°C (in comparison to conditions experienced over the last 30 years, 1981–2010). Precipitation projections for the next century vary between GCMs (ranging from 2 to 15 percent wetter than the 20th century average), and temperature forcings increase the variability of modeled runoff, recharge, and stream discharge, and shift seasonal hydrologic cycle timing. For both high and low rainfall scenarios, by the close of the 21st century warming is projected to amplify late season climatic water deficit (a measure of drought stress on landscapes) by 8–21 percent, however, fine-scale modeling can be used to illustrate local scale resiliency of habitats to climate change. Hydrologic variability within a single river basin demonstrated at the scale of subwatersheds may prove an important consideration for water managers in the face of climate change, and regional fine-scale modeling can provide insights of landscape response from the watershed to local scale for the purposes of resource planning in the face of a changing climate.

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Simulation of Hydrologic Response to Climate and Landscape Change Using the Precipitation Runoff Modeling System in the Apalachicola–Chattahoochee–Flint River Basin in the Southeastern USA

Jacob H. LaFontaine, Lauren E. Hay, Roland J. Viger, Steven L. Markstrom, R. Steven Regan

Abstract

The U.S. Geological Survey (USGS) Southeast Regional Assessment Project (SERAP) was initiated in 2009 to help environmental resource managers assess the potential effects of climate change on ecosystems. One component of the interdisciplinary SERAP program is the development and calibration of a set of multiresolution hydrologic models of the Apalachicola–Chattahoochee–Flint (ACF) River Basin. The ACF River Basin, home to numerous fish and wildlife species of conservation concern, is regionally important for water supply as well as the focus of complementary environmental and climate change research. Hydrologic models of varying spatial extents and resolutions are required to address varied local-to-regional water resource management questions as required by the scope and limitations of potential management actions. In the ACF River Basin, these models (coarse and fine scales) were developed using the USGS Precipitation Runoff Modeling System (PRMS). The coarse-scale model has a contributing area of approximately 50,700 square kilometers. Six fine-resolution models, ranging in size from 396 to 2,690 square kilometers, are nested within the coarse-scale model and have been developed for the following basins: the upper Chattahoochee, Chestatee, and Chipola Rivers and the Ichawaynochaway, Potato, and Spring Creeks. Both coarse- and fine-scale models simulate basin hydrology using daily timesteps, measured climatic data, and basin characteristics, such as land cover and topography. Measured streamflow data are used to calibrate and evaluate computed basin hydrology. Being able to project future hydrologic conditions for this set of models will rely on the use of land cover projections in conjunction with downscaled global climate model results.

Keywords: hydrology, hydrologic modeling, climate change, model optimization

Introduction

To help environmental resource managers assess potential effects of climate change on ecosystems, the U.S. Geological Survey (USGS) Southeast Regional Assessment Project (SERAP) is developing regional models and other science tools (Dalton and Jones 2010). Models and data produced by SERAP will be used in a collaborative process between the USGS, U.S. Fish and Wildlife Service, State and Federal partners, nongovernmental organizations, and academia. Integration of the models developed by SERAP is shown in Figure 1. One component of the SERAP is development and calibration of a set of multiresolution hydrologic models, as highlighted in Figure 1, of the Apalachicola–Chattahoochee–Flint (ACF) River Basin. The ACF River Basin (Figure 2), regionally important as a source of water supply, contains numerous fish and wildlife species of conservation concern; as a result, the ACF River Basin is a focus of complementary environmental and climate change research. Hydrologic models of varying spatial extents and resolutions are required to address varied local-to-regional water resource management questions as required by the scope and limitations of potential

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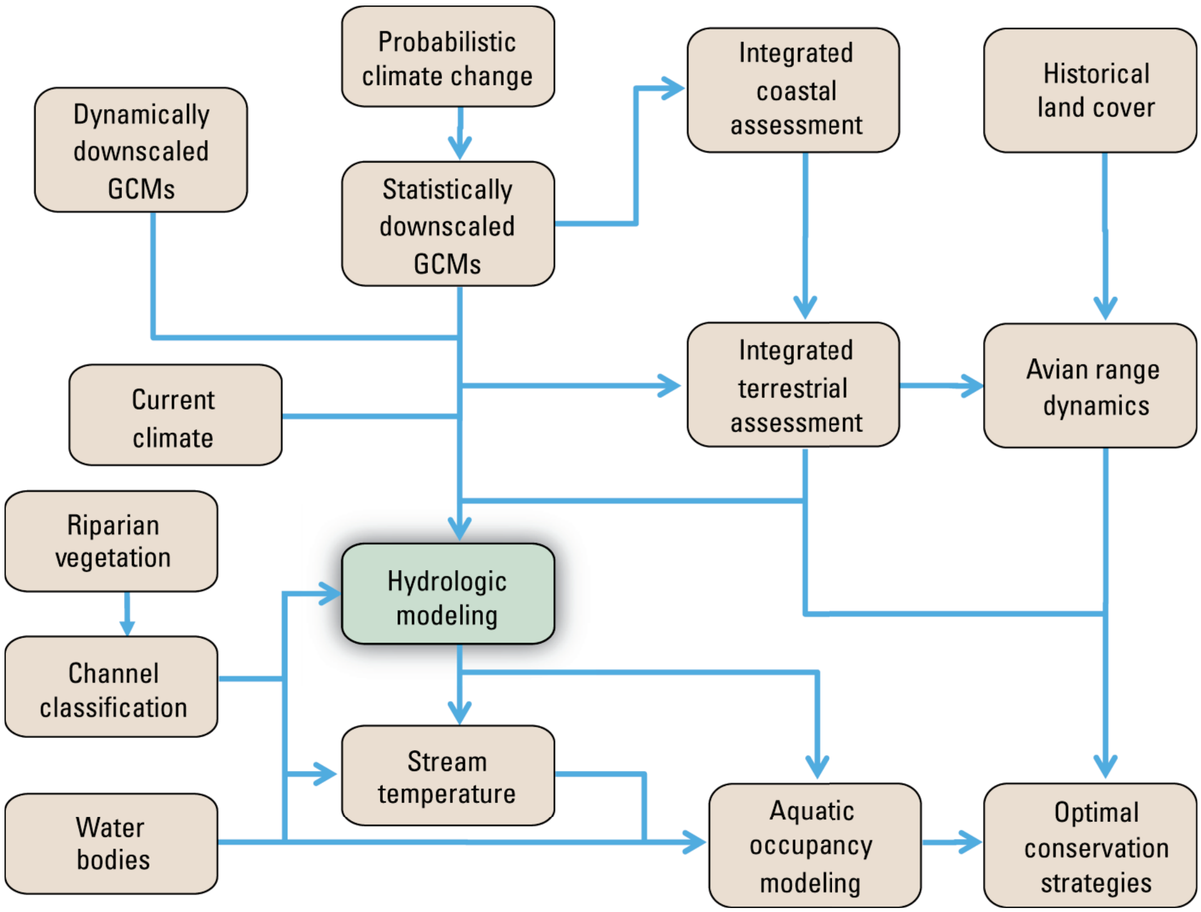

Figure 1. Information flow diagram for the Southeast Regional Assessment Project. The hydrologic modeling team receives information from several project teams and provides output that will be used as input by the stream temperature and aquatic occupancy models. [GCMs, global climate models]

management actions. These models were developed using the USGS Precipitation Runoff Modeling System (PRMS). A coarse-scale hydrologic model of the ACF River Basin (approximately 50,700 square kilometers [km^2]) and six fine-scale models (ranging in size from 396 to 2,690 km^2) were developed as part of this study to simulate natural streamflow in the basin and to compare simulated streamflows at various model scales using climate data from both point and gridded sources. The models simulate natural streamflow for the period 1950–1999 based on a daily timestep.

Description of Study Area

The ACF River Basin includes three major rivers—the Apalachicola, Chattahoochee, and Flint Rivers (Figure 2). The Chattahoochee River begins in the mountains of northeast Georgia and flows southwest through metropolitan Atlanta to the Alabama-Georgia border, where the river flows south to Lake Seminole at the Florida-Georgia border. The Flint River begins in north-central Georgia, just south of Atlanta, and flows south to Lake Seminole. The Apalachicola River begins at Lake Seminole, the confluence of the Chattahoochee and Flint Rivers, and flows south through Florida to the Gulf of Mexico. The Chattahoochee River is highly regulated by four U.S. Army Corps of Engineers reservoirs and several run-of-the-river dams, while the Flint River is relatively unregulated with just two run-of-the-river dams. The lower Flint River Basin also has substantial agricultural land use, for which groundwater and surface water are used for irrigation. The ACF River Basin includes the Blue Ridge, Piedmont, and Coastal Plain Physiographic Provinces (Figure 2). In the ACF River Basin, the Blue Ridge and Piedmont Physiographic Provinces are underlain with crystalline rock; the Coastal Plain Physiographic Province is underlain with sedimentary rocks and unconsolidated sediments (Couch and others 1995).

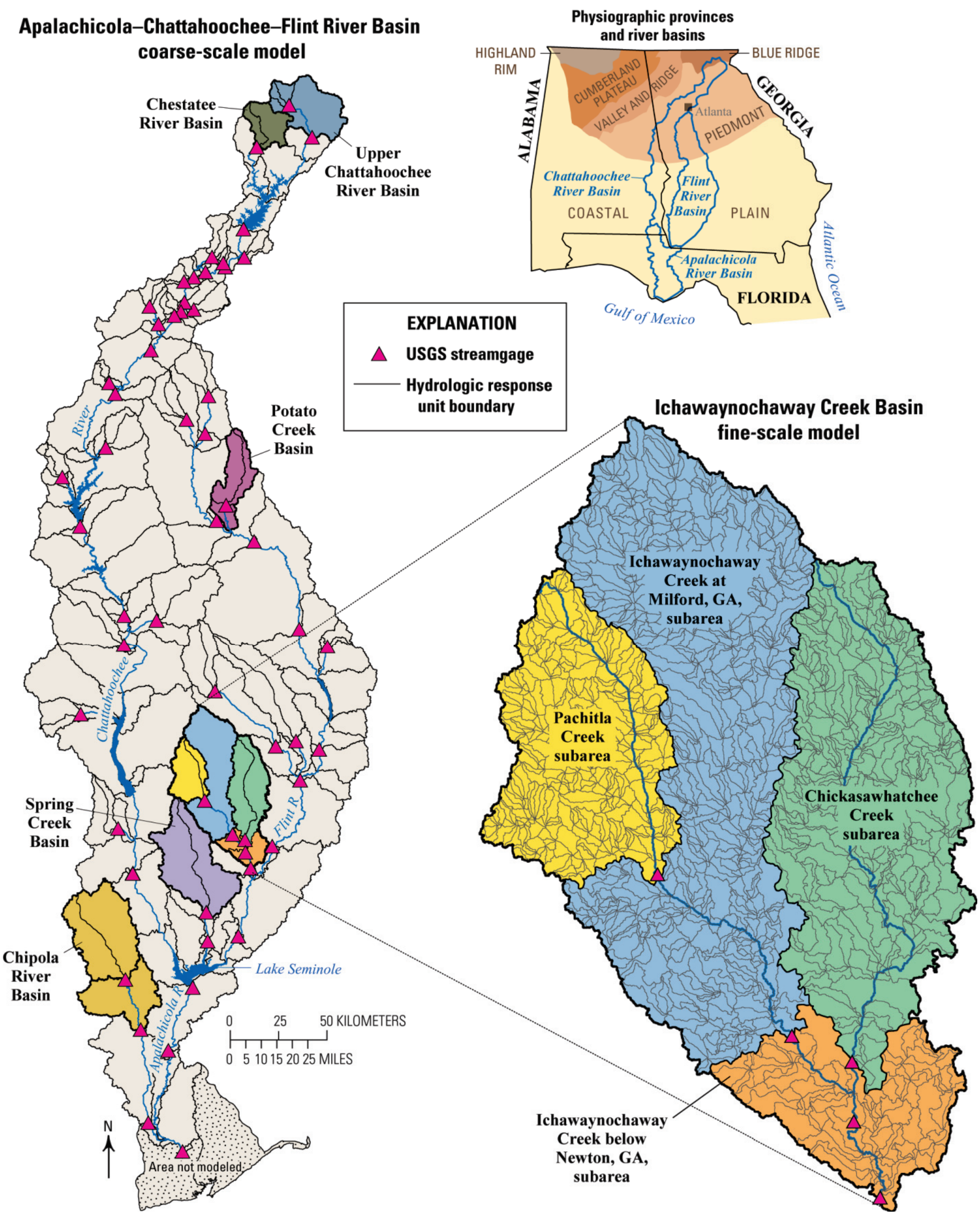

Figure 2. Example of a fine-scale hydrologic model nested within the coarse-scale hydrologic model. Ichawanochaway Creek is located in the lower Flint River Basin and comprises 2,690 square kilometers (km^2). Each of the coarse-scale hydrologic response units (HRUs) in the fine-scale model areas are split into smaller HRUs that provide detailed flow information used by the aquatic occupancy modeling shown in Figure 1. The HRUs range in size from 5 km^2 to 1,900 km^2 for the coarse-scale model and from <0.1 km^2 to 17 km^2 for the fine-scale models.

Description of Precipitation-Runoff Models

The ACF River Basin was modeled using a nested approach composed of a coarse-scale model of the entire basin and six fine-scale models for subbasins of interest (Table 1). The coarse-scale model simulates streamflow to address regional hydrologic questions and provides a regional framework for identifying future fine-scale models in the basin. The fine-scale models simulate streamflow at more points in a given subbasin than the coarse-scale model. In the collaborative process, aquatic occupancy modeling (Figure 1) makes use of the detailed streamflow information simulated by the fine-scale hydrologic models to simulate the presence and persistence of fishes and mussels.

Precipitation-Runoff Modeling System

PRMS is a deterministic, distributed-parameter, physical-process based hydrologic model (Leavesley et al. 1983). There are four primary objectives of this modeling system: (1) simulation of land-surface hydrologic processes, including evapotranspiration, runoff, infiltration, interflow, snowpack, and soil moisture on the basis of distributed climate information (temperature, precipitation, and solar radiation); (2) simulation of hydrologic water budgets at the watershed scale with temporal scales ranging from days to centuries; (3) integration with models used for natural-resource management or other scientific disciplines; and (4) creation of a modular design that allows the selection of alternative hydrologic process algorithms from either the standard module library or user-provided provisional modules.

Delineation and Parameterization

Typically, the delineation of a PRMS hydrologic model is done by overlaying a USGS digital elevation model (DEM) with the study basin using the geographical information system (GIS) Weasel software developed by Viger and Leavesley (2007). The DEM was used to develop the modeled stream network and divide the basin into a series of areas called hydrologic response units (HRUs; Figure 2). HRUs simulate the hydrologic response of the basin (streamflow) to air temperature and precipitation. The stream network is used to route streamflow from the HRUs through the basin. Initially, HRUs were delineated based on the stream network, a maximum area threshold, and changes in elevation from the DEM. These delineations were further refined by including points of interest in the basin, such as locations of streamflow gages, minimum flows, sampling sites, etc. Once the stream network and HRUs were defined, the GIS Weasel software was used to parameterize the model by using terrain, soil, land-cover, impervious-area, and vegetation data. Included in the land cover category is a GIS coverage of surface depressions in the basin. Large numbers of these relatively small waterbodies can have substantial hydrologic effects on streamflow. The simulation of these surface depressions, as discussed in Viger et al. (2009), was used in this set of hydrologic models. Each segment in the stream network and each HRU were treated as a homogeneous entity with parameters that represent an aggregation of the information contained in the data coverages used.

Table 1. Description of coarse- and fine-scale hydrologic models in the Apalachicola–Chattahoochee–Flint River Basin.

Model	Drainage area (square kilometers)	Number of USGS streamgages	Number of stream segments	Hydrologic response unit	Physiographic province
Coarse-scale model					
Apalachicola–Chattahoochee–Flint River Basin	50,700 (approx.)	57	128	258	Blue Ridge, Coastal Plain, Piedmont
Fine-scale model					
Upper Chattahoochee River	815	2	328	600	Blue Ridge, Piedmont
Chestatee River	396	1	168	312	Blue Ridge, Piedmont
Chipola River	2,020	2	778	1,472	Coastal Plain
Ichawaynochaway Creek	2,690	5	824	1,542	Coastal Plain
Potato Creek	616	1	221	427	Piedmont
Spring Creek	1,260	1	345	674	Coastal Plain

Climate Data

PRMS requires the input of daily maximum and minimum air temperatures and daily precipitation data. Climate station data provided by the National Weather Service Cooperative Observer Program (National Oceanic and Atmospheric Administration 2010) typically are used for PRMS models. Initially, 79 of these climate stations were used for the coarse-scale hydrologic model. The model was then adjusted to use climate data from one-eighth-degree (about 12 kilometers [km]) gridded products developed for the conterminous United States by Maurer et al. (2002). A Web-based GIS interface called the Geo Data Portal was then used to spatially transfer the gridded climate data to the model HRUs. The gridded climate data currently available are for the period 1950–1999; however, the gridded climate data and projections are being downscaled for the period 2000–2099 by using statistical and dynamical procedures.

Streamflow Data

The USGS streamflow-gaging network (http://waterwatch.usgs.gov/) was used to obtain daily-flow data for calibration and evaluation of the hydrologic models. For this study, 57 streamgages were selected based on a minimum drainage area of 25 km^2 and a minimum of 10 years of daily-flow record. The spatial distribution of the selected streamgages is shown in Figure 2. Streamflow data are retrieved and formatted for the hydrologic models by using Downsizer, a graphical user interface (GUI) developed by Ward-Garrison and others (2009).

Nested Modeling Approach

Efficient development and interpretation of hydrologic models for the SERAP required that models of varying spatial scales be developed. A single, fine-scale model of the whole ACF River Basin would be time and cost prohibitive. For computational efficiency, the coarse-scale model was developed to (1) represent the overall water balance and hydrologic processes of the system and (2) provide a regional framework for fine-scale hydrologic models. The calibrated coarse-scale model also can provide initial pre-calibration values for some parameters used in the fine-scale models.

By using the coarse-scale HRUs and the stream network, selected fine-scale basins were delineated so the fine-scale HRUs nest within the coarse-scale HRUs, and the fine-scale stream-segment nodes include the coarse-scale stream-segment nodes (Figure 2). By matching the fine- and coarse-scale model delineations, direct comparisons can be made across model scales. In the event that the calibrated fine-scale models outperform the coarse-scale model, outputs from the fine-scale models can be used to refine the coarse-scale model.

Hydrologic Models

Seven hydrologic models were developed for the ACF River Basin—one coarse-scale basinwide model and six fine-scale models. The six fine-scale models simulate two subbasins (upper Chattahoochee and Chestatee Rivers) in the northern part of the ACF River Basin, one subbasin (Potato Creek) in the central part of the basin, and three subbasins (Chipola River and Ichawaynochaway and Spring Creeks) in the southern part of the basin (Figure 2). The subbasins for which fine-scale models were developed were selected based on representing the different physiographic provinces in the ACF River Basin, current and projected urbanization, and critical areas of ecological habitat. The upper Chattahoochee River, Chestatee River, and Potato Creek subbasins are relatively undeveloped in terms of urbanization and agriculture, whereas the Chipola River and Ichawaynochaway and Spring Creek subbasins are heavily developed by agriculture.

The models were calibrated using Luca software, a multi-objective, stepwise, wizard-style graphic user interface (GUI; Hay et al. 2006, Hay and Umemoto 2006). This GUI uses the Shuffled Complex Evolution (Duan et al. 1993) global search algorithm to calibrate parameters for PRMS hydrologic models. A procedure has been developed to calibrate each model by using the following variables: (1) mean monthly solar radiation, (2) mean monthly potential evapotranspiration, (3) annual and monthly flows, (4) timing of daily flows, (5) magnitude of high-flow days, and (6) magnitude of low-flow days. Model parameters were adjusted to optimize the simulation of these six variables for historical climate and streamflow data for the period 1990–1999. Once the models were calibrated, they were evaluated using historical climate and streamflow data for the period 1980–1989. Plots of annual-, monthly-, and daily-flow statistics were used to evaluate the accuracy of the model simulations.

This suite of hydrologic models can be used to study the effects of changing climate and landscape on the hydrologic response of the ACF River Basin. The hydrologic modeling output can also be used as input for stream temperature and aquatic occupancy modeling being done by others in the SERAP collaborative process (Figure 1).

Summary

Multiresolution hydrologic models of the Apalachicola–Chattahoochee–Flint (ACF) River Basin, developed as part of the Southeast Regional Assessment Project (SERAP), are helping assess the potential effects of climate change on ecosystems. Hydrologic models of varying spatial extents and resolutions were developed to address varied local-to-regional water resource management questions as required by the scope and limitations of potential management actions. Seven models were developed by using PRMS.

A coarse-scale model for the ACF River Basin, with a contributing area of approximately 50,700 km^2, is coupled with six fine-scale subbasin models, ranging in size from 396 to 2,690 km^2, for the upper Chattahoochee, Chestatee, and Chipola Rivers and the Ichawaynochaway, Potato, and Spring Creeks. These subbasins were selected based on representation of the different physiographic provinces, current and projected urbanization, and critical areas of ecological habitat. All of the models simulate basin hydrology for the period 1950–1999 using a daily timestep, measured climate data, and basin characteristics, such as land cover and topography. Measured streamflow data from 57 USGS streamgages were used to calibrate and evaluate computed basin hydrology. The ability to project future hydrologic conditions for this set of models will rely on the use of land cover projections in conjunction with downscaled global climate model results.

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Geomorphology and Watershed Management

Temporal and Spatial Distribution of Landslides in the Redwood Creek Basin, Northern California

M.A. Madej

Abstract

Mass movement processes are a dominant means of supplying sediment to mountainous rivers of north coastal California, but the episodic nature of landslides represents a challenge to interpreting patterns of slope instability. This study compares two major landslide events occurring in 1964–1975 and in 1997 in the Redwood Creek basin in north coastal California. In 1997, a moderate-intensity, long-duration storm with high antecedent precipitation triggered 317 landslides with areas greater than 400 m^2 in the 720-km^2 Redwood Creek basin. The intensity-duration threshold for landslide initiation in 1997 was consistent with previously published values. Aerial photographs (1:6,000 scale) taken a few months after the 1997 storm facilitated the mapping of shallow debris slides, debris flows, and bank failures. The magnitude and location of the 1997 landslides were compared to the distributions of landslides generated by larger floods in 1964, 1972, and 1975. The volume of landslide material produced by the 1997 storm was an order of magnitude less than that generated in the earlier period. During both periods, inner gorge hillslopes produced many landslides, but the relative contribution of tributary basins to overall landslide production differed. Slope stability models can help identify areas susceptible to failure. The 22 percent of the watershed area classified as moderately to highly unstable by the SHALSTAB slope stability model included locations that generated almost 90 percent of the landslide volume during the 1997 storm.

Keywords: landslides, rainfall intensity duration, slope stability model

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Introduction

Mass movement processes are a dominant means of supplying sediment to mountainous rivers of north coastal California, but the episodic nature of landslides represents a challenge to interpreting patterns of slope instability. Natural factors, including bedrock geology, topography, and vegetation, influence landslide occurrence. Many anthropogenic factors, such as logging and road construction, also affect slope stability (Swanston and Swanson 1976, Sidle and Wu 2001). These factors commonly do not come into play until a major storm occurs and high rainfall triggers landslide processes, including shallow debris slides and debris flows. It would be useful to know what type of rainfall conditions initiate landslides in this region and whether areas that exhibited landslide activity previously are susceptible to further mass movement in subsequent storms. Rainfall-triggering events were described by Caine (1980), who suggested a threshold of landslide initiation on the basis of a combination of intensity and duration for 73 rainfall events. More recently, Guzzetti et al. (2008) updated this threshold on the basis of 2,626 rainfall events. To date, landslide occurrence in north coastal California has not been evaluated in terms of these thresholds.

Landslide hazard assessment may use multivariate correlations between mapped landslides and landscape attributes or may employ a physics-based approach (Montgomery et al. 1998). The physical models of slope stability use information on topographic convergence and slope steepness to identify potentially unstable sites. One such model is SHALSTAB (Montgomery and Dietrich 1994), which has been used to assess the relation between forest clearing and shallow debris sliding in Oregon and Washington (Montgomery et al. 2000).

The objectives of this study are to compare the spatial distribution of landslides in the Redwood Creek basin during two periods of high landslide activity, to relate the timing of landslide occurrence to precipitation patterns, and to assess the use of a slope stability model in predicting landslide occurrence.

Field Setting

Redwood Creek drains a 720-km^2 watershed in north coastal California and empties into the Pacific Ocean near Orick, CA. The basin is underlain primarily by Cretaceous and Late Jurassic metamorphic and sedimentary units of the Franciscan assemblage (Harden et al. 1982). Massive sandstones and interbedded sandstones and mudstones of the Coyote Creek and Lacks Creek units are on the east side of the watershed, whereas the west side is dominated by the Redwood Creek schist. The contrasting lithologies are separated by the highly sheared, slightly metamorphosed sandstone and mudstone of the Grogan Fault zone. Redwood Creek generally follows the trace of the fault, establishing a 100-km-long watershed with an average width of only 9 km. For about 50 km of its length the river flows through an inner gorge, where slopes that are adjacent to the stream channel are steeper than those farther upslope and are separated by a distinct break-in-slope (Kelsey 1988). Basin relief is 1,615 m and mean slope in the watershed is 25 percent. Mean annual rainfall in the watershed varies by location, ranging from about 2,500 mm at the headwaters to 1,520 mm near the mouth (Iwatsubo et al. 1975). Summers are dry, and almost all rainfall occurs between October and April. Snow is often present for several months at the highest elevations but is uncommon below 800 m.

Old-growth redwood (*Sequoia sempervirens*) and Douglas-fir (*Pseudotsuga menziesii*) originally dominated the downstream third of the Redwood Creek basin. Farther inland, Douglas-fir was the dominant conifer on the hillslopes, except for grasslands and oak woodlands on the drier ridges on the eastern divide. Intensive logging began in the watershed in the late 1940s, with most activity in the Douglas-fir forests of the middle and upper basin. Logging of redwoods accelerated in lower Redwood Creek in the early 1960s, and about 37 percent of the basin had been logged by the time of a large flood in 1964 (Best 1995). Most of the area was clearcut logged in large units and yarded using tractors, which resulted in extensive ground disturbance and a high density of roads. Stands of old-growth redwood remained in the lower basin, and in 1968 about 124 km^2 of the lower basin was acquired to form Redwood National Park (RNP). By 1978 about 80 percent of the original coniferous forest had been logged. At that point, RNP boundaries were expanded to include an additional 150 km^2 of recently logged land that would be restored and managed as a protective buffer for the old-growth stands.

The Redwood Creek watershed has been subjected to a sequence of major floods since logging began. Since 1953, the U.S. Geological Survey (USGS) has operated a stream gauging station near the mouth of Redwood Creek at Orick, CA, where stream discharge and suspended sediment are measured. The 1964 flood attained the largest discharge on record (1,430 m^3/s, a 50-year recurrence interval) and triggered the most landsliding (Harden 1995, Kelsey et al. 1995, Pitlick 1995). Additional floods of 1955, 1972, and 1975 all exceeded 1,390 m^3/s. The 1997 flood was the largest since 1975 and peaked at 1,140 m^3/s, representing a 10-year flow.

Surficial failures generated by the storms included debris slides and debris flows. Debris slides are rapid, shallow, dominantly translational slope failures along nearly planar surfaces, whereas debris flows are rapid failures along narrow, well defined tracks which show signs of flowing. In localized areas, Redwood Creek undercut adjacent hillslopes, resulting in large streambank failures. Earthflows, slow-moving complexes of slumping, flowing, and translational sliding that produce hummocky and lobate topography, are present in the Redwood Creek basin but were not quantified in the present study.

Methods

This study uses data on landslide distribution from several principal sources. Earlier studies of landslides in the Redwood Creek basin (Kelsey et al. 1995, Pitlick 1995) included extensive field measurements of landslide length, width, and depth on about 1,200 streamside landslides along the mainstem and in 16 of the largest tributaries. These studies provided the basis for comparison with more recent landsliding. Curry (2007) mapped the distribution of landslides generated by the January 1997 storm on 1:6,000 monochrome aerial photographs taken in June 1997, and the maps were updated for this study with further photographic examination. Landslide locations were transferred to orthophotos and then digitized. Landslide volumes were estimated from scar areas using a relation between area and volume constructed from previous field measurements of landslide dimensions in the basin (Kelsey et al. 1995, Pitlick 1995) and assuming the average hillslope gradient of 55 percent applies to the scar site. Mean depths for debris flow tracks with areas greater than 3,000 m^2 were assumed to be 1.2 m on the basis of field observations. The Prairie Creek basin, a tributary basin near the mouth of Redwood Creek that is primarily covered with old-growth redwood forests, was not included in the earlier landslide studies so is not addressed in this update. The present comparison of landslides is restricted to landslides larger than 400 m^2 on the basis of a comparison of sizes of field-mapped landslides with the ability to detect them on 1:6,000 air photos.

Precipitation data for various time intervals for the 1997 storm are available from a tipping bucket gauge on the eastern divide of Redwood Creek at an elevation of 750 m. Several other nearby rain gauges were plagued with equipment problems, resulting in data gaps. Equivalent ridge-line data are not available for earlier storms, but storm totals near the mouth of Redwood Creek (station elevation of 50 m) for 1964 were compiled by Harden (1995).

The SHALSTAB slope stability model (Montgomery and Dietrich 1994) previously had been applied to the Redwood Creek basin on the basis of 10-m digital elevation models (DEMs) (Hare 2003). A stability rating, ranging from unconditionally stable to unconditionally unstable, was computed for each DEM cell on the basis of log (q/T), where T is the soil transmissivity and q is the effective precipitation (Montgomery and Dietrich 1994). To compare actual landslide locations with predicted unstable areas, the landslide maps were overlain onto a map of log (q/T) values. The stability of a landslide location was classified according to the least stable value of log (q/T) that was touched by the landslide scar, assuming that the least stable cell controlled the stability of the landslide.

Results

Following the 1997 storm, 317 landslides >400 m^2 in area were mapped in the Redwood Creek basin. Cumulatively these landslides mobilized 596,000 m^3 of material, of which about 441,000 m^3 was delivered to stream channels on the basis of air photo interpretation. Shallow debris slides (n = 157, median volume of 1,280 m^3) accounted for about half the landslide features and 43 percent of the total landslide volume. Debris flows were slightly fewer in number (n = 147) but were somewhat larger (median volume of 1,870 m^3) and accounted for 54 percent of the landslide volume. Only 13 streambank failures >400 m^2 in area were identified (median volume of 1,060 m^3). The volume of landslide material delivered to Redwood Creek and its tributaries during the 1997 storm was an order of magnitude less than the 4,000,000 m^3 estimated to have been produced by storms between 1954 and 1978, of which most is likely to date from the 1964 storm (Kelsey et al. 1995).
In 1997, 20 percent of the landslides accounted for 56 percent of the total volume (Figure 1). The maximum size was 28,000 m^3. Large landslides also dominated the size distribution in earlier periods, when 20 percent of the landslides contributed 72 percent of the volume and the largest landslide was 86,000 m^3. The smallest 20 percent of the landslides only contributed 4 percent of the landslide volume in 1997 and 2 percent in the earlier period.

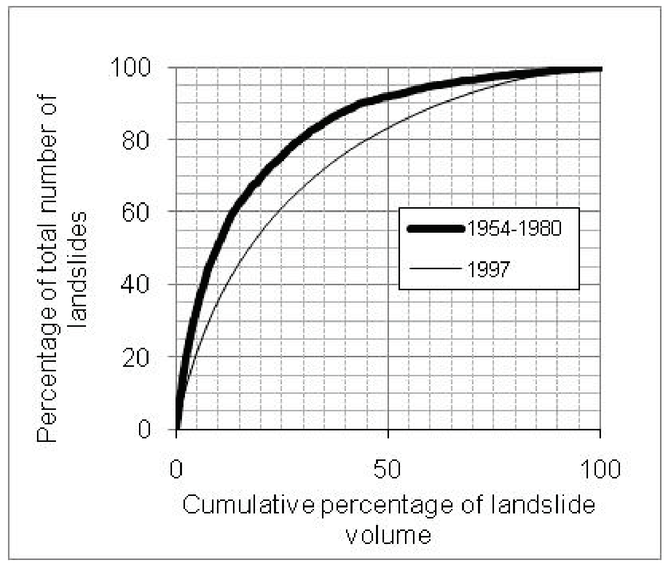

Figure 1. Cumulative volume-frequency relations for landslides in the Redwood Creek basin during two time periods.

The spatial distribution of landslide occurrence between the two time periods of 1954–1978 and 1997 is compared in Figure 2. In general, although the earlier time period produced an order of magnitude greater landslide volume, two inner gorge reaches along the mainstem of Redwood Creek identified as high landslide input areas in the earlier study (0–18 km, and 60–70 km from the headwaters) also had high landslide activity in 1997. In contrast, some tributary basins that had many active landslides in earlier periods were relatively inactive in 1997. Note that large tributary inputs are indicated by steep steps in the cumulative volume profile in Figure 2. For example, the reach between Copper Creek and Bridge Creek had high landslide activity in the earlier period but was relatively quiescent in 1997. Windy, Minor, and Copper creeks were high sediment producers during the period of 1954–1978 but proportionately were not as great in 1997. The rates of timber harvest and road construction were lower in recent years, and most of the logging roads in the Copper Creek basin had been decommissioned in the mid 1980s, well before the 1997 storm. As a fraction of total landslide volume, Bridge and McArthur creek watersheds, where several large road-associated debris flows occurred, were relatively larger landslide producers in 1997 than in the earlier period.

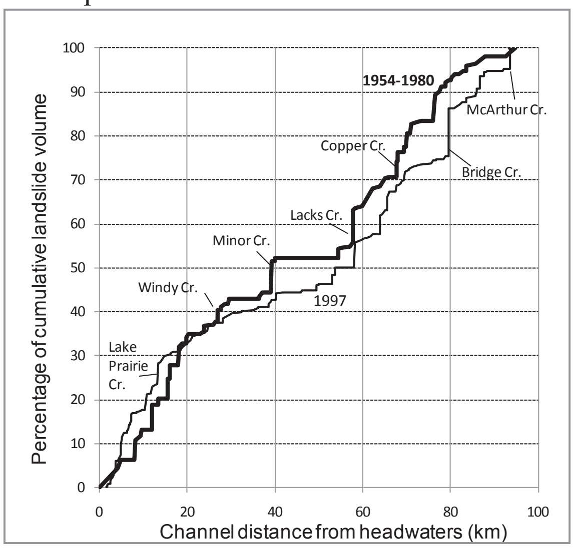

Figure 2. Comparison of volume of landslide material, cumulative over distance downstream along mainstem Redwood Creek, contributed for two time periods.

To assess whether the landslide-producing storms met the threshold rainfall values proposed by previous studies, data from Redwood Creek were compared to previously published relationships (Figure 3). For the 1- to 48-hour durations, rainfall in the Redwood Creek basin slightly exceeded the lower threshold defined by Caine (1980) and significantly exceeded the threshold for cool, Mediterranean climates defined by Guzzetti et al. (2008). Only the 15-minute rainfall duration did not have a corresponding intensity that exceeded the minimum threshold. The values for the 1964 storm also were consistent with the Caine relationship. The Guzzetti threshold for a cool, Mediterranean climate has a similar slope as Caine's but predicts the likely occurrence of shallow failures at lower rainfall intensities. Caine's threshold was established for catastrophic slope failures, whereas the Guzzetti threshold may represent a minimum boundary for smaller landslides. In the case of the Redwood Creek watershed, we have observed occasional failures during years with rainfall intensities or durations less than Caine's threshold, but widespread landsliding did not occur until Caine's threshold was exceeded.

Areas predicted to be unstable through the SHALSTAB slope stability model were compared to the volume of landslide material mobilized during the 1997 storm (Table 1). High, moderately high, and moderate potential for instability are represented by areas where log (q/T) is less than or equal to -3.1, -2.8, and -2.5, respectively (Montgomery et al. 1998). These stability classes accounted for 22 percent of the watershed area and included 85 percent of the number of sites that failed and 87 percent of the volume of material mobilized by landslides during the 1997 storm.

Discussion

A few large landslides account for the bulk of hillslope material eroded in widespread landsliding events. In the earlier period, massive inner gorge debris slides as large as 86,000 m^3 stripped the hillslopes up to the break-in-slope. Although the inner gorge also produced many moderately large debris slides in 1997, the largest landslides at that time were debris flows originating from road fill failures on unpaved forest roads. It is possible that the earlier events removed the most unstable material in the inner gorge so that the supply of weathered colluvium available to fail in subsequent storms was diminished. The present focus of RNP and other basin landowners on decommissioning abandoned roads may decrease the number of road failures in future storms, but this erosion control work has not yet been "tested" by storms of the size seen in 1964, 1972, and 1975.

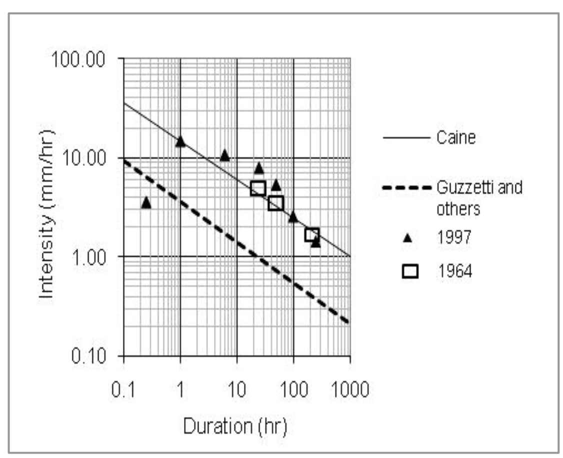

Figure 3. Rainfall intensity-duration for Redwood Creek storms and from other areas.

Table 1. Distribution of landslides and landslide scar volume by stability classes defined by the SHALSTAB slope stability model.

Stability class	Value range, $\log_{10}(q/T)$	Percent of watershed area[1]	Percent of slides	Percent of slide volume
Stable or low instability	> –2.2	71.2	9.3	8.7
Moderately low	–2.5 to –2.2	6.7	5.6	4.2
Moderate	–2.8 to –2.5	10.7	17.8	28.6
Moderately high	–3.1 to –2.8	6.4	21.5	19.9
High instability	< –3.1	4.8	45.8	38.5

[1]Redwood Creek watershed, excluding Prairie Creek.

The initiation of widespread landsliding in the Redwood Creek basin seems to coincide with the intensity-duration threshold of initiation proposed by Caine (1980). The values of rainfall intensity-duration for the 1964 storm measured at an elevation of 50 m were less than those for the 1997 storm, measured at 630 m. It is very likely that the 1964 values were much higher at upper elevations owing to orographic effects but no high elevation rain gauges were present in the Redwood Creek basin at that time. In recent years more automated weather stations have been installed in the basin, which will help assess rainfall trends in future storms. Other factors in landslide initiation, such as land use and antecedent precipitation, probably also play roles. The 60-day antecedent precipitation index associated with the beginning of the 1997 storm, at 136 mm, was the fifth highest of the last seven flood-producing storms, previously analyzed by Harden (1995). Future research will examine the relationships among landslide occurrence, bedrock geology, timber harvest, and roads.

The slope stability model SHALSTAB performed well in identifying susceptible terrain producing large volume landslides in the Redwood Creek watershed. SHALSTAB was designed to model the susceptibility of terrain to shallow debris slides (Montgomery and Dietrich 1994). In the Redwood Creek basin, the model was able to account for 75 percent and 90 percent of the volume of shallow debris slides by classifying the most unstable 10 percent and 20 percent of the basin, respectively, which is similar to results obtained from other northern California watersheds (Dietrich et al. 2001). Because the 10-m DEMs underestimate slope steepness at the bottom of narrow, V-shaped valleys, the SHALSTAB results are conservative. Although not designed to predict debris flow locations, the model also accounted for debris flow volumes with accuracies similar to those for shallow debris slides. Failures from both roads and harvest areas are more likely in steep, convergent areas, which SHALSTAB was able to detect from 10-m DEMs. Failures can, of course, also occur on slopes classified as stable or moderately stable if roads, landings, or timber harvest alter the hydrologic regime or mass balance of materials at a site. SHALSTAB can be used as a tool by land managers in timber harvest plan reviews or other planning activities to highlight areas of potential slope instability, and these areas can then be targeted for more detailed field investigations.

Conclusion

A large storm in January 1997 initiated 317 landslides in the Redwood Creek basin in north coastal California. In comparison to previous large storms, the 1997 event produced an order of magnitude less landslide volume. A few large landslides accounted for the bulk of the sediment volume, as 20 percent of the landslides produced 56 percent of the total landslide volume. Inner gorge slopes had a large number of landslides in earlier storms, and this pattern also held in the 1997 storm. Nevertheless, the largest landslides in 1997 were debris flows originating on forest roads. The rainfall intensity-duration threshold proposed by Caine (1980) held true for landslide initiation in 1997. The slope stability model, SHALSTAB, also performed well in predicting locations of landslide occurrence. The 22 percent of the watershed area classified as moderately to highly unstable by the model included locations generating almost 90 percent of the landslide volume during the 1997 storm.

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Holocene Extraordinary Floods and the Social Impacts in the Weihe River Basin, China

Chunchang Huang, Jiangli Pang

Abstract

Holocene Palaeoflood slackwater deposits (SWD) were often found inserted in the loess-soil stratigraphy along the Weihe River valley near Xi'an in the middle Yellow River basin. These compacted silt or clayey silt beds were deposited from the suspended sediment load in the floodwater. They have recorded the extraordinary palaeoflood events during the Holocene. Typical SWD beds were identified at four sites during field investigations in the last three years. The SWD beds have recorded large Holocene floods that occurred between 3,200 and 3,000 years before present (B.P.) when the mid-Holocene Climatic Optimum was coming to a close. Using the end point of the SWD as the minimum flood stage, the minimum flood peak discharges were estimated to between 22,560 and 25,960 m^3/s by using slope-area methods. The magnitude of these floods is 4.5–5.0 times the largest gauged flood (5,030 m^3/s) that has ever occurred. The episode at about 3,000 B.P. in Chinese history is known as a most disastrous time because of the droughts, harvest failure, and great famine in connection with a monsoonal climatic shift, and also the marked social change, i.e. the replacement of the Shang dynasty by the Zhou dynasty. The frequently occurring natural hazards, including floods and droughts, may have destroyed the agricultural economy of the Shang dynasty and resulted in the collapse of the Shang dynasty in the Yellow River basin. These results are very important in understanding the social effect of hydroclimatic change in the semiarid to subhumid regions.

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Paleochannels as Reservoirs for Watershed Management

R.P. Gupta, S. Kumar, R.K. Samadder

Abstract

Conservation of water needs suitable storage and management strategies. Surface storage through large dams has been a conventional method of water conservation, but it has its own well-known problems of land submergence, displacement of population, loss of agricultural land, etc. By integrating satellite sensor data, dedicated drilling borehole data, and field soil characteristics, etc., we have located three major paleochannels of the Ganges river in the Gangetic Plains and have studied their aquifer geometry and hydrogeologic characteristics. The three mapped paleochannels of the Ganges river cover an area of about 396 km^2 and would have a cumulative water storage capacity of approximately 4.65 billion cubic meters (idealized). Hydrogeologically, the aquifer is well interconnected with the adjacent alluvial aquifers. Monitoring of groundwater levels for two successive years (2006 and 2007) indicates that aquifers in alluvial plains are also recharged through paleochannels. Thus, it is inferred that such paleochannels can play a very important role in storage of water, artificial recharge of adjacent aquifers, and overall watershed management.

Keywords: Paleochannels, Gangetic Plains, hydrogeological characteristics, watershed management

Introduction

The Ganges is a major transboundary river of the Indian subcontinent, covering parts of four countries—India, Nepal, Tibet, and Bangladesh. It has a highly variable seasonal flow as its hydrologic cycle is governed by the South-West Monsoon. The average ratio of dry season to monsoon discharge is considered to be 1:6. Though the average annual rainfall in the basin is >1,100 mm, it is largely confined to the monsoonal period (15 June–15 September), so severe water scarcity conditions often occur during the summer months (April–June). The water scarcity crisis is increasing in view of increased water demand for industrialization, intensive agriculture, and rising living standards.

A key issue lies, therefore, in water resource management through storage of a major part of the monsoonal flow, which now drains almost unabated into the ocean, and its beneficial use later during the lean period.

Surface storage through large dams has been a conventional method of water conservation, but it has its own well-known problems of land submergence, displacement of population, loss of agricultural land, etc. For example, the Tehri dam, one of the largest dams built on the Ganges river in the Himalayas, has a storage volume of 3.54 billion cubic meters (BCM) but has an areal submergence of 42 km^2 (at FRL) and population displacement of about 100,000 from nearly 100 villages.

Because of the steep slopes in the mountainous region and the topographically flat nature of the Gangetic Plains, surface sites for water storage are scarce and expensive and are laced with various issues and problems. On the other hand, there exist great possibilities for underground storage, which should be relatively inexpensive. In a classic paper (published in *Science*), Revelle and Lakshminarayana (1975) articulated the conjunctive use of ground and surface water in the Ganges basin. They suggested ways to increase infiltration along the piedmont zones and parts of the upper Ganges river system into the water table during the monsoon season, followed by abstracting groundwater from the aquifers during the dry period. This process could be operated cyclically like a machine.

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We would like to take the argument further in this paper. By integrating satellite sensor data, dedicated drilling borehole data, and field soil characteristics, etc., we have located three major paleochannels of the Ganges river in the Gangetic Plains and have studied their hydrogeologic characteristics. We believe that these paleochannels, and similar ones elsewhere, can play a very important role in water storage as well as in artificial recharge of adjacent aquifers.

Study Area and Methodology Overview

The area of study falls between longitudes 77°30′E to 78°10′E and latitudes 29°10′N to 29°50′N and lies in the districts of Saharanpur and Muzaffarnagar of Uttar Pradesh, India (Figure 1). The Indo-Gangetic Plains are almost devoid of any significant relief features. Geologically, the Pleistocene to Recent alluvial deposits cover the area. Morphologically, four major landforms—piedmont zone, plains associated with river, interfluves, and paleochannels—have been recognised in the area (Kumar et al. 1996).

The data used in this study can be broadly categorized into three main groups: (a) remote sensing data, (b) ancillary data, and (c) field data. Remote sensing data from the Indian Remote Sensing (IRS) satellite mission (www.nrsc.gov.in) has been used. The IRS-LISS-II sensor image data in multispectral (green, 0.52–0.59 µm; red, 0.62–0.69 µm; near-infrared, 0.76–0.89 µm) bands have been geometrically registered, rectified for atmospheric haze effects, and edge-sharpened (Laplacian isotropic filter) for interactive interpretation.

Toposheets from Survey of India have been used to generate the base map. Soil map and hydrogeological data (such as specific yield, storage coefficient, etc.) of the aquifer have also been extracted from various existing maps, reports, and literature. In addition, extensive dedicated field work was carried out during July 2005−July 2007. Mention must be made of dedicated drilling operations carried out at selected 17 locations to varying depths of up to 60 m to collect subsurface lithologic data. Sites for drilling have been selected systematically within the paleochannels and on either side on the adjacent alluvial plains. Existing well log data from 70 vertical boreholes in the study area also have been collected, tabulated, and digitized. Software used in the study includes ERDAS Imagine-8.7, ILWIS-3.3, ARCVIEW-3.2, and ROCKWORKS-2006. The entire study has been carried out in the GIS environment.

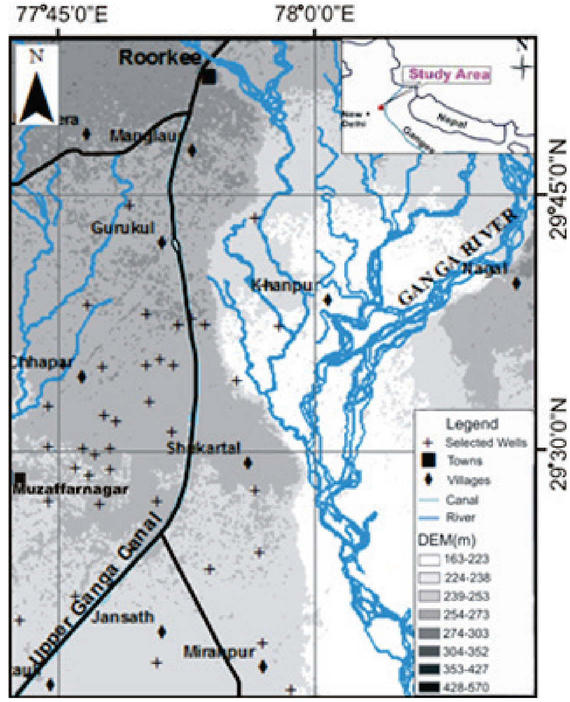

Figure 1. Location map of the study area over the digital elevation model.

Paleochannel Mapping

Owing to its synoptic view and map-like format, the satellite remote sensing imagery is a viable source for gathering quality regional data on landforms and land use/landcover (LULC) (Gupta 2003, Lillesand et al. 2007). In this area, five LULC classes have been identified: agricultural plains, paleochannel, water body, built-up area, and marshy land (Figure 2). The landcover types are closely related to the landform units (for details, see Samadder et al. 2011).

Integrating information from color infrared (CIR) composites, LULC map, field observations, and litholog data, paleochannels have been traced and mapped. Three major paleochannels exhibiting a broadly successive shifting and meandering pattern have been deciphered (Figure 2). All the paleochannels are quite wide (4–6 km), suggesting their formation by a large river. Litholog data indicates that the river paleochannel deposits are significantly different from the vast alluvial deposits in soil characteristics (particle size distribution, see below). The paleochannels are composed of coarse sand with pebbles, boulder, cobbles, etc., and appear genetically related to the well-developed regionally extensive historic river system. Low surface moisture and rather sparse vegetation on the paleochannels indicate highly permeable, porous, coarse grained materials with high infiltration rates.

Paleochannel Aquifer Geometry

Study of aquifer geometry is extremely important as it provides insight on aquifer areal extent, thickness, volume, aquifer boundaries, and interconnectivity between adjacent aquifers. This information has implications in water storage, lateral groundwater movement, and artificial groundwater recharge. Constructions of a subsurface lithological cross-section and aquifer geometry have been made by aggregating and synthesizing all the information, such as the base map, the CIR composite image, the paleochannel map, well location, and well log and elevation data. During the construction of a lithological cross-section and interpretation of aquifer geometry, it was observed that the aquifers also exhibit vertical displacements along normal faults at places (Samadder et al. 2007).

Hydrogeological Characteristics

Soil texture

Twenty surface soil samples collected from different sites located on both the paleochannels and alluvial plains were analyzed to determine soil texture. It is found that that the paleochannels comprise dominantly the sandy loam type of soil; on the other hand, the alluvial plains are characterised by a relatively finer silty loam type of soil.

Hydraulic conductivity

Hydraulic conductivity was estimated to ascertain the relative hydraulic properties of the paleochannels and the adjacent alluvial plains. A total of 82 soil samples

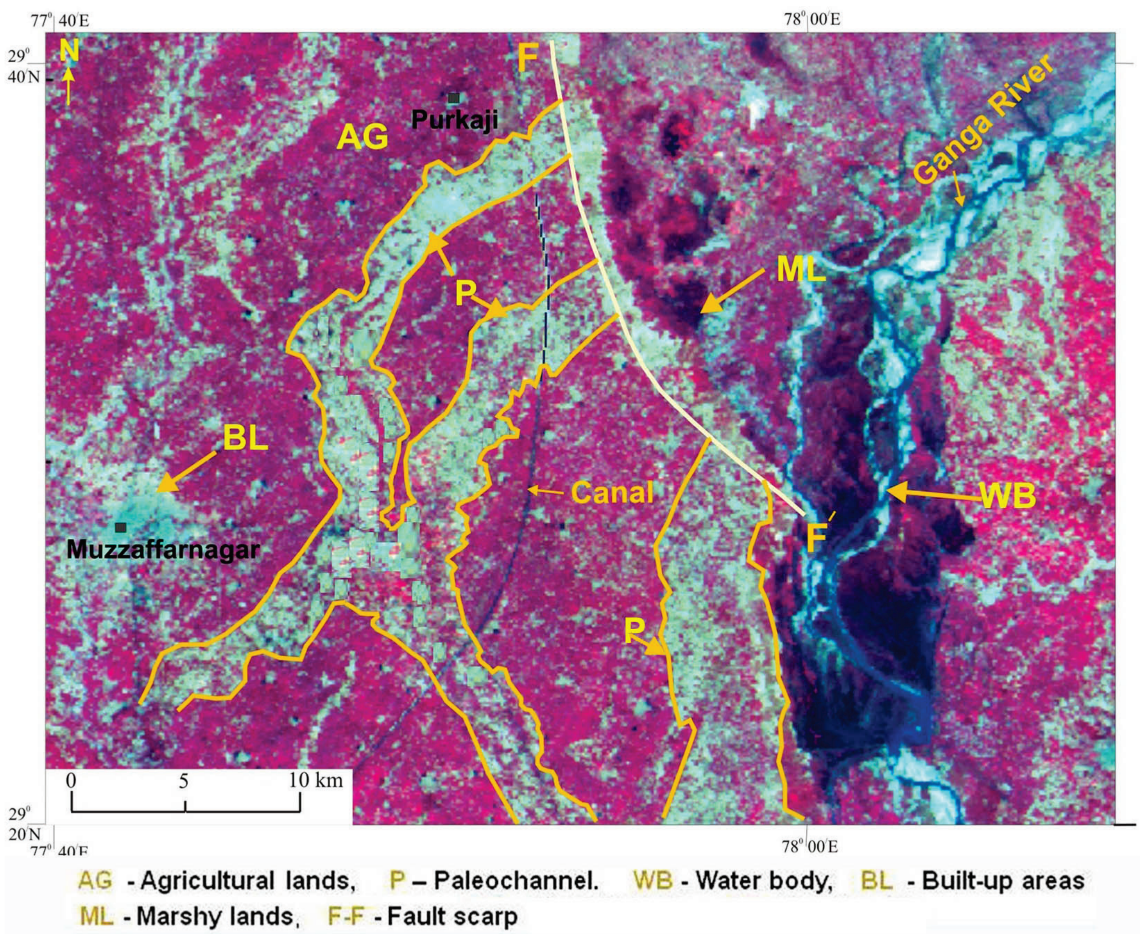

Figure 2. Color infra-red (CIR) composite generated from IRS-LISS-II satellite sensor data. Near-infrared band coded in red color; red band coded in green color, green band coded in blue color. Note the prominent paleochannels marked *P*.

were collected from different depths of wells. Bulk hydraulic conductivity was estimated for each sample form the grading curve method of Hazen approximation using D_{10} values (Hazen, 1911). The estimated bulk hydraulic conductivity for samples at different depths is found to range from 30 to 75.3 m/day for the paleochannel samples and from 13.5 to 22.3 m/day for the alluvial plains. These values are broadly corroborated by pumping test data where the values are found to range from 10 to 48 m/day in and around the area (Pandey et al. 1963).

Rate of natural groundwater recharge

Estimation of the natural rate of groundwater recharge (i.e., infiltration) is very important for assessing storage and artificial groundwater recharge. This can be done by measuring soil moisture movement in the unsaturated zone. The technique of estimation of recharge rate by using artificial tritium as a tracer (Zimmerman et al. 1967, Athavale and Rangarajan 2000) has been applied in this study area. The major source of recharge to groundwater in the study area is precipitation, about 85 percent of which occurs during the monsoon period (15 June–15 September). Field experiments have been performed at fourteen sites—eight within the paleochannel and six on the adjacent alluvial plains—for natural recharge estimation. Tritium was injected at 70-cm depth (so that it is below the normal root zone) immediately before the monsoon period (June 2006). The soil samples were collected just before injection (June 2006) and after the monsoon (October 2006) and analyzed, as per the standard procedure (Samaddar et al. 2011). The results indicate a higher rate of recharge in the paleochannels (17.0–28.7 percent) as compared to the alluvial plains (6.3–11.0 percent).

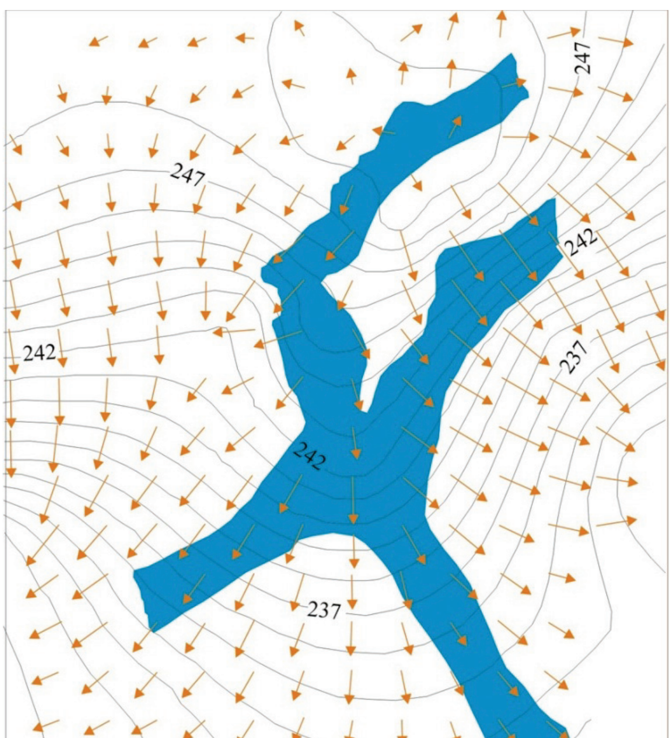

Figure 3. Typical groundwater level contours and flow direction map. Note that the paleochannels (blue) are recharging the aquifers.

Groundwater flow

A groundwater table map has been generated to establish the groundwater flow direction within the area. For this purpose, the depth to groundwater levels have been monitored in 37 observation wells (12 in the paleochannel aquifers and 25 in the adjacent alluvium plains) over a period of 2 years (2006 and 2007) both before and after the monsoon period. The computed groundwater elevation data was used to generate groundwater table contour maps and flow vector maps (Figure 3). The figure indicates that groundwater flows away from the paleochannel both before and after the monsoon period. The typical contour pattern (convex downwards) in the paleochannel aquifer is interpreted to be due to high porosity and permeability and its higher vertical hydraulic conductivity, further suggesting that groundwater recharge through paleochannels could lead to a gradual recharge of the adjacent alluvial plains.

Potential of Paleochannels for Acting as Reservoirs

The above results show that the paleochannels of the Ganges in this area possess an average width of almost 4–6 km and are quite extensive, running for a strike length of about 60–80 km. The paleochannel aquifer is unconfined and is mainly composed of coarse sandy material along with boulder and pebble beds and extends to a depth of about 65–70 m. The three mapped paleochannels of the Ganges river cover an area of about 396 km^2 on the surface and are computed to have a cumulative water storage capacity of approximately 4.65 BCM (idealized). The value of hydraulic conductivity ranges from 30 to 75 m/day in the paleochannel and from 13.5 to 22.3 m/day in the adjacent alluvial aquifers. The natural groundwater recharge rate due to precipitation is found to be 18.9–28.7 percent in the paleochannel area and 6.3–8.9 percent in the in the alluvial plains.

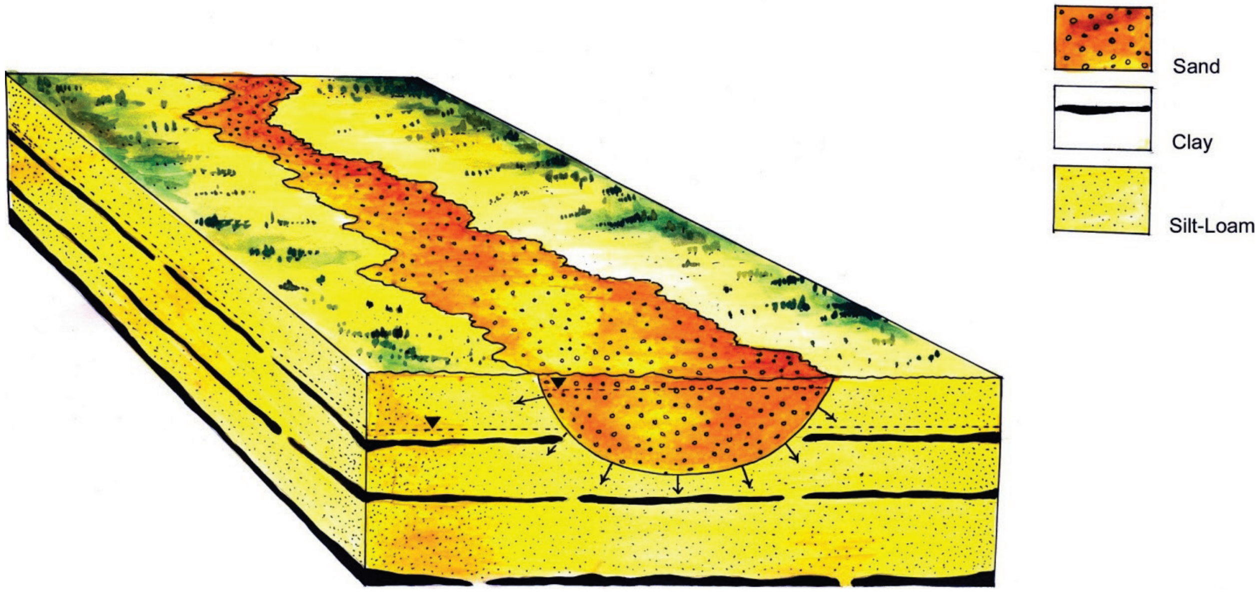

Figure 4. Schematic showing a paleochannel as a subsurface reservoir and source of continuous gradual recharge of groundwater.

It is observed that the paleochannel aquifer is well interconnected with the adjacent alluvial aquifers. Monitoring of groundwater levels for two successive years (2006 and 2007) indicates that the paleochannels also recharge the aquifers in the alluvial plains. Therefore, if paleochannel aquifers were used for storage of water and artificial recharge, water from the paleochannels would keep continuously flowing into the adjacent alluvial aquifers, making the paleochannels an almost infinite source of slow and gradual recharge (Figure 4).

In the area adjacent to the paleochannels, an average decline in the water table of about 0.24 m/year has been observed over a 10-year period. To arrest this decline, it is estimated that an annual recharge of about 30 MCM water is required per 1,000 km^2 of alluvial aquifer area (assuming specific field = 0.15), which can be easily accomplished by recharging through the paleochannel aquifers.

Thus, it is inferred that paleochannels of such hydrogeological characteristics and setting can play a very important role in water storage as well as in artificial recharge of adjacent aquifers. Apart from the Ganges and Indus Plains in India, the methodology can be applied for identification of similar suitable paleochannels in the alluvial tracts of the world, such as the Hwang Ho Plains in Northern China, the Po-Lombardy Plains of Italy, the Nile Plains of Egypt, and other areas.

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The Effectiveness of Aerial Hydromulch as an Erosion Control Treatment in Burned Chaparral Watersheds, Southern California

Peter M. Wohlgemuth, Jan L. Beyers, Peter R. Robichaud

Abstract

High severity wildfire can make watersheds susceptible to accelerated erosion, which impedes resource recovery and threatens life, property, and infrastructure in downstream human communities. Land managers often use mitigation measures on the burned hillside slopes to reduce postfire sediment fluxes. Hydromulch, a slurry of paper or wood fiber that dries to a permeable crust, is a relatively new erosion control treatment. Delivered by helicopter, aerial hydromulch has not been rigorously field tested in wildland settings. Concerns have been raised over its ability to reduce watershed erosion along with its potential for negative effects on postfire ecosystem recovery. Since 2007 we have compared sediment fluxes and vegetation regrowth on plots treated with aerial hydromulch versus untreated controls for three wildfires in southern California. The study plots were all on steep slopes with coarse-textured soils that had been previously covered with mixed chaparral. Sediment production was measured with barrier fences that trapped the eroded sediment. Surface cover was repeatedly measured on meter-square quadrats. The aerial hydromulch treatment did reduce bare ground, and at least some of this cover persisted through the first postfire winter rainy season. Aerial hydromulch reduced hillslope erosion from small and medium rainstorms (peak 10-minute intensities of <30 mm/hr), but not during an extreme high-intensity rain event (peak 10-minute intensity of >70 mm/hr). Hydromulch had no effect on regrowing plant cover, shrub seedling density, or species richness. Hence, in chaparral watersheds, aerial hydromulch can be an effective postfire erosion control measure that is environmentally benign with respect to vegetation regrowth.

Keywords: wildfire, erosion control, hydromulch

Introduction

Wildfires can increase flooding and accelerate erosion in upland watersheds, adversely affecting natural resources and downstream human communities. Burned watersheds coupled with heavy winter rains can produce floods and debris flows that may affect riparian refugia of endangered species. They may also threaten life, property, and infrastructure (roads, bridges, utility lines, communication sites, pipelines) some distance from the fire perimeter. Land managers often use mitigation measures on the burned hillside slopes to reduce postfire sediment fluxes as the first step in ecosystem restoration and to protect human developments. Some of these rehabilitation treatments are costly but have not yet been proven to reduce erosion in wildland settings and may have serious consequences for postfire watershed recovery.

The physical landscape in southern California reflects the balance between active tectonic uplift and the erosional stripping of rock and soil material off the upland areas, along with the delivery of this sediment to the lowlands. Fire is a major disturbance event in southern California chaparral shrublands, and much of the erosion occurs immediately after burning. The postfire landscape, with the removal of the protective vegetation cover, is susceptible both to dry season erosion—ravel—and to wet season erosion—raindrop splash, sheetflow, and rilling (Rice 1974). Moreover, fire alters the physical and chemical properties of the soil—bulk density and water repellency—reducing infiltration and promoting surface runoff (DeBano 1981). The enhanced postfire runoff can remove additional soil material from the denuded hillsides and can mobilize sediment deposits in the stream channels

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to produce debris flows with tremendous erosive power (Wells 1987). Postfire erosion rates decline as the regrowing vegetation canopy and root system stabilizes the hillslopes, providing critical watershed protection (Barro and Conard 1991). In the southern California foothills, erosion is driven by winter cyclonic storms. Summer thunderstorms are rare, and snowmelt runoff is virtually nonexistent, except at the highest elevations.

Hydromulch, a slurry of paper or wood fiber that dries to a permeable crust and is used to increase groundcover, is one treatment option for reducing postfire erosion. Hydromulch has been used extensively on road cut slopes and construction sites. For burned areas with steep slopes with no road access, the hydromulch needs to be applied by fixed-wing aircraft or helicopter. The aerial hydromulch used in southern California is a wood and (or) paper mulch matrix with a non-water-soluble binder, often referred to as a bonded fiber matrix (BFM) (Hubbert 2007). The BFMs are a continuous layer of elongated fiber strands held together by a water-resistant, cross linked, hydrocolloid tackifier (bonding agent) that anchors the fiber mulch matrix to the soil surface (Hubbert 2007). BFMs provide a thicker cover than ordinary hydromulch and are recommended for steeper ground and areas frequented by high intensity storms. BFMs largely eliminate direct rain drop impact onto the soil, have high water holding capacity, are sufficiently porous not to inhibit plant growth, and will biodegrade completely. Breakdown of the product does not occur for up to 6–12 months through multiple wetting and drying cycles (Hubbert 2007).

Aerial hydromulch is a relatively new erosion control treatment that has not been extensively tested under field conditions in burned upland areas. Its ability to reduce erosion has not been quantitatively demonstrated, while its effects on regrowing vegetation are virtually unknown (Robichaud et al. 2000). The objectives of this study are to quantify the ability of aerial hydromulch to reduce postfire hillslope erosion and to document its effect on vegetation regrowth.

Study Sites

Aerial hydromulch has been used on three large wildfires located on National Forest lands in southern California in close proximity to the wildland/urban interface since 2007 (Figure 1). In each case a U.S. Department of Agriculture Forest Service Burned Area Emergency Response (BAER) team determined that there were significant threats to life, property, and infrastructure in the downstream human communities and recommended aerial hydromulch as a postfire erosion control treatment.

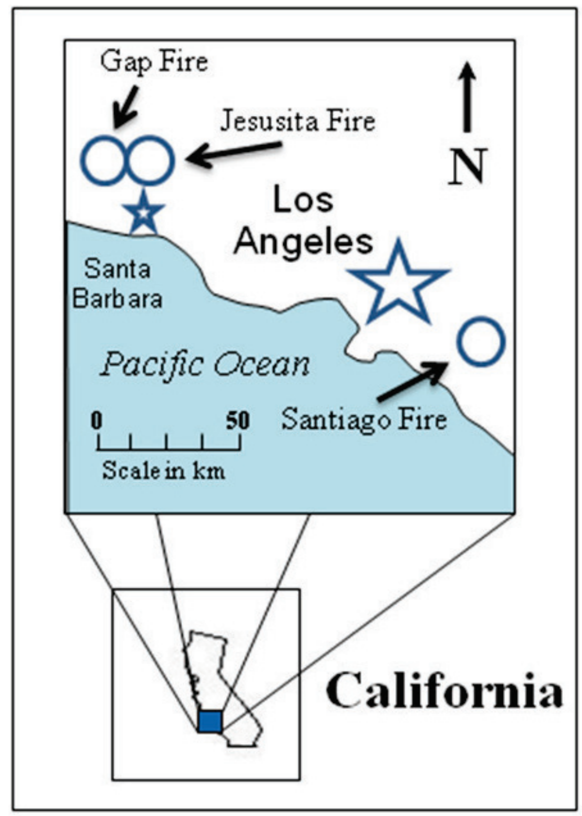

Figure 1. Location map of the study areas.

The Santiago Fire burned over 11,300 ha in Santiago Canyon, northeastern Orange County, CA (Figure 1). The burned area is a deeply dissected mountain block underlain by sedimentary and metamorphic rocks that produces a rocky, highly erosive soil (Wachtell 1975). The area was covered with thick chaparral vegetation, with some areas having no previous record of burning. Approximately 500 ha were treated with aerial hydromulch at a total cost of just under $5 million.

The Gap Fire burned nearly 3,850 ha in the Santa Ynez Mountains in Santa Barbara County, CA, in July 2008 (Figure 1). The burned area consists of the upper half of the coastal face of a linear mountain range underlain by sedimentary rocks that produce an erosive coarse-grained soil (Shipman 1981). Prior to the fire, the area was covered with mixed chaparral. Nearly 625 ha were treated with aerial hydromulch at a total cost of just under $5 million.

The Jesusita Fire burned roughly 3,540 ha in the Santa Ynez Mountains, Santa Barbara County, CA, in May 2009 (Figure 1). The burned area included the middle to lower slopes and canyons of the coastal face of a linear mountain range underlain by sedimentary rocks that produce an erosive coarse-grained soil (Shipman 1981). Nearly identical in site characteristics (topography, geology, soils) to the Gap Fire, the area was also covered with mixed chaparral vegetation prior to the fire. Over 80 ha were treated with aerial hydromulch at a total cost of $640,000.

Methods

Sites were operationally treated with aerial hydromulch, and plots were established on uniform slope facets. Companion control plots were established in nearby untreated areas, ideally within a few hundred meters of the treated plots with similar site characteristics. A minimum of four raingages were deployed at each study site to measure precipitation amounts and intensities. County gages within 5 km of the sites provided long-term data to put the measured rainfall in historical context. The studies were funded to measure erosion and vegetation regrowth for 3 years after each fire.

Hillslope erosion was measured using barrier fences constructed of high tensile strength nylon landscape fabric wired to t-posts (Robichaud and Brown 2002). The sediment fences were arranged in discrete clusters on the Santiago and Gap sites and along a series of adjacent interior spur ridges at the Jesusita site. On the Gap and Jesusita sites, the sediment fence plots were bounded by an upper trench, creating an area 15–20 m in length. The plots on the Santiago site were unbounded, averaging about 55 m in length. The fences built for these studies were approximately 5 m wide and 1 m high, with the capacity of the fence determined by its height, width, and hillslope gradient. Sediment captured by the fences was cleaned out using shovels and buckets after each rainstorm or series of storms. The sediment from each fence cleanout was weighed in the field and subsampled for moisture, and the field weights were corrected to account for the weight of the water.

Cover was measured in 1 m × 1 m quadrats using a grid frame and a pointer. The pointer was lowered at 100 points in a 10 cm × 10 cm grid within each quadrat. Hits were recorded for the various classes of cover and were converted to a percentage. Groundcover categories consisted of hydromulch, organic material (stumps, wood, live plant bases, litter), and bare soil including gravel and rocks. If the aerial hydromulch covered pieces of rock or wood, it was counted as mulch. Aerial cover from the regrowing vegetation was tallied separately from groundcover provided by plant bases. Two quadrats were initially sampled after site establishment just upslope of each sediment fence. An additional five quadrats were established for each fence in the first postfire spring season. These latter quadrats were placed from 4 to 20 m from the fence along vertical transects at the edges of each fence contributing area. Aerial plant cover was recorded by species. Surveys were conducted 2–3 times during the first postfire year, then annually in the spring. Indicators of postfire vegetation response include the amount of aerial plant cover (as opposed to groundcover), the density of shrub seedlings (the eventual climax vegetation), and a measure of species diversity or richness.

Results and Discussion

Initially, aerial hydromulch greatly reduced bare ground on the treated plots compared to the controls (Tables 1–3), presumably affording a greater level of watershed protection. Treatment cover of 17–24 percent persisted through the first postfire winter (over 65 percent on the Santiago site), but the hydromulch was less than 5 percent by the end of the second or third year after the fire. Cover of organic matter, less than 10 percent at the time of site establishment, reflected differences in rainfall as well as inherent site characteristics. Organic cover accumulated slowly on the Santiago site (which experienced a postfire drought) compared to the spectacular regrowth on the Jesusita site.

Total annual rainfall (and percent of long-term normal from nearby county gages), peak 10-minute rainfall intensity, and hillslope erosion aggregated to annual totals for treated and untreated plots are arrayed in Tables 4–6. The 64–84 percent reduction in first-year sediment yield compared to the untreated controls suggests that the aerial hydromulch was effective in controlling erosion on the Gap and Jesusita sites. At the Santiago site, initial storms of small and moderate amounts and intensities showed a reduction in hillslope erosion on the treated plots similar to the two Santa Barbara sites. However, an unusual short-duration but very high-intensity thunderstorm (>70 mm/hr) at the end of May completely overwhelmed the site, causing massive erosion to treated and untreated plots alike. Sediment fences were overtopped and the differences in first-year erosion (Table 4) are minimum values and merely reflect the differences in fence capacity. Thus, while aerial hydromulch can reduce hillslope erosion from small and medium rainstorms, it was ineffective during an extreme high-intensity rainfall event.

The hydromulch treatment caused no substantial differences in the amount of aerial plant cover, shrub seedling density, or species richness (Tables 7–9). There was no evidence that any species were eliminated or suppressed by the presence of the hydromulch. Thus, apart from minor differences attributed to inherent site characteristics, the aerial hydromulch was environmentally benign with respect to vegetation regrowth.

Conclusions

Resources on public lands need wise management while human development requires prudent hazard protection. Both are threatened by accelerated erosion in the aftermath of wildland fire. Aerial hydromulch is a relatively new erosion control technique that was previously untested in southern California watersheds and had raised concerns about unwanted environmental side effects. A recent series of wildfires and the application of aerial hydromulch as a BAER treatment prompted this study to evaluate the effectiveness of the mulch in reducing erosion and its affect on regrowing chaparral vegetation.

The aerial hydromulch does increase groundcover, some of which persists through the first postfire rainy season. The aerial hydromulch reduced hillslope erosion by 64–84 percent compared to untreated controls for small and moderate rainstorms, but not for an extreme rainfall event (>70 mm/hr). Moreover, the aerial hydromulch appeared to have no affect on postfire vegetation cover or species richness. Thus, in southern California chaparral watersheds, aerial hydromulch appears to be an effective postfire erosion control treatment from all but the largest storms and is environmentally benign with respect to vegetation regrowth.

Table 1. Average groundcover—Santiago Fire.

Survey	**Hydromulch (n=10) (%)**	**Control (n=10) (%)**
	Site establishment	
Treatment	66.0	0
Organics	0.8	2.1
Bare soil	33.2	97.9
	Year 1	
Treatment	65.7	0
Organics	3.1	2.8
Bare soil	31.2	97.2
	Year 2	
Treatment	18.4	0
Organics	27.8	20.5
Bare soil	53.8	79.5
	Year 3	
Treatment	3.6	0
Organics	64.0	41.7
Bare soil	32.4	58.3

Table 2. Average groundcover—Gap Fire.

Survey	**Hydromulch (n=10) (%)**	**Control (n=6) (%)**
	Site establishment	
Treatment	87.8	1.7
Organics	1.2	2.4
Bare soil	11.0	95.9
	Year 1	
Treatment	24.9	0
Organics	21.0	17.0
Bare soil	54.1	83.0
	Year 2	
Treatment	0.6	0
Organics	72.6	83.7
Bare soil	26.8	16.3

Table 3. Average groundcover—Jesusita Fire.

Survey	Hydromulch (n=10) (%)	Control (n=9) (%)
Site establishment		
Treatment	80.6	0
Organics	10.2	11.0
Bare soil	9.2	89.0
Year 1		
Treatment	17.7	0
Organics	66.7	62.6
Bare soil	15.6	37.4

Table 4. Average hillslope erosion—Santiago Fire.

Collection period	Hydromulch (n=10)	Control (n=10)
Year 1		
TAR	275 (59%)	
I_{10}	70.1	
HE	20.67*	26.1*
Year 2		
TAR	336 (64%)	
I_{10}	38.6	
HE	6.4	8.6
Year 3		
TAR	547 (93%)	
I_{10}	58.8	
HE	10.3	10.8

TAR, total annual rainfall (mm) (percent of normal)
I_{10}, peak 10-minute intensity (mm/hr)
HE, hillslope erosion (Mg/ha)
*Minimum value (fences overtopped)

Table 5. Average hillslope erosion—Gap Fire.

Collection period	Hydromulch (n=10)	Control (n=6)
Year 1		
TAR	469 (54%)	
I_{10}	59.4	
HE	7.8	21.5
Year 2		
TAR	1,055 (113%)	
I_{10}	27.4	
HE	2.8	5.1

TAR, total annual rainfall (mm) (percent of normal)
I_{10}, peak 10-minute intensity (mm/hr)
HE, hillslope erosion (Mg/ha)

Table 6. Average hillslope erosion—Jesusita Fire.

Collection period	Hydromulch (n=10)	Control (n=9)
Year 1		
TAR	554 (87%)	
I_{10}	41.1	
HE	5.3	33.7

TAR, total annual rainfall (mm) (percent of normal)
I_{10}, peak 10-minute intensity (mm/hr)
HE, hillslope erosion (Mg/ha)

Table 7. Average vegetation response—Santiago Fire.

Survey	Hydromulch (n=10)	Control (n=10)
Year 1		
APC	13.1	20.9
SSD	NA	NA
SR	1.7	3.2
Year 2		
APC	95.0	99.5
SSD	2.6	2.7
SR	4.1	5.5
Year 3		
APC	141.5*	115.4*
SSD	NA	NA
SR	8.4	10.5

APC, aerial plant cover (percent)
SSD, shrub seedling density (seedlings per quadrat)
SR, species richness (species per quadrat)
*overlapping plant cover can exceed 100 percent

Table 8. Average vegetation response—Gap Fire.

Survey	Hydromulch (n=10)	Control (n=6)
	Year 1	
APC	75.3	47.0
SSD	6.9	6.4
SR	6.2	3.2
	Year 2	
APC	160.4*	143.9
SSD	4.8	5.3
SR	9.8	6.8

APC, aerial plant cover (percent)
SSD, shrub seedling density (seedlings per quadrat)
SR, species richness (species per quadrat)
*overlapping plant cover can exceed 100 percent

Table 9. Average vegetation response—Jesusita Fire.

Survey	Hydromulch (n=10)	Control (n=9)
	Year 1	
APC	155.3	158.7*
SSD	7.5	3.2
SR	7.4	8.4

APC, aerial plant cover (percent)
SSD, shrub seedling density (seedlings per quadrat)
SR, species richness (species per quadrat)
*overlapping plant cover can exceed 100 percent

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Remote Sensing of Soil Tillage Intensity in a CEAP Watershed in Central Iowa

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Abstract

Crop residues on the soil surface decrease soil erosion, increase water infiltration, increase soil organic matter, and improve soil quality. Thus, management of crop residues is considered an integral part of most conservation tillage systems. Crop residue cover is used to classify soil tillage intensity and assess the extent of conservation tillage practices. Our objectives were (1) to review remote sensing methods for assessing crop residue cover and soil tillage intensity, and (2) to examine process models for assessing the effects of crop management decisions on soil and water quality.

The spectral properties of crop residues and soils in crop production fields in Maryland, Indiana, and Iowa were measured with ground-based spectroradiometers and airborne and satellite imaging spectrometers. Physically-based spectral indices that detect absorption features associated with cellulose and lignin were robust and required minimal surface reference data for mapping soil tillage intensity across agricultural landscapes. Stratified sampling protocols are needed that use a limited number of hyperspectral images to provide reliable data for training classifiers of widely available multispectral images.

The South Fork watershed is a Conservation Effects Assessment Project (CEAP) watershed in central Iowa. Farmer surveys, surface reference data, and remotely sensed data provided spatially-explicit input data for hydrologic and soil carbon models. Crop and soil management scenarios were evaluated using watershed- and field-scale models. An interconnected suite of models is required to address the wide range of agronomic, environmental, and economic questions likely to be posed by farmers, stakeholders, and policymakers related to harvesting crop residues for biofuels.

Keywords: crop residue cover, remote sensing, SWAT, EPIC, water quality, soil erosion, CEAP

Introduction

Crop residues on the soil surface can decrease soil erosion and runoff, increase soil organic matter, improve soil quality, increase water infiltration, and reduce the amounts of nutrients and pesticides that reach streams and rivers (Lal et al. 1999). Three useful categories of tillage based on crop residue cover after planting have been defined: intensive tillage has <15 percent residue cover; reduced tillage has 15–30 percent residue cover; and conservation tillage has >30 percent residue cover (Daughtry et al. 2006). Thus, quantification of crop residue cover is required to evaluate the effectiveness and extent of conservation tillage practices as well as the extent of biofuel harvesting. For agricultural fields, the standard technique for measuring crop residue cover is the line-point transect method, which is time consuming and prone to errors (Morrison et al. 1993).

Remote sensing approaches for assessing soil tillage intensity and crop residue cover have frequently used broadband multispectral sensors, such as Landsat Thematic Mapper (TM). Within these broad bands, crop residues and soils are spectrally similar and often differ only in amplitude. Crop residue can be brighter or darker than soils depending on soil type, crop type, moisture content, and residue age (Daughtry and Hunt 2008), making it difficult to distinguish residues from soils.

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With hyperspectral reflectance data, three relatively narrow absorption features—centered near 1,730, 2,100, and 2,300 nm—can be detected that are primarily associated with nitrogen, cellulose, and lignin concentrations. These features are not readily discernible in the spectra of fresh vegetation, but they are evident in reflectance spectra of dry non-photosynthetic vegetation, including crop residues. Reflectance spectra of dry soils also lack these absorption features but may have additional absorption features associated with minerals (Serbin et al. 2009 a).

Our objectives were (1) to review remote sensing methods for assessing crop residue cover and soil tillage intensity, and (2) to examine process models for assessing the effects of crop management decisions on soil and water quality. We started with the fundamental spectral properties of crop residues and soils, extended the results to fields and watersheds, and finally simulated the effects of biofuel harvesting on soil erosion and soil C.

Materials and Methods

Laboratory to Small Plot Scales

Reflectance spectra were acquired with a FieldSpec Pro spectroradiometer (Analytical Spectral Devices, Boulder, CO), equipped with an 8°-fore optic, over the 350–2,500-nm wavelength region. The samples were illuminated by six 100-W quartz-halogen lamps mounted on the arms of a camera copy stand and stabilized by a current-regulated power supply. A Spectralon reference panel (Labsphere, Inc, North Sutton, NH,) was also measured and reflectance factors were calculated (Walter-Shea and Biehl 1990).

Crop residues of corn (*Zea mays* L.), soybean (*Glycine max* Merr.), and wheat (*Triticum aestivum* L.) were collected from fields near Beltsville, MD, at 1 week and 7 months after harvest. Spectral reflectance of the intact dry crop residues were measured in 45 × 45 cm sample trays filled to a depth of 2 cm.

To provide a wide range of soil colors, the topsoils for this study included Loring (fine-silty, mixed, active, thermic Oxyaquic Fragiudalfs) from Como, MS; Sverdrup (sandy mixed, frigid, Typic Hapludolls) from Morris, MN; and Gaston (fine, mixed, active, thermic Humic Hapludults) from Salisbury, NC. Each soil was dried, crushed to pass a 2-mm screen, and placed in 45-cm sample trays to a depth of 2 cm.

Reflectance spectra of crop residues and soils were acquired in situ from production fields in Maryland, Indiana, and Iowa. The 18°-fore optic of the spectroradiometer and a digital camera were mounted on a pole and positioned at 2.3 m above the soil at a 0º view zenith angle. A Spectralon reference panel was also measured and reflectance factors were calculated. Crop residue cover within the field of view of the spectroradiometer was assessed visually with a dot-grid overlay (Daughtry and Hunt 2008).

The cellulose absorption index (CAI) approximated the depth of the 2,100-nm feature using three narrow spectral bands (Daughtry and Hunt 2008). The CAI = $100(0.5(R_{2.0}+R_{2.2}) - R_{2.1})$, where $R_{2.0}$, $R_{2.1}$, and $R_{2.2}$ refer to reflectance values in 10-nm bands centered at 2,030 nm, 2,100 nm, and 2,210 nm, respectively.

Field to Watershed Scales

Airborne hyperspectral images were acquired using a ProspecTIR VS System (SpecTIR LLC, Sparks, NV) over 5 test sites in Maryland, Indiana, and Iowa (Serbin et al. 2009 b). The images covered the 450–2,500-nm wavelength region at approx. 5-nm intervals with a 4-m spatial resolution. The Hyperion imaging spectrometer on the NASA Earth Observing-1 spacecraft acquired images over test sites in Iowa that included the Walnut Creek watershed (Daughtry et al. 2006) in 2004 and 2005 and the South Fork watershed (Tomer et al. 2008) in 2009 and 2011. The images covered the 400–2,500-nm wavelength region at approx. 10-nm intervals with a 30-m spatial resolution. Landsat TM multispectral images were also acquired.

For each test site and year, crop residue cover was determined in over 50 corn and soybean fields using the line-point transect method (Daughtry et al. 2006). The distributions of crops for each test site and year were extracted from the Cropland Data Layer product (Johnson and Mueller 2010). Corn and soybeans accounted for >90 percent of the cropland in each test site.

South Fork Watershed

The South Fork watershed of the Iowa River covers about 788 km^2 and is part of the U.S. Department of Agriculture Conservation Effects Assessment Project (CEAP) (Duriancik et al. 2008). The watershed is dominated by pothole depressions and subsurface tile drainage is needed to drain the hydric soils that cover 54 percent of the watershed (Tomer et al. 2008). The watershed is 84 percent cropland, and the rest is mostly pasture or forest with very limited urban areas. Corn and soybeans are grown on more than 99 percent of the cropland area. Temporal and geospatial data for the watershed include soil properties, climate, crop classification, and management practices.

Results and Discussion

Laboratory to Plot-Scales

The mean reflectance spectra of crop residues and soils have similar shapes (Figures 1, 2). Closer examination of the 2,000–2,400-nm regions reveals two broad absorption features that are primarily associated with cellulose and lignin. As the crop residues decomposed and the relative proportions of lignin, cellulose, and other structural polysaccharides changed, the intensities of these absorption features diminished (Daughtry et al. 2010).

The spectra of three diverse agricultural soils lack the absorption features associated with cellulose and lignin but have absorption features near 2,200 nm that are associated with clay minerals (Figure 2). Kaolinite (clay), with its characteristic doublet absorption feature at 2,150 nm and 2,200 nm, is the dominant mineral in the Gaston (Ultisol) soil spectrum (Serbin et al. 2009 a). The single absorption feature at 2,200 nm is associated with illite and montmorillonite clays.

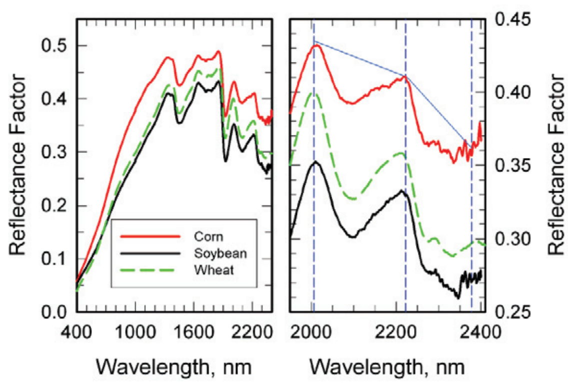

Figure 1. Reflectance spectra of corn, soybean, and wheat residues collected from fields 7 months after harvest. Absorption features near 2,100 nm and 2,300 nm are associated with cellulose and lignin. Reflectance is less than the continuum line (blue) for each feature.

Crop residue cover was linearly related to CAI from ground-based reflectance data (Figure 3A) and provided more robust assessments of crop residue cover for the diverse soils than the Landsat TM band residue/tillage indices (Serbin et al. 2009 b). Similar results have been reported for other crop residue and soil combinations.

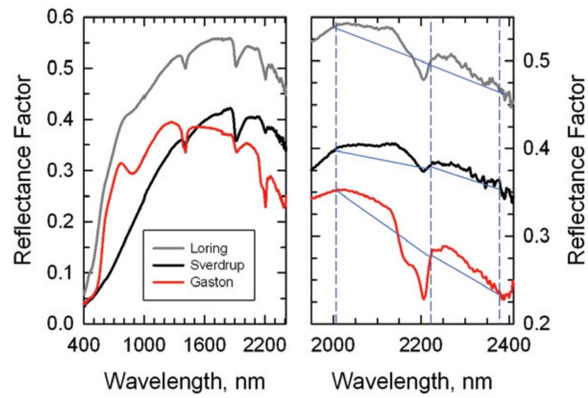

Figure 2. Reflectance spectra of three diverse agricultural soils. Absorption features near 2,200 nm are associated with clay mineralogy. Reflectance is greater than the continuum line (blue) except near 2,200 nm.

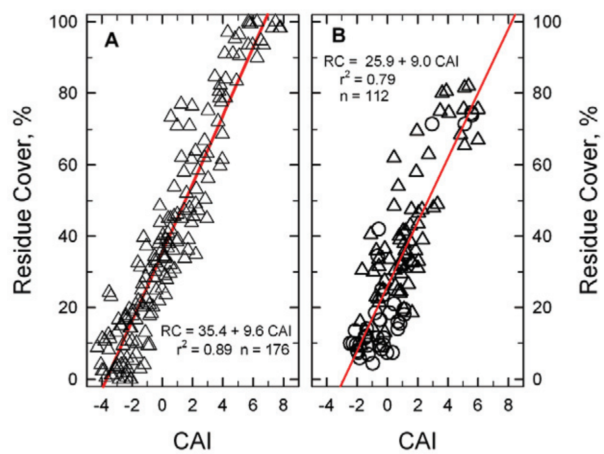

Figure 3. Crop residue cover as a linear function of CAI for (A) ground-based ASD spectroradiometer and (B) satellite-based Hyperion imaging spectrometer data.

Field to Watershed Scales

Crop residue cover was also linearly related to CAI from satellite reflectance data (Figure 3B). Corn fields typically had more residue cover than soybean fields; however, crop type did not significantly affect the slope of the regression lines. Other years and test sites had similar relationships (Table 1), which indicates that the crop residue cover versus CAI relationship is stable and robust (Daughtry et al. 2006, Serbin et al. 2009 b).

Shifts in soil tillage intensity over time and space can be tracked by combining georeferenced information on previous crops from the Cropland Data Layer with georeferenced information on crop residue cover. For example, 37 percent of the cropland in the Walnut Creek test site was classified as conservation tillage (>30 percent cover) in 2004 compared with 65 percent conservation tillage in 2005 (Table 2). This large shift in the proportion of conservation tillage may be related to spring weather conditions. Spring 2004 weather was warm and dry, which allowed ample time for additional tillage, but spring 2005 weather was cool and wet, which reduced opportunities for tillage and encouraged farmers to plant as quickly as possible.

Table 1. Sensors, locations, and statistics for crop residue cover as a linear function of CAI. Within a sensor, changes in slope are related to scene moisture.

Sensor	Loc	Date	Slope	RMSE	R^2
ProspecTIR	IN	2006	23.1	0.12	0.80
	IN	2007	27.6	0.11	0.83
	IA	2007	23.8	0.08	0.83
Hyperion	IA	2004	9.0	0.10	0.79
	IA	2005	12.3	0.15	0.65

Table 2. Percent of crop acreage in three soil tillage intensity classes. Tillage intensity is based on crop residue cover estimated using Hyperion imagery acquired shortly after planting for test sites in Iowa. Tillage classes are intensive tillage (IT) with <15 percent residue cover; reduced tillage (RT) with 15–30 percent residue cover; and conservation tillage (CT) with >30 percent residue cover.

Previous crop	3 May 2004			22 May 2005		
	IT	RT	CT	IT	RT	CT
Corn	18	36	46	7	38	55
Soybean	35	40	25	3	21	76
Total	25	38	37	5	30	65

Watershed-Scale Simulations

Currently, remote sensing cannot directly assess soil carbon (except at the soil surface), or water quality (except as turbidity and color); however, remote sensing can provide some of the biophysical variables, including soil and crop management practices and local topography, needed by physically-based process models to predict carbon dynamics and water quality across agricultural landscapes. These process-based models can provide accurate estimates of soil erosion and nutrient loading, which are both major components of carbon dynamics and water quality.

Remote sensing offers a practical method to account for the spatial and temporal variability inherent across agricultural landscapes. However, current hyperspectral sensors have a very narrow swath and do not have the capacity to map watersheds in a timely manner. Therefore, the challenge is how to best use a few hyperspectral images and many multispectral images to produce regional surveys and maps of crop residue cover and tillage intensity.

The Soil and Water Assessment Tool (SWAT) is a quasi-physically-based water quality simulation model that predicts the effect of management on water, sediment, and agricultural chemical yields in watersheds (Arnold et al. 1998). After successful model calibration, sediment discharges predicted by SWAT were well correlated with sediment discharges measured near the outlet of the South Fork watershed (Beeson et al., in press).

SWAT estimated annual sediment discharge for the South Fork watershed using three different scenarios of soil tillage intensity— (1) all fields with conventional tillage (<30 percent residue cover); (2) all fields with conservation tillage (≥30 percent cover); and (3) all fields with no tillage (except for planting and fertilizer applications; >60 percent cover). In addition, two different above ground residue removal rates (0 and 80 percent) were simulated for each tillage scenario.

Water quality as measured by sediment discharge in the South Fork river is strongly affected by the amount of precipitation received. Our simulation results revealed that sediment discharge in a wet year, such as 2008, was an order of magnitude greater than the sediment load in a normal year (Figure 4). Because of the environmental risks associated with above normal precipitation, proactive management strategies for the South Fork watershed should encourage reducing soil tillage intensity to maintain crop residue cover on the soil and establishing filter strips to retain sediment. Residue removal increased the amount of sediment discharge from the watershed because of the decreased infiltration and reduced cover on the soil surface.

Field-Scale Simulations

The Erosion Productivity Impact Calculator (EPIC) is an ecosystem model capable of simulating many processes in agricultural land such as crop growth and yield, water balance, and nutrient cycling as affected by weather, soil, and management practices (Williams et al. 1984). EPIC is well suited for addressing the effects of crop and soil management practices at the individual field unit. A series of EPIC simulations evaluated the effects of management practices on soil sustainability over time.

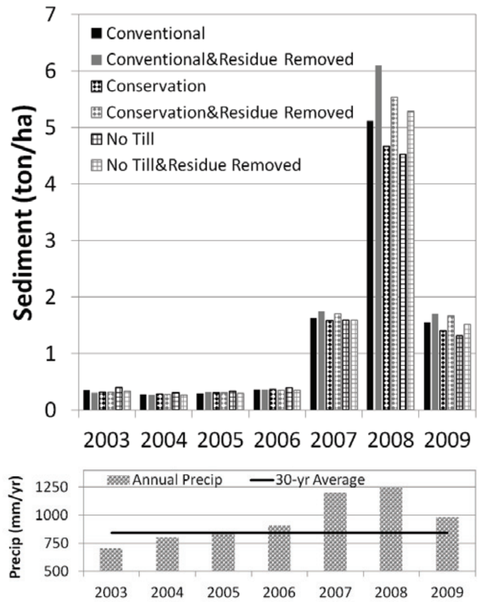

Figure 4. SWAT simulations of annual sediment discharge at the outlet of the South Fork watershed for 6 management scenarios (2003–2009). Each scenario included continuous corn with different tillage intensities and residue removal rates.

As a case study, we considered the following scenario: a farmer in the South Fork watershed has two fields with different dominant soil types. Both fields have similar topographic relief (about 2 percent slope) and have grown corn continuously with conventional tillage for several years. The dominant soil in one field is a loam and the other field is a silty clay loam (Table 3). The farmer would like to evaluate the effects of harvesting corn residues for biofuel and changing tillage practices on the stable soil carbon in the top 20 cm of each soil and the sediment loss in each field over the next 10 years. His tillage options are to continue with the conventional tillage program (<30 percent crop residue cover) or switch to a no-till program. Biofuel harvesting would remove 80 percent of the corn residue remaining on the soil surface after corn grain harvest.

After 10 years of simulations, the stable organic carbon in the top 20 cm of these soils increased and the soil loss decreased as soil tillage intensity changed from conventional tillage to no-till (Table 3). Conventional tillage with crop residue removal reduced soil organic carbon and increased soil loss from both soils, particularly for the field with silty clay loam soil. The positive effects of switching from conventional tillage to no-till more than offset any losses due to crop residue removal for both soils. These simulations indicate that removal of up to 80 percent of the corn residue from these two nearly flat soils is sustainable if tillage intensity is reduced by switching from conventional tillage to no-till management.

Table 3. EPIC simulated changes in stable carbon pools and annual sediment loss from a silty clay loam (SCL) and a loam soil in the South Fork watershed after 10 years with different management scenarios.

Soil texture	**Tillage intensity**	**Initial conditions**	**Residue removal**	
			0%	**80%**
		Stable soil carbon, t/ha		
SCL	Conventional	92.5	93.5	88.5
	No-till	92.5	96.1	93.3
Loam	Conventional	66.1	67.8	63.8
	No-till	66.1	70.4	67.3
		Annual sediment loss, t/ha		
SCL	Conventional	2.3	2.8	6.1
	No-till	2.3	0.5	1.1
Loam	Conventional	2.6	3.0	6.1
	No-till	2.6	0.6	1.1

These EPIC model simulations demonstrated that the environmental effects of crop residue removal can be minimal in some geographic areas. However, as slope increased, the amount of crop residue that can be harvested in a sustainable manner is much lower. For other geographic regions, the effects of residue removal can be different. For example, corn residue harvest adversely affected soil organic carbon of the loamy sand soils of the Coastal Plain region of the southeastern United States (Gollany et al. 2010).

Well validated models can be valuable tools for evaluating management practices for a given soil-climate-crop combination that maximize biomass production and minimize environmental effects. Spatial and temporal variability at field and landscape scales is great and a robust decision support system must include a suite of models to address the wide range of agronomic, environmental, and economic questions likely to be posed by farmers, stakeholders, and policy makers. The issues to be addressed and the level of detail (spatial and temporal resolution) required would determine which model of the suite should be used.

Conclusions

While in situ measurements can accurately describe conditions in plots and small fields, remote sensing offers a practical method to account for the spatial variability inherent across agricultural landscapes. Many of the biophysical characteristics of vegetation and soils that are needed by field-and watershed-scale process models can be derived directly or indirectly from remotely sensed data.

Currently, remote sensing systems are constrained by compromises in spatial and temporal resolution. However, data fusion and data assimilation techniques have shown considerable promise for merging data from multiple sources and creating synthetic datasets with enhanced spatial and temporal resolutions. Based on the field and watershed scenarios evaluated, process models of varying scales together with these enhanced datasets appear to provide much better descriptions of ecosystem functions at field to national scales.

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Spatiotemporal Analysis of Surface and Subsurface Soil Moisture for Remote Sensing Applications within the Upper Cedar Creek Watershed

Gary C. Heathman, Michael H. Cosh, Eunjin Han, Venkatesh Merwade, Thomas J. Jackson

Abstract

Soil moisture is a key state variable that varies considerably in space and time. From a hydrologic viewpoint, soil moisture controls runoff, infiltration, storage, and drainage. Soil moisture determines the partitioning of the incoming radiation between latent and sensible heat fluxes. Although soil moisture is highly variable, if measurements of soil moisture at the field or small watershed scale are repeatedly observed, certain locations often can be identified as being temporally stable and representative of an area average. Temporal stability analysis (TSA) is a statistical approach for describing the persistence of spatial patterns and characteristic behavior of soil moisture. Using TSA, this study was aimed at determining the adequacy of long-term point-scale surface and subsurface soil moisture (θ_v) measurements in representing field- and watershed-scale averages that will serve as in situ ground truth locations for remotely sensed soil moisture calibration and validation programs, as well as applications for hydrologic modeling. Spatiotemporal analysis revealed persistent patterns in surface soil moisture and identified sites that were temporally stable at both study scales. However, soil water patterns differed between preferred states (wet/dry) and were primarily controlled by lateral and vertical fluxes, respectively. At the field scale, locations that were optimal for estimating the area average water contents were different from permanent sensor locations. However, minimum offset values could be applied to the permanent sensor data to obtain representative field average values of surface θ_v. A TSA of 20-cm θ_v showed little correlation with surface θ_v TSA results in terms of comparable stable sites at either scale. The results are of relevance for interpreting and downscaling coarser resolution soil moisture data such as that retrieved from remotely sensed active and passive microwave platforms and in terms of modeling soil moisture at multiple scales.

Keywords: soil moisture, temporal stability, remote sensing, NASA, SMAP

Introduction

From a hydrologic viewpoint, soil moisture is the main source of memory that controls runoff, infiltration, storage, and drainage. It is a key state variable of the hydrologic cycle that varies in space and time. In meteorology, soil moisture determines the partitioning of the incoming radiation between latent and sensible heat flux and plays a crucial role in the land surface and atmospheric feedback system. From an agricultural aspect, soil moisture controls irrigation scheduling and yield forecasting. Implicit in the above mentioned is recognition that the land surface and atmosphere, as well as groundwater storage, are inextricably linked to the soil water content; therefore, detailed information of the soil water content and its spatiotemporal dynamics are necessary for sustained agricultural production, soil resource conservation, and efficient management of water resources in streams and reservoirs (Starks et al. 2003).

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At present, point-scale ground-based measurements of soil moisture are typically obtained using periodic gravimetric sampling, neutron attenuation, time-domain reflectometry (TDR), or frequency-domain reflectometry (FDR). Typically, the large spatial and temporal variability of soil moisture is not well represented with these methods. At larger scales, however, remote sensing (RS) techniques have demonstrated that spatial and temporal characterizations of surface soil moisture fields can be estimated to augment sparsely distributed point measurements from in situ networks (Njoku et al. 2002).

Several in situ studies have been designed to determine the footprint scale mean values to validate remotely sensed soil moisture products (Njoku et al. 2002, Cosh et al. 2004, Cosh et al. 2006). In situ observations are usually made point-wise, but the problem in using point measurements for validation of a measurement over a sizeable footprint is the representativeness of those point measurements with respect to the footprint measurement. In order to use the point measurements for the validation of the footprint measurement, a scaling methodology must be used. Thus, the need for better estimates of surface, as well as profile soil moisture, has heightened interest in a combination of techniques that evaluate the spatial and temporal characteristics of surface soil moisture. One approach that has been successfully used is temporal stability analysis (Vachaud et al. 1985, Cosh et al. 2004). Although surface soil moisture is highly variable, if measurements of soil moisture at the field or small watershed scale are repeatedly observed, certain locations often can be identified as being temporally stable and representative of the an area average (Vachaud et al. 1985). However, using this approach for validation has not been without errors due to limited sample size and the mismatch between point-scale field studies and the sensor footprint which may limit practical application of remotely sensed soil moisture products. Additional statistical tools may be used to characterize the sampling points to establish reliable scaling methodology. For example, a geophysical model can be used to characterize the area and to relate the point measurements to the regions whose relevance and representativeness can be evaluated through the model (Crow et al. 2005).

Our current study is, in part, one of a series of spatiotemporal studies to more effectively characterize an in situ soil moisture network in preparation for prelaunch and postlaunch National Aeronautics and Space Administration (NASA) Soil Moisture Active/Passive (SMAP) Cal/Val field activities. The objective of this study was to determine the adequacy of long-term point-scale soil moisture measurements at depths of 5 and 20 cm to represent field- and watershed-scale averages within an agricultural watershed and thus to better characterize and quantify interactions between surface and subsurface soil moisture spatiotemporal dynamics at multiple scales. This, in turn, should improve our efforts to link point-scale measurements of soil moisture to field- and watershed-scale soil moisture dynamics, on the order of 1 to 7,000 ha in size, and should provide greater confidence in calibrating and validating remotely sensed observations because of the enhanced area of representative measurement.

Methods

Study Area and in situ Soil Monitoring Network

The experimental monitoring network shown in Figure 1 is maintained by the U.S. Department of Agriculture–Agricultural Research Service (USDA–ARS) and is located within the 19,200-ha Upper Cedar Creek watershed (UCCW) of northeastern Indiana (41°27'38.11777" N. by 84°58'30.09636" W.).

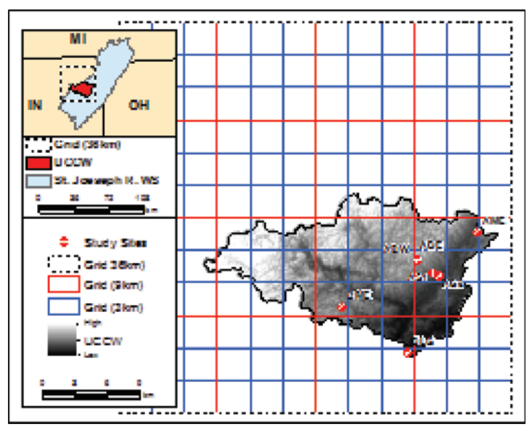

Figure 1. Upper Cedar Creek watershed study area within the St. Joseph River watershed.

The watershed monitoring network is part of the USDA's nationwide Conservation Effects Assessment Project (CEAP). The predominant land use in the UCCW is agricultural (79 percent), with major crops of corn and soybeans and minor crops of winter wheat, oats, alfalfa, and pasture. In situ soil moisture measurements were obtained from permanently installed sensors at depths of 5, 20, 45, and 60 cm at seven sites located within the UCCW monitoring network in northeastern Indiana (Figure 1). In two agricultural fields (AS1, AS2), twenty additional soil moisture sensors spaced 70 m apart were installed at depths of 5 and 20 cm in each field with automated data collection being transmitted every 30 min from June 29 through September 21, 2010. Both fields are in a corn/soybean rotation with the AS1 field being no-till (NT) and the AS2 having rotational tillage (RT) each spring in years when corn is planted. In 2010 both fields were planted to corn in April. The predominant soils at the field sites have been classified as a Glynwood silt loam (Fine, illitic, mesic Aquic Hapludalfs) in the 2.23-ha AS1 field and as a Blount silt loam (Fine, illitic, mesic, Aeric Epiaqualfs) in the 2.71-ha AS2 field. A list of soil properties with depth for each field is given in Table 1.

Table 1. Field site soil properties.

		Soil properties				
Field site	**Depth (cm)**	**%Sand**	**%Clay**	**%Silt**	**Texture name**	**Bulk density (g/cm^3)**
AS1	0–5	51.25	17.13	31.63	loam	1.42
	5–15	42.33	20.66	37.02	loam	1.42
	15–30	37.64	26.03	36.33	loam	1.49
	30–45	45.71	30.93	23.36	sandy clay loam	1.44
	45–60	36.38	36.50	27.13	clay loam	1.48
AS2	0–5	34.23	21.69	44.08	loam	1.39
	5–15	27.69	26.35	45.96	loam	1.39
	15–30	25.37	33.35	41.29	clay loam	1.53
	30–45	26.29	38.04	35.68	clay loam	1.40
	45–60	25.05	39.43	35.52	clay loam	1.38

For this study, all measurements of soil moisture are based on the frequency-domain reflectometry method of measuring the dielectric permittivity of soil to determine volumetric soil water content. As described in the manufacture's soil sensor manual (Stevens Water Monitoring Systems, Inc., Portland, OR), the Steven's Hydra Probe II (HP-II) sensor consists of a 4-cm-diameter cylindrical head that has four 0.3-cm-diameter tines that protrude 5.8 cm. A 50-MHz signal is generated in the head and transmitted via planar waveguides to the tines, which constitute a coaxial transmission line. The following parameters are derived from this basic information: temperature corrected ε' (real) and ε'' (imaginary permittivity), volumetric soil water content, soil salinity, soil conductivity, and temperature-corrected soil conductivity. All sensors were programmed using calibration coefficients based on soil texture and according to the linear relationship between volumetric water content and dielectric permittivity (Topp et al. 1985, Heathman et al. 2003). Additionally, meteorological data (i.e., rainfall, air temperature) were obtained from existing USDA–ARS weather stations in the UCCW network.

Statistical and Temporal Stability Analysis

Exploratory data analysis was first conducted with the measured surface soil moisture data (θ_v) using the mean, standard deviation, box plot, and cumulative frequency plot. In addition, normality of the dataset was tested through Lilliefors test using the *lillietest* function in MATLAB. The Lilliefors test did not reject the null hypothesis that the data are normally distributed at significance level, P = 0.05 for most of the observations. Temporal stability was evaluated using the relative difference between individual measurements of soil moisture at location *i* at time $j(\theta_{i,j})$ and the areal mean soil moisture at the same time ($\overline{\theta_j}$). The relative difference ($\delta_{i,j}$) can be expressed as

$$\delta_{i,j} = \frac{\theta_{i,j} - \overline{\theta_j}}{\overline{\theta_j}} \tag{1}$$

where $\overline{\theta_j} = \frac{1}{N}\sum_{i=1}^{N}\theta_{i,j}$ and N is the number of sensor locations. A temporal mean of the relative difference ($\overline{\delta_i}$) and its standard deviation ($\varsigma(\overline{\delta_i})$) for each location were used to determine the most temporally stable sites according to

$$\overline{\delta_i} = \frac{1}{m}\sum_{j=1}^{m}\delta_{i,j} \tag{2}$$

and the standard deviation as

$$\varsigma(\overline{\delta_i}) = \sqrt{\sum_{j=1}^{m}\frac{(\delta_{i,j} - \overline{\delta_i})^2}{m-1}} \tag{3}$$

where m is the number of measurements at a location.

When temporal mean values are plotted from smallest to largest, one can determine whether a sensor location underestimates or overestimates the field average soil moisture, which has a zero mean relative difference. A smaller standard deviation of a sensor location indicates a greater tendency for that location to be temporally stable.

The nonparametric Spearman's test was also applied to evaluate the temporal stability of different observations. The Spearman's rank correlation coefficient is expressed as

$$r_s = 1 - \frac{6\sum_{i=1}^{N}\left(R_{ij} - R_{ij'}\right)^2}{n(n^2-1)} \tag{4}$$

where R_{ij} is the rank of $\theta_{i,j}$ at location *i* and time *j* and $R_{ij'}$ the rank of the same variable at the same location but at a different time *j'*. A Spearman's rank correlation value closer to 1 indicates a stronger tendency of temporal stability between different times.

The root mean square error (RMSE) was determined to further assess offset estimates (Equation 5) at the permanent sensor sites and the most time-stable sites in each field; it was calculated as

$$RMSE = \sqrt{\frac{\sum_{i=1}^{N}(S_i - O_i)^2}{N}} \qquad (5)$$

where *N* is the number of measured and estimated soil moisture and *O* and *S* are the observed and estimated data, respectively. The closer the RMSE value is to 0, the less error in the offset estimate.

Results

In this section we present the experimental results from our agricultural field-scale temporal stability analyses and the results obtained at the larger watershed-scale permanent sensor network. Brief discussions will include the 5- and 20-cm soil moisture data analyses and the possibility of using an offset approach in adjusting the 5-cm permanent sensor data to better represent field-scale average conditions (for field AS1 only).

In Figure 2A, the ranked mean relative differences (MRD) for 5-cm temporary and permanent (P) sensor locations in field AS1 indicate that site 9 is the most temporally stable (smallest standard deviation, ±0.025) and is close to the field average moisture content (mean offset = 0.059 m^3/m^3). However, results for the edge-of-field permanent sensor location in Figure 2A show the data to be temporally unstable and to underestimate the field mean by nearly 30 percent. Figure 2B is a plot of the 20-cm sensor results that show site 1 as being temporally stable and the permanent site being unstable and more than 20 percent below the field average soil moisture conditions during the study period. It is important to note that none of the sites at each depth maintain the same ranking in Figures 2A and B. The results for field AS2 (not shown) were very similar to the results for field AS1, although permanent sensor results were closer to field average.

Cumulative frequency (CF) plots for 5- and 20-cm soil moisture data are presented in Figures 3A and B, respectively, for field AS1. The CF plots allow one to determine if a given location maintains its rank across the study period while identifying sites that represent the watershed average (CF = 0.5). General observation of the plots indicates that the permanent site maintains a low rank and significantly underestimates the field average moisture content at both depths. Also apparent in the graphs are the different ranges in moisture conditions between the two depths, with the range being greatest at the 5-cm depth from 0.15–0.40 m^3/m^3.

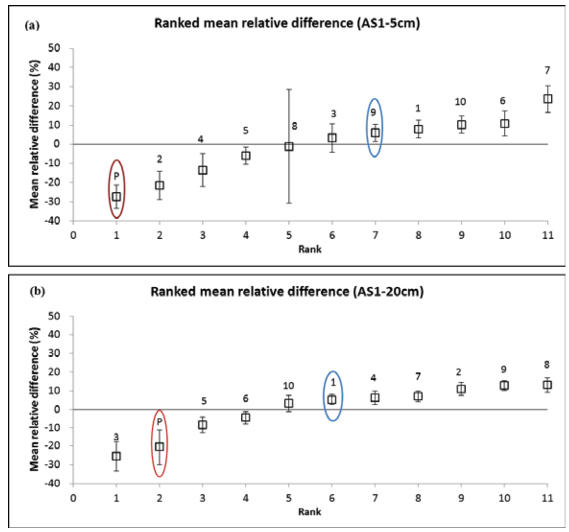

Figure 2. Field AS1 ranked mean relative differences at (A) 5-cm depth, and (B) 20-cm depth.

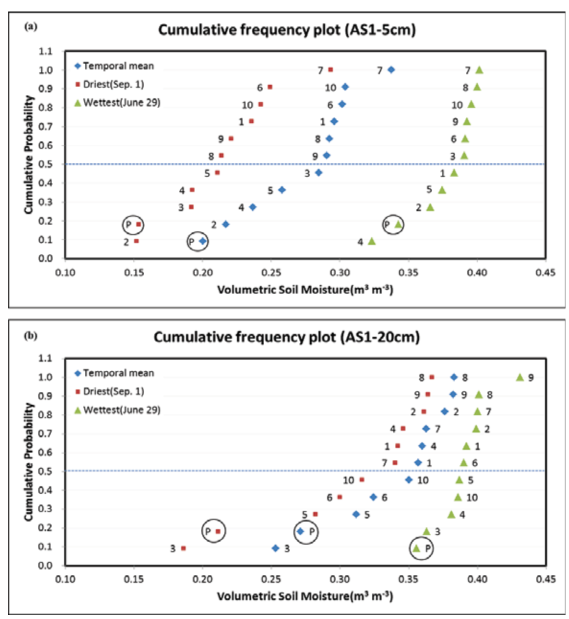

Figure 3. Field AS1 cumulative frequency plots at (A) 5-cm depth, and (B) 20-cm depth.

Grayson and Western (1998) noted that time-stable sites having a nonzero $\overline{\delta_i}$ could be used to represent the field or watershed average θ provided that the offset between the mean values and the nonzero time-stable sites was known. To demonstrate this, Figures 4A and B show linear regression of the estimates based on the work of Grayson and Western (1998). Accordingly, both sites in AS1 give acceptable offset estimates; however, temporary site 9 has the highest R^2 coefficient and the smallest offset and standard deviation.

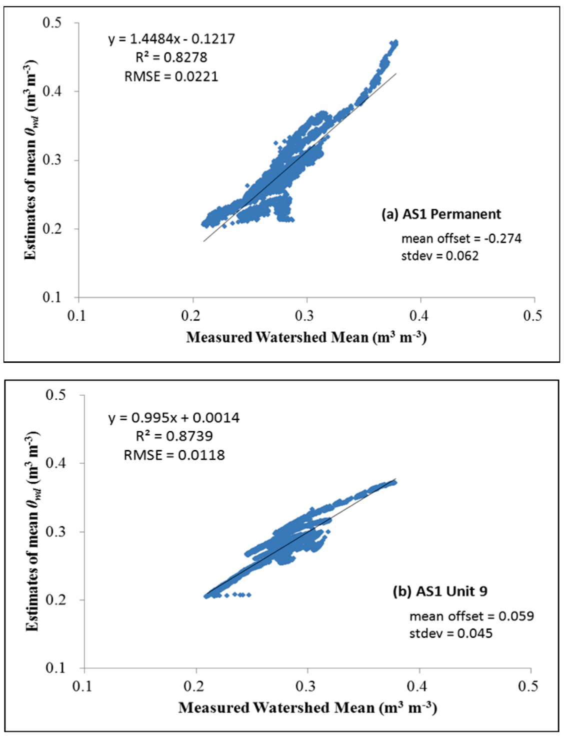

Figure 4. Offset estimates for 5-cm soil moisture in field AS1 for (A) permanent sensor and (B) site 9 locations.

The previous results provide field-scale temporal stability analyses at two sites within the UCCW environmental monitoring network. The final part of this section will show results for all seven permanent soil moisture sites during the study period, covering an area of approximately 6,200 ha, including the two field sites.

The ranked mean relative differences for UCCW soil moisture 5- and 20-cm sensors are given in Figures 5A and B, respectively. Interestingly, the AS1 field site is shown as the most temporally stable site and has an average value very close to the watershed (MRD nearest 0.0 percent). However, results differ at the 20-cm depth with site AME being the most time-stable.

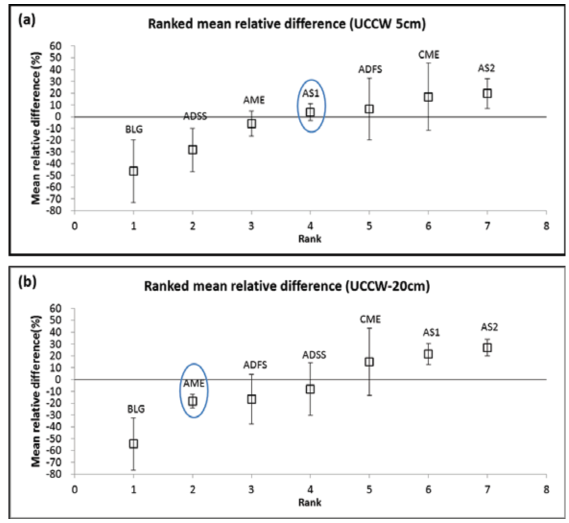

Figure 5. UCCW ranked mean relative differences at (A) 5-cm depth, and (B) 20-cm depth.

The cumulative frequency plots in Figures 6A and B indicate that the range in soil moisture is similar for both depths and that, for a particular site, rank depends on θ conditions at either depth. Sites representative of the watershed mean θ differ depending on depth and θ conditions. Thus, it is important to have θ values ranging from dry to wet conditions to avoid bias.

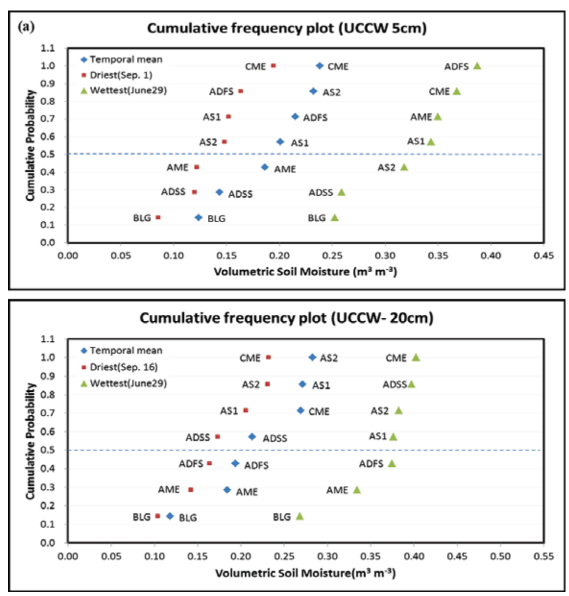

Figure 6. UCCW cumulative frequency plots at (A) 5-cm depth, and (B) 20-cm depth.

Figure 7 illustrates the linear regression for estimates of the watershed mean, again, based on the work of Grayson and Western (1998). For the time period of study, site AS1 appears acceptable as a good representative for estimating the 3- to 9-km-scale watershed mean (see Figure 1 for scale), which is important considering that the SMAP Cal/Val will consist of data at 3-, 9-, and 36-km resolutions.

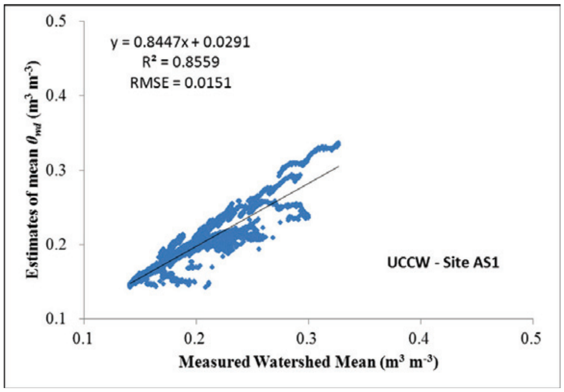

Figure 7. Offset estimates for 5-cm soil moisture for field AS1 in the UCCW soil moisture network.

Conclusions

Temporal stability analyses (TSA) of soil moisture were conducted in an effort to improve the use and representation of permanent in situ sensor data at multiple scales within the Upper Cedar Creek watershed (UCCW) in northeastern Indiana. In general, results indicate that the analyses are sensitive to the scale of observational data (2.5 to 6,200 ha). Spatiotemporal analysis revealed persistent patterns in surface soil moisture and identified sites that were temporally stable at both study scales. However, soil water patterns differed between preferred states (wet/dry) and were primarily controlled by lateral and vertical fluxes. At the field scale, locations that were optimal for estimating the area average water contents were different from permanent sensor locations. However, minimum offset values could be applied to the permanent sensor data to obtain representative field average values of surface θ_v. A TSA of 20-cm θ_v showed little correlation with surface θ_v TSA results in terms of comparable stable sites at either scale. Recommendations include additional installations of temporary sensors during remote sensing campaigns or special field projects, as well as investigating new approaches for geospatial data analysis in regards to upscaling or downscaling of soil moisture information and interpolation schemes.

Acknowledgments

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Vachaud, G., A. Passerat de Silans, P. Balabanis, and M. Vauclin. 1985. Temporal stability of spatially measured soil water probability density function. Soil Science Society of America Journal 49:822–828.

Poster Session

Building a Hydrology Science Plan for the Arctic Landscape Conservation Cooperative

B.T. Crosby, Arctic LCC Hydrology Work Group

Abstract

The Arctic Landscape Conservation Cooperative (LCC), administered by the U.S. Fish and Wildlife Service, was one of the first LCCs to receive funding. As a consequence, this LCC bears the significant responsibility of building a framework for both science and implementation that other Alaskan and national LCCs will look to for guidance. Along with a suite of other science and management concerns, the Arctic LCC must address a diverse and abundant suite of hydrologic questions. These questions are complicated in the Arctic by the strong climatic and process gradients between the coastal wetlands and the glaciated landscape of the Brooks Range. Monitoring and implementation of management decisions is also complicated by the difficulty and cost of access in a region bisected by only one road. These challenges are buffered by a suite of existing and forthcoming observational sites spanning this gradient and supported by a diverse group of academic and agency scientists. Potential for continuing hydrologic work and receiving funding in the future is high given the rapid rate and dramatic scale of changes occurring in the far north. During 2011 and 2012, the Arctic LCC Hydrology Work Group will formulate three documents that will serve the LCC community and beyond: (1) the Arctic LCC Hydrology Science Plan, (2) the Arctic LCC Hydrologic Science Implementation Plan, and (3) a Report on Common LCC Hydrologic Concerns and Coordination Opportunities. Data synthesis and distribution will be facilitated through the use of new tools such as the Hydrologic Information System developed by the Consortium of Universities for the Advancement of Hydrologic Science, Inc., and other spatial data clearinghouses.

Crosby is an Assistant Professor in the Department of Geosciences at Idaho State University, 921 S. 8th Ave, Stop 8072, Pocatello, ID 83209-8072. Email: crosby@isu.edu. The Arctic LCC Hydrology Work Group is a large, diverse group of agency and academic scientists guiding the LCC hydrology documentation.

Renewable Energy Locations for Existing and Potential Facilities within BLM Leased Land

S. Davis, C. Doughty

Abstract

In 2009, secretarial orders directed the Bureau of Land Management (BLM) to prioritize the development of renewable energy on public lands to reduce the nation's dependence on foreign oil and to reduce greenhouse gas pollution.

The Western Governors Association (WGA) collaborated with numerous stakeholders to develop the Western Renewable Energy Zones, identifying areas in 11 western states with vast renewable energy resources. The next steps support commercial transmission projects and their markets along with building interstate cooperation for transmission approvals, cost recoveries, and the evaluation of transmission strategies. A goal will be to facilitate the development of high voltage transmission to areas with high renewable energy and low or easily mitigated environmental effects.

The processing of renewable energy applications (existing and potential) for solar and wind applications and transmission lines were improved by providing complete, accurate, and easily accessible source documents, records, digital data, and map products. Environmental analysis included balancing the benefits of renewable energy with the needs to protect other resources such as crucial wildlife habitat. Data on census records, cities, roads, and wildfire-urban interface areas show areas of increased community growth.

The potential energy records are derived from the National Renewable Energy Laboratory models provided by the WGA. The probability classification system relates to electricity that could be produced (1500 MW of solar or wind and 500 MW of geothermal generating capacity), and that is needed to support the delivery of energy to major centers. The U.S. Geological Survey provided estimates for the geothermal energy sites.

The Western Interconnection is the electricity grid that includes the western states and is overseen by the Western Electricity Coordinating Council. Additional information is available at http://www.westgov.org/index.php?option=com_content&view=article&id=219&Itemid=81.

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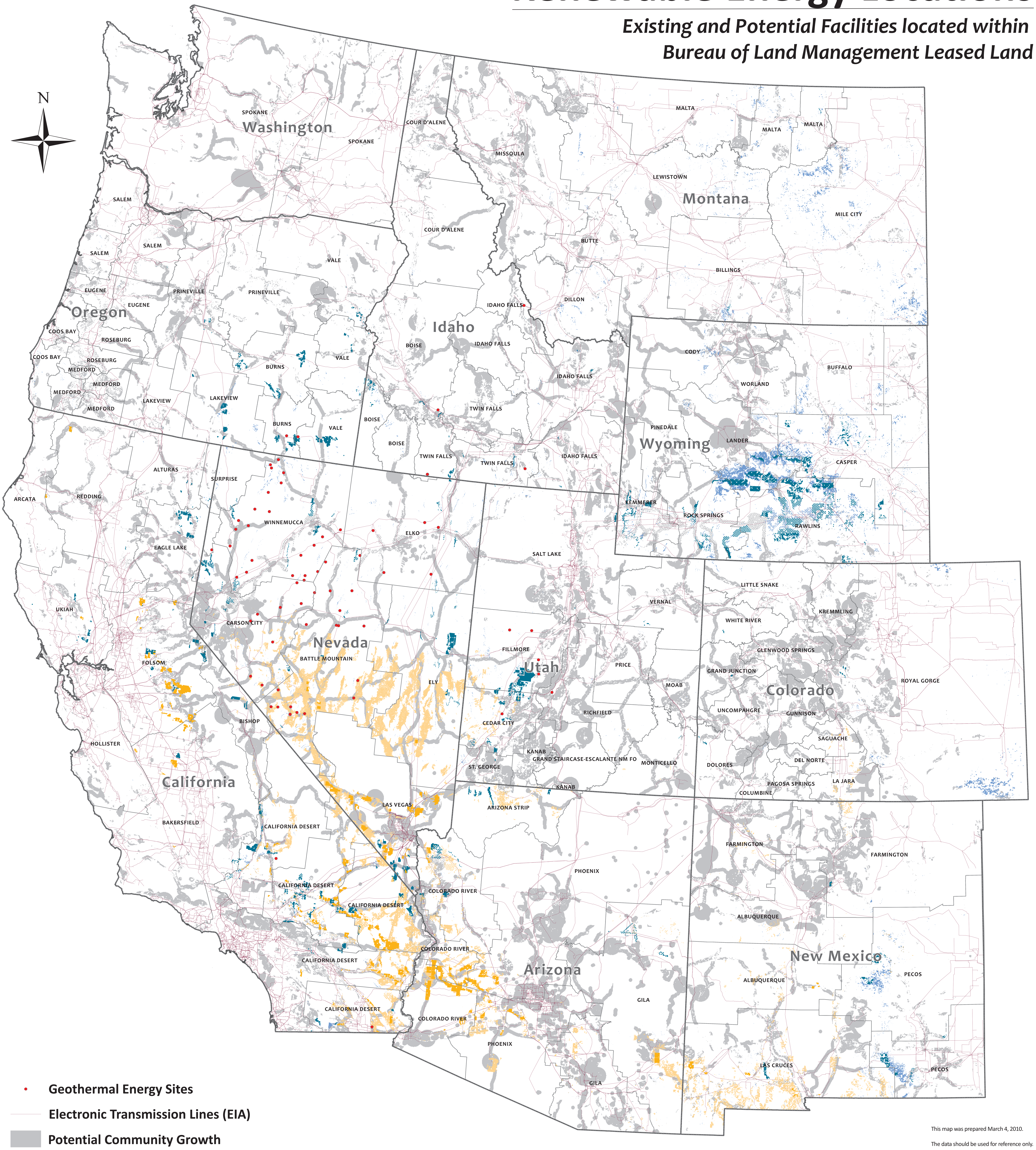
Renewable Energy Locations

Existing and Potential Facilities located within Bureau of Land Management Leased Land



This map was prepared March 4, 2010.

The data should be used for reference only.

Renewable Energy Datasets

Solar Facilities
(13,214,217.7 Total Acres)

EXISTING	POTENTIAL
2,106,496.4 Acres	11,107,721.3 Acres
219 Records (LR2000)	

Wind Facilities
(8,014,165.1 Total Acres)

EXISTING	POTENTIAL
3,736,416.5 Acres	4,277,748.6 Acres
430 Records (LR2000)	

The Existing Energy records were extracted from the LR2000 database (PDSStudio) then plotted using the CarteView application in ArcGIS. The LR2000 Existing Energy records have a disposition of Authorized or Pending. Data Source: Premier Data Services, July 2009

The Potential Energy records are derived from NREL models provided by the Western Governors Association. The displayed Potential Energy polygons are located on surface land that is managed by the BLM. These polygons have a probability classification of 4 or greater.

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An Investigation of Carbon Dynamics in Beaver Creek, Alaska, Using in situ Sensors

Mark M. Dornblaser, Robert G. Striegl, Heather Best

Abstract

Carbon dioxide (pCO_2), chromophoric dissolved organic matter (CDOM), and water-quality sensors were deployed at two remote sites on subarctic Beaver Creek, Alaska, to characterize carbon dynamics during the open water season of 2010. Beaver Creek is a tributary of the Yukon River, with nearly half of its 300-mile length classified as a National Wild and Scenic River. Beaver Creek above Victoria Creek (BCV) drains 3,315 km^2 and receives water inputs primarily from the White Mountains and other headwater catchments. Beaver Creek near Michel Lake (BCM) drains 6,164 km^2 and is located 180 km downriver from BCV in the Yukon Flats. The location of the sites permitted the study of lake and wetland inputs between the sites. Seasonal pCO_2 ranged from approx. 1,000 to 2,200 ppm at BCV and from approx. 600 to 1,200 ppm at BCM. Diel pCO_2 variations were as high as 500 ppm at BCV and 200 ppm at BCM. Both sites were supersaturated in pCO_2 with respect to atmospheric levels for the entire open water season. CO_2 fluxes from water to atmosphere ranged from 38 to 52 moles/m^2/yr at BCV and from 58 to 98 moles/m^2/yr at BCM. While rapidly changing river levels resulted in sensors being exposed to the atmosphere for varying periods of time, the use of these in situ sensors provided a means to explore C dynamics on scales that would be impossible to investigate with random discreet sampling in this remote area of Alaska.

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The Yukon River Basin Indigenous Observation Network: Preliminary Results from Baseline Datasets Highlighting Climate Variation Indicators

L.M. Mackey, C. Thomas, R. Toohey, P.F. Schuster, N. Herman-Mercer, Indigenous Observation Network Technicians

Abstract

Long-term datasets have proven to be indispensable in documenting trends in systems and identifying drivers that cause noticeable shifts from baseline conditions. These datasets are becoming increasingly important as changing climate begins to alter the livelihoods of people around the world. The livelihoods of Indigenous populations, whom rely on subsistence resources, will likely be the first to feel the more substantial effects of a changing climate. The Yukon River Inter-Tribal Watershed Council (YRITWC) Science Department was developed to address increasing concerns about observed changes and the health of the Yukon River voiced by Indigenous Peoples living in the Yukon River Basin (YRB). As a result, the YRITWC Science Department has partnered with the U.S. Geological Survey (USGS) and over 20 Alaska Native Tribes and Canadian First Nations to create the largest international Indigenous Observation Network (ION) in the world. This partnership is generating a long-term baseline dataset of water quality parameters to investigate evidence of climate change through changes in water chemistry. In 2001, the USGS initiated a five-year comprehensive water quality study throughout the YRB. In 2005, the YRITWC began collaboration with the USGS and local Tribal environmental coordinators to develop a seamless continuation water quality monitoring for the YRB that extends to the present day.

The foundation of the YRITWC is the participation and support of Tribes and First Nations within the YRB in Alaska and Canada. Interested participants from the Tribes and First Nations were trained to collect water samples and follow protocols in accordance with USGS method standards. Water quality parameters including pH, alkalinity, major ions, dissolved organic carbon, greenhouse gases, and water isotopes have been collected from 45 sites during 2001–2010. Of these sites, five have been continuously sampled since 2001. The ION expanded sampling efforts to include simultaneous biweekly collections from widely distributed sites on the Yukon River and tributaries by the indigenous technicians. From these data we have been able to detect natural trends in the river system, document major tributary influence over the course of the Yukon River, observe river seasonality from break-up to freeze-up, and calculate loads of chemical constituents.

The strong relationships and the high quality of data developed during the water quality study spurred the collaborative Active Layer Network (ALN) project in 2009. The long term ALN project monitors systematic changes in active layer thickness with correlating soil temperature and moisture analysis. Over the course of two years 20 ALN sites were installed in locations across the YRB. Each site contains a marked monitoring grid (50 m×50 m) that serves as a location for annual measurements of the active layer during maximum thaw, a set of soil moisture and soil temperature sensors placed just above permafrost and just below soil surface levels, and an air temperature sensor. The YRITWC and the USGS train the ION technicians to make the manual measurements and download the air temperature, soil temperature, and soil moisture data from each of the sites. Active layer grid measurement and accompanying data have been added to the online international Circumpolar Active Layer Monitoring (CALM) database. Though it is still early in the study, preliminary results are proving to be valuable components of the project's long-term strength and sustainability. Preliminary data for selected climate-sensitive water quality parameters and ALN datasets will be presented in a poster.

Mackey is a biologist with environmental focus in the Science Department (2008), Thomas is an environmental scientist (2009), and Toohey is the Science Department Manager (2011), all with the Yukon River Inter-Tribal Watershed Council, 323 2nd Street, Fairbanks, AK 99712 and 725 Christensen Dr., Anchorage, AK 99501. Schuster is a research hydrologist and Herman-Mercer is a social scientist, both with the U.S. Geological Survey National Research Program, 3215 Marine Street, Boulder, CO 80303. Over 100 ION technicians are trained throughout the YRB. Email: lmackey@yritwc.org, cthomas@yritwc.org, rtoohey@yritwc.org, pschuste@usgs.gov, nhmercer@usgs.gov.

Evaluating Cumulative Effects of Logging and Potential Climate Change on Dry-Season Flow in a Coast Redwood Forest

Leslie M. Reid, Jack Lewis

Abstract

Comparisons based on pretreatment calibrations between summer flows and antecedent precipitation indices (APIs) at the Caspar Creek Experimental Watersheds show increased dry-season flow for 8 yr after selective logging, followed by at least 27 yr of depressed flow. In contrast, summer flow in a partially clearcut watershed remained higher than expected for 18 yr after logging. The API-based models were used to evaluate the effects of selected climate change scenarios when combined with logging-related hydrologic changes, with the effects assumed to act independently. Changes in rainfall late in the wet season have a disproportionate effect on dry-season flows, while autumn rains have little effect.

Keywords: cumulative effect, low flow, climate change, logging, instream flow

Introduction and Study Area

Low-flow changes associated with forest management can affect downstream water supply and alter habitat quality for instream biota. Quantification of the long-term effects of various silvicultural strategies on dry-season flows would provide information needed to assure desired flow in target stream reaches. However, such an analysis requires lengthy flow records after silvicultural treatments and also generally requires that analogous records be available from a control site. Although logging has long been known to augment runoff over the short term (Moore and Wondzell 2005), studies in the Pacific Northwest and elsewhere show that flow may eventually decline to below pretreatment levels as forest regrows (Hicks et al. 1991). Earlier work at the Caspar Creek Experimental Watersheds in northwest California (Keppeler 1998, Figure 2) suggested that such a pattern may also have been emerging after selective logging of a coast redwood (*Sequoia sempervirens*) forest.

Flow has been measured since 1962 at gaging weirs in the North and South Forks of Caspar Creek (Figure 1, 39°21' N., 123°44' W.). The 424-ha South Fork watershed underwent 67 percent volume-selection logging in 1971–73 for a study of the effects of tractor yarding. In 1985–86, about 13 percent of the 473-ha North Fork watershed was clearcut, and in 1989–92 an additional 37 percent was clearcut and mostly cable-yarded during a study of cumulative watershed effects. The 48-yr flow records thus provide an opportunity to compare summer flow responses to contrasting silvicultural strategies in a pair of matched watersheds.

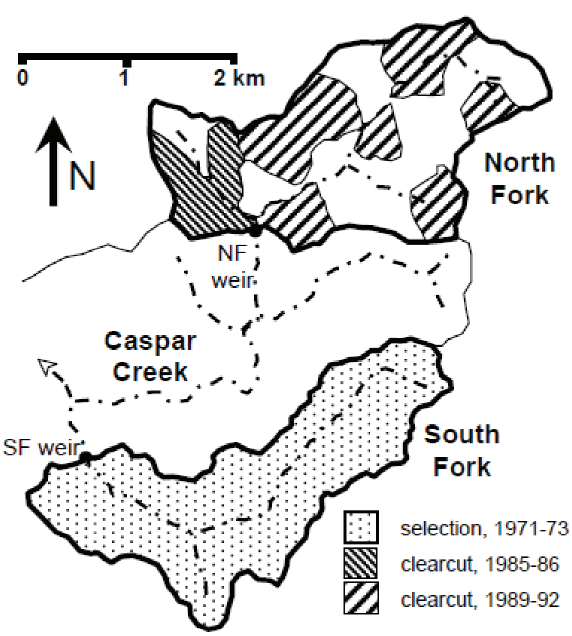

Figure 1. Caspar Creek Experimental Watersheds.

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The watersheds are underlain by marine sandstones and siltstones, and most slopes are mantled by 0.5- to 1.5-m-deep clay-loam to loam soils, often with high gravel components. Annual rainfall averages 1,170 mm, and about half runs off as streamflow; snow is negligible. About 95 percent of the rain at Caspar Creek falls in October–May, a period that also accounts for 95 percent of runoff. The minimum flow usually occurs in early October, but most streams draining <20 ha are dry by June. Old-growth forest in the area was logged in the mid to late 1800s, and by 1960 the watersheds supported 60- to 100-yr-old second-growth stands dominated by coast redwood and Douglas-fir (*Pseudotsuga menziesii*).

Anadromous coho salmon (*Oncorhynchus kisutch*) and steelhead trout (*Oncorhynchus mykiss*) often spawn in the watersheds, but portions of the channel system used for summer rearing are constrained in part by the extent of dry-season flow. Salmonid populations have declined in the area, and there is concern that potential effects from broad-scale climate change may augment the effects of silviculturally-induced flow changes. We here adapt methods used by Reid (in press) to model the potential effects of altered rainfall regimes on dry-season flows at Caspar Creek when combined with changes due to selective and clearcut logging.

Methods

The South and North Fork weirs are concrete sharp-crested compound weirs, and flow has been measured there using a sequence of float gages (recorded using strip charts) and data-logged pressure transducers. For the first experiment, calibrations established between North and South Fork flows for the pre-logging period were used to estimate expected South Fork flows after logging, and the observed deviations from expected values allowed characterization of the initial South Fork dry-season flow response to selective logging (Keppeler 1998).

This analytical strategy was no longer useful after 1985, when logging began in the North Fork. In addition, because South Fork flows had not reliably returned to pretreatment levels by 1985, the North Fork now lacked a paired control watershed, and treatment effects from the new North Fork experiment thus could not be evaluated at the weir gage. The experimental design instead employed subwatersheds as controls, and the weir gage was not directly used in the experiment. Subwatershed gaging flumes were installed in 1984, and part of the watershed was logged soon after. Subwatersheds are not gaged during summer months because many of the gage sites run dry. Consequently, dry-season flow analysis must rely on gaging records from the North and South Fork weirs.

A method was thus needed to estimate expected dry-season flows at the weirs after the 1985 logging. Rainfall has been measured nearly continuously in the South Fork and in nearby Fort Bragg for the duration of the study period. If pretreatment flows at each weir could be predicted using the rainfall record, it would be possible to calculate expected flows after logging in the absence of a flow record from a control watershed.

Preliminary analyses suggested that an antecedent precipitation index (API) might be a useful predictor. The Fort Bragg gage provides the most complete record of summer rain, while winter rains are well represented by the South Fork gage. Summer rainfalls for events >0.6 mm are strongly correlated at the two gages ($SF\ Rain = 1.03\ FB\ Rain - 0.73$ mm, $r^2 = 0.89$). Smaller events generally represent fog drizzle, which is more common at coastal Fort Bragg than at the South Fork gage, located 6 km inland. We thus combined the October–May record from the South Fork with the June–September Fort Bragg record (with daily rainfalls <0.6 mm set to 0) to construct a continuous rainfall record from 1962 through 2008. A suite of APIs with recession coefficients ranging from 0.993 to 0.600 was then calculated from the rainfall record.

Late-summer flow data are unavailable for years when weir ponds were drained to remove sediment. For years with dry-season data, we selected three to five dates in August and September that had no rainfall during the previous 3 days, had <9 mm of rain during the previous 30 days, and, when possible, were more than 6 days apart. Mean daily flows on the selected dates during the pre-logging periods were then regressed against the suite of APIs to identify the API that best predicts observed flows at each site. The resulting calibrations allow estimation of expected August and September flows at the South Fork (L_{eS}, L/km^2-s) and North Fork (L_{eN}) weirs for pretreatment (unlogged) conditions:

$$L_{eS} = 0.0143\ API_{0.985} - 0.0320 \quad (1)$$

$$L_{eN} = 0.0272\ API_{0.977} + 0.0366 \quad (2)$$

with $API_{0.985}$ (mm) calculated using a recession coefficient of 0.985 and $API_{0.977}$ using a coefficient of 0.977 (Figure 2). Flow changes after treatment were then described using ratios of observed flows to those expected for forested conditions.

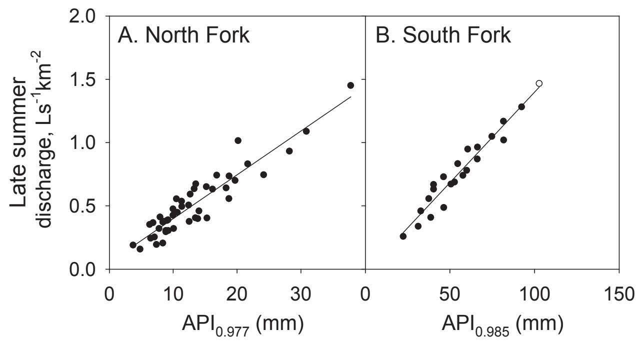

Figure 2. Calibration relations between late-summer flows and APIs in the (a) North Fork (n = 46, r^2 = 0.87) and (b) South Fork (n = 22, r^2 = 0.94).

We wanted to evaluate interactions between logging-related flow changes and those arising from potential climate change, but no significant trends in annual, spring, or autumn rainfall were evident over the 48-yr record, and climatic projections for the area are uncertain. We decided to test outcomes from six plausible rainfall regimes constructed by modifying the existing rainfall record to reflect altered annual rainfalls and changes in the seasonal rainfall distribution. We selected change scenarios within the observed range of variability so that they could reasonably be described by the API model. We did not consider indirect interactions between altered rainfall and logging effects, and we did not evaluate changes in other climatic attributes.

The 24 wettest years on record show an annual average 22 percent higher than the 48-yr average, so we constructed one 48-yr record by multiplying the recorded daily rainfalls by 1.22 and a second record by multiplying daily totals by 0.78. Rainfall in April and May accounts for an average of 10.4 percent of the annual rain over the 48-yr record and 14.9 percent during the 24 yr of that record with the highest percentages. We constructed a third record by increasing April–May daily rainfalls by a factor of 14.9/10.4 while multiplying rainfalls in other months by 85.1/89.6 to maintain the observed annual average. A fourth record was constructed by multiplying April–May rainfalls by 5.8/10.4 and rain in other months by 94.2/89.6.

Summer rainfall is minimal at Caspar Creek, with June and July accounting for only 1 percent of the annual rainfall. Years with lower than the median proportion of summer rain show a mean percentage 77 percent lower than average, so we constructed two additional records that reflect a 77 percent increase and decrease in June–July rainfall without modifying annual rainfall totals.

For each constructed 48-yr rainfall sequence, rainfall in August was set to 0 (a mean reduction of 6 mm), and values of $API_{0.977}$ and $API_{0.985}$ were calculated for September 1 of each year. Equations 1 and 2 were then used to estimate expected flow under unlogged conditions at each weir on that date for each of the six API sequences. The proportional logging-related changes in flow, having been defined as a function of time after clearcutting or selective logging, were then applied to the climatically altered flows predicted for unlogged conditions to estimate the combined effects of logging and hypothetical changes in rainfall.

Results and Discussion

Influence of Logging

Ratios between observed and expected flows show different post-logging trajectories for the partially clearcut and selectively logged watersheds (Figure 3). Late summer flows increased soon after selective logging began in the South Fork and remained high for 8 yr. By 1986, 15 yr after the 3-yr logging period began, flows had consistently dropped to below levels expected for unlogged conditions. Flows continued to drop until 1992, 21 yr after logging began. Flows have increased since then, but after 2000 they have remained slightly lower than pre-logging levels.

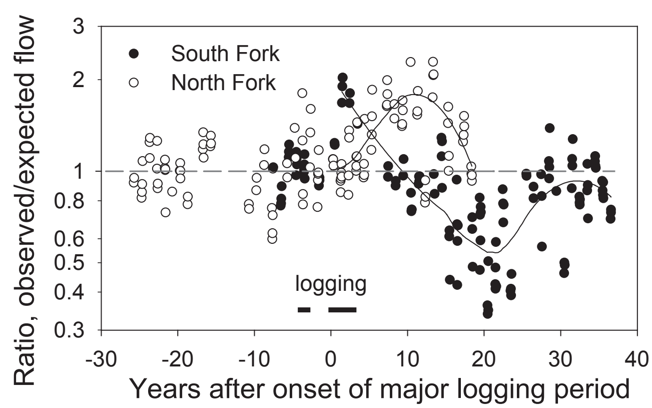

Figure 3. Late-summer flow changes after South Fork selective logging and North Fork clearcutting. Post-logging curves are fitted by loess regression. North Fork 12-yr and South Fork 30-yr values are considered outliers due to an unusual 50 mm/day June storm.

Flows took longer to respond after North Fork clearcutting, but over 11 yr the proportional increase reached a level similar to the maximum at the South Fork. The maximum mean increase at 11 yr is equivalent to a 1.57 percent increase per percent of forest logged for the 50 percent clearcut North Fork (considering both the 1985–86 and 1989–92 logging), a rate 1.2 times the maximum 1.33 percent increase per percent of forest removed by 67 percent selection logging in the South Fork. North Fork flow again dropped to pretreatment levels by 19 yr after the onset of clearcutting, but the slope of the regression (Figure 3) suggests that further decline to below pretreatment levels might soon be expected.

The large difference in response patterns for selectively logged and partially clearcut watersheds probably reflects a difference in the distribution of trees that remain after logging. Before logging, potential transpiration in this area begins to exceed actual in late spring; by June, a mature forest could utilize more water than is available. In second-growth redwood forests, many of the trees originated as stump sprouts, so clusters of trees often share a common root system. Consequently, when neighboring trees are selectively logged, the remaining trees already have the plumbing in place to take advantage of soil moisture no longer tapped by the logged trees. After an initial rise, dry-season flow thus dropped quickly to pretreatment levels as the remaining trees used up excess moisture and then continued to drop while the newly established cohort of young trees grew larger.

In contrast, on a clearcut slope, the nearest trees are off the site, and significant regrowth of foliage must occur onsite before excess soil moisture can be fully used. Dry-season flow thus remains elevated longer than in the selectively logged watershed. In addition, effects in the North Fork clearcuts are likely to have been renewed in 1995–96 by herbicide application and in 1998 and 2001 by precommercial thinning.

The contrast in initial response times may reflect differences in the distribution of logging: South Fork logging began near the weir and progressed upstream, while North Fork logging began at the headwaters. Because buffer strips left the North Fork riparian zone largely undisturbed, root networks along the full channel length could utilize the hyporheic flow associated with initial increases in dry-season runoff

Combined Effects of Logging and Climate

Of the three kinds of rainfall change modeled for the South Fork, the 22 percent change in annual rainfall produces the largest effect (Figure 4). A shift to a rainfall regime having an average annual rainfall equivalent to that of the driest 50 percent of years in the study period would lead to a 23 percent reduction in the 10^{th} percentile September 1 flow under unlogged conditions. By 21 yr after selective logging, the 10^{th} percentile flow declines to 41 percent of unlogged levels, compared to 54 percent under the present rainfall regime. For unlogged conditions under the current regime, a September 1 flow <0.55 L/km^2-s is expected an average of once in 5 yr, while lower flows would be expected nearly twice as often during the 36-yr post-logging period. Such a decrease in average flow for the post-logging period would be similar to that expected from a 10 percent decrease in mean annual rainfall under unlogged conditions.

Because recent rains affect API more strongly than earlier ones, a shift in seasonal rainfall distribution can influence dry-season flows even if annual rainfall does not change (Figure 4B). In this case, a 44 percent reduction of spring rainfall (and corresponding increases of rainfall in other months to preserve the annual average) would reduce the 10^{th} percentile flow at 21 yr after selection logging to 47 percent of forested levels.

Because of the disproportionate influence of late rainfall, a 2.5 percent increase in annual rainfall, if it occurred only by increased May rain (equivalent to an 81 percent increase in May rain), would affect September 1 flows as much as a 10 percent change distributed through the year (Figure 5). Similar calculations for other months indicate the same effect would be attained by 6.3 and 16 percent increases in annual rainfall if restricted to March and January, respectively; these would correspond to 46 and 83 percent increases in March and January rain. In effect, rainfall occurring after February has considerably more influence on dry-season flow than that occurring earlier in the wet season.

A 77 percent reduction of June–July rainfall, with no change in annual rainfall, produces a calculated flow reduction of about half of that caused by the 44 percent decrease in April–May rain. However, the actual influence of altered summer rainfall is uncertain. Some evidence (Figure 3) suggests that summer rainfall may have less effect on dry-season flows than the API-based models predict, possibly because after a seasonal soil moisture deficit accumulates, a higher proportion of rainfall may be stored in the soil and transpired before it contributes to runoff. Field experience suggests that Caspar Creek hydrographs begin to show their characteristic responses to wet-season rainfall only after about 200 mm of autumn rain has fallen.

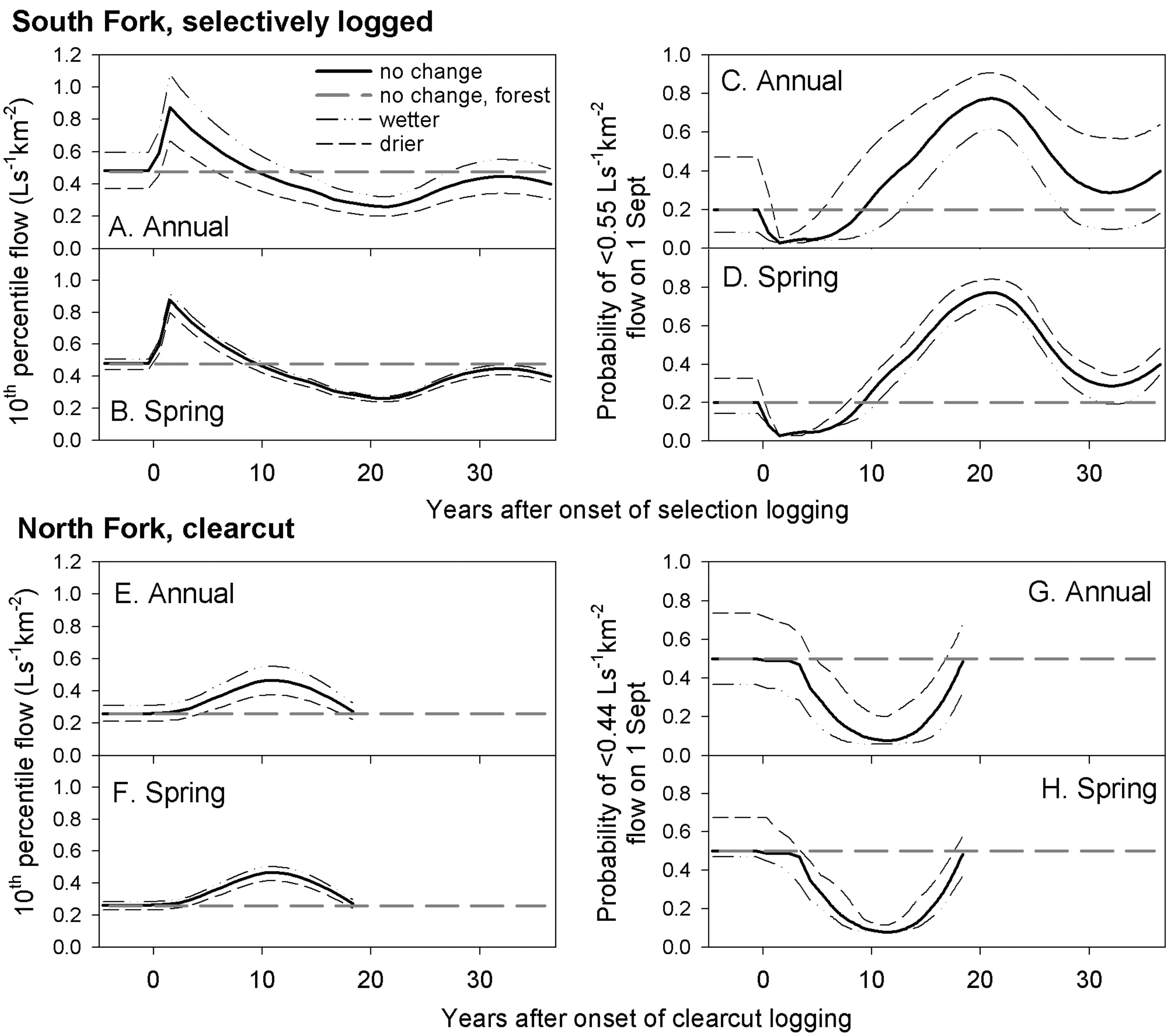

Figure 4. Modeled response of September 1 weir flows to logging and hypothetical rainfall changes. *Upper left:* Comparison of flow for which 10 percent of September 1 South Fork flows are lower for (A) a 22 percent increase and decrease in annual rainfall and (B) a 44 percent increase or decrease in April–May rainfall with no change in annual rainfall. *Upper right:* Comparison of calculated frequencies for September 1 South Fork flow <0.55 L/km^2-s (5-yr return interval) for the same changes in (C) annual and (D) spring rainfall. *Lower left:* Analogous plots for 10th percentile North Fork Caspar Creek September 1 flows after clearcut logging for the same changes in (E) annual and (F) spring rainfall. *Lower right:* Calculated frequencies for a September 1 North Fork flow <0.44 L/km^2-s (2-yr return interval) for the same changes in (G) annual and (H) spring rainfall.

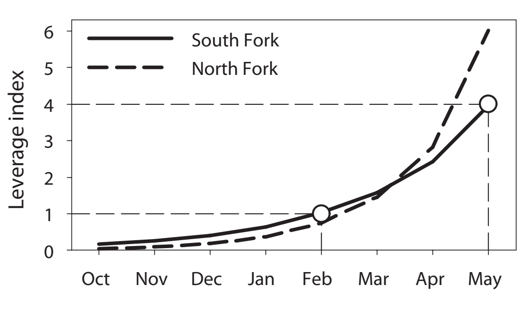

Figure 5. Influence of rainfall timing on September 1 flows. For example, a 1 percent change in annual rainfall that occurs through altered May rainfall has an effect on South Fork flow equivalent to that of a 4 percent change distributed through the year, while a 1 percent change restricted to February is equivalent to a 1 percent distributed change.

Modeling of the same hypothetical changes in rainfall regime evaluated above, but this time in combination with clearcut logging, can be carried out only for the initial 19-yr period of flow increase at the North Fork (Figure 4). Under unlogged conditions, dry-season flows are lower per unit area at the North Fork, so although the 10^{th} percentile flows increase by a maximum factor of 1.8 at both sites, the magnitudes differ (Figure 4). For the North Fork, too, seasonal redistribution of rainfall can affect the post-logging response even without a change in annual rainfall, and the relative influence of rainfall late in the wet season is even more pronounced for the North Fork than for the South Fork (Figure 5).

Conclusions

Data from Caspar Creek show that partial clearcutting in a coast redwood forest, in combination with forestry activities that often ensue, produced a larger (per unit area of clearcut equivalent) and lengthier increase in dry-season flow than selective logging. Results also show a long period of dry-season flow depression after selective logging. Data are not yet available to determine whether the partially clearcut watershed will also undergo a period of reduced flow.

A variety of climatic projections suggest that major changes in annual rainfall are unlikely in the area, though an analysis by Madej (p. 40, this volume) indicates that the seasonal distribution of rainfall may be shifting. Calculations of the combined effects of logging and potential changes in rainfall regime indicate that a change in the proportion of rainfall that occurs late in the wet season may augment or reduce dry-season flows, in concert or opposition to logging effects, even if mean annual rainfall does not change.

Although we have considered only changes in rainfall regime, other kinds of climatic change may be important in the region, such as altered frequency of summer fogs and associated changes in dry-season temperature. Should conditions become warmer in the region, the major climatic contribution to an effect on summer flows may well be indirect. Increased temperatures during the growing season increase the water demands of crops, leading to increases in water diversions for irrigation. If such extractions are superimposed on a period of flow depression following logging, changes in dry-season flow might become of particular concern.

The contrasting patterns of flow change after selective and clearcut logging may allow design of watershed-scale silvicultural strategies to lessen risks of adverse dry-season flow changes in stream reaches of concern. In any case, it may be useful to employ a variety of silvicultural strategies well-dispersed through time and space to avoid synchronizing dry-season flow responses in watersheds where altered low flows may adversely affect beneficial uses in downstream reaches.

Acknowledgments

This study is part of the Caspar Creek research program conducted since 1960 by the U.S. Forest Service Pacific Southwest Research Station and the California Department of Forestry and Fire Protection. We thank Peter Cafferata, John Griffen, and Michael Furniss for reviews.

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Making Long Term Watershed Research Accessible

K.J. Cole, D.E. James and J.D. Obrecht

United States Department of Agriculture, Agricultural Research Service, Ames, IA & GIS Support and Research Facility, Iowa State University

USDA Agricultural Research Service

Conservation Effects Assessment Project

Abstract

The USDA's Agricultural Research Service (ARS) has conducted watershed-scale research since the 1930's. Recent efforts have been made to compile the independently collected research data collections into a system that would help in data preservation and facilitate research within and across watersheds with diverse collaborators. The driving force for the database project has been the Conservation Effects Assessment Project (CEAP) conducted in cooperation with the NRCS. The system that was developed (STEWARDS) uses an enterprise relational database and geospatial web server. The most challenging design issue involved the creation of a metadata system that would fully describe the data collections used in the watershed projects. The resulting metadata describes research methods, site descriptions, and spatial data. The methods cataloging system uses a data structure that was adapted from the National Environmental Methods Index (NEMI) as a searchable compendium of environmental methods. The implementation of the database system has elucidated some challenges. Compiling historic research data is very time consuming and there can be resource and technology limitations with the data producers. Management has to be supportive and provide consistent focus. Data policies need to be specific and part of research plans.

Description and Data Holdings

- Data were obtained from ARS CEAP long-term research watersheds. Themes include: hydrology, meteorology, land use, water-quality, soil quality, agronomic practices, and economics.
- The data table contains unique records based on the pairing of the Site ID and Date/Time stamp.
- A data-definition table maps to the unique columns of the data table. The definition table identifies the unique columns for the Site ID, Date/Time stamp, individual parameters, methods, and comments associated with the parameters (Fig. 1).
- Application was created using ArcGIS Server 9.3.1 and ArcGIS API for Flex (Fig. 2).
- The STEWARDS database consists of approximately 7 million records (Fig. 3).
- Thirteen ARS CEAP watersheds have contributed records to date.
- Data are added from the watersheds periodically.

Acknowledgements: We would like to thank the ARS team of J.L. Hatfield, G.J. Wilson, J.L. Steiner, E.J. Sadler and B.C. Vandenberg, who were instrumental in the development and support of the database system.

We would also like to thank the data providers who contributed to the STEWARDS system.

To access STEWARDS: http://www.ars.usda.gov/watersheds/stewards

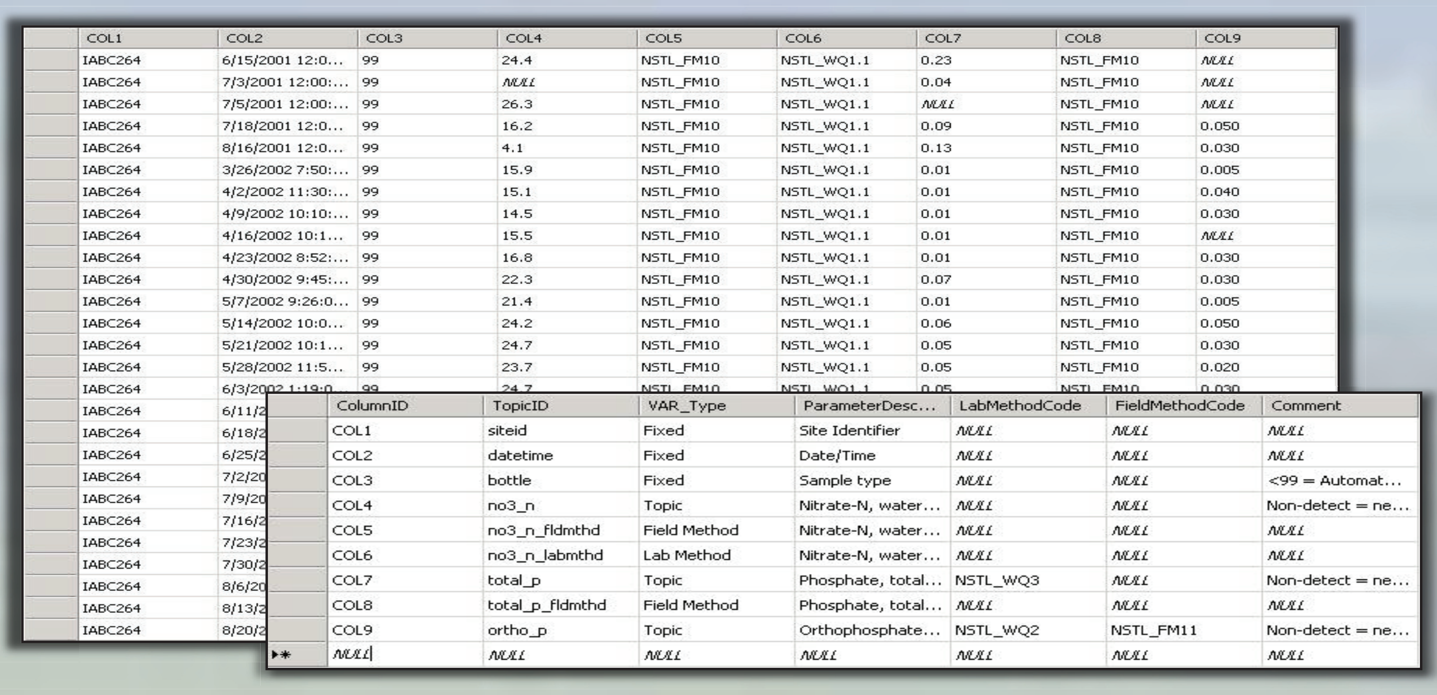

Figure 1. Data table and data-definition table link data columns to metadata.

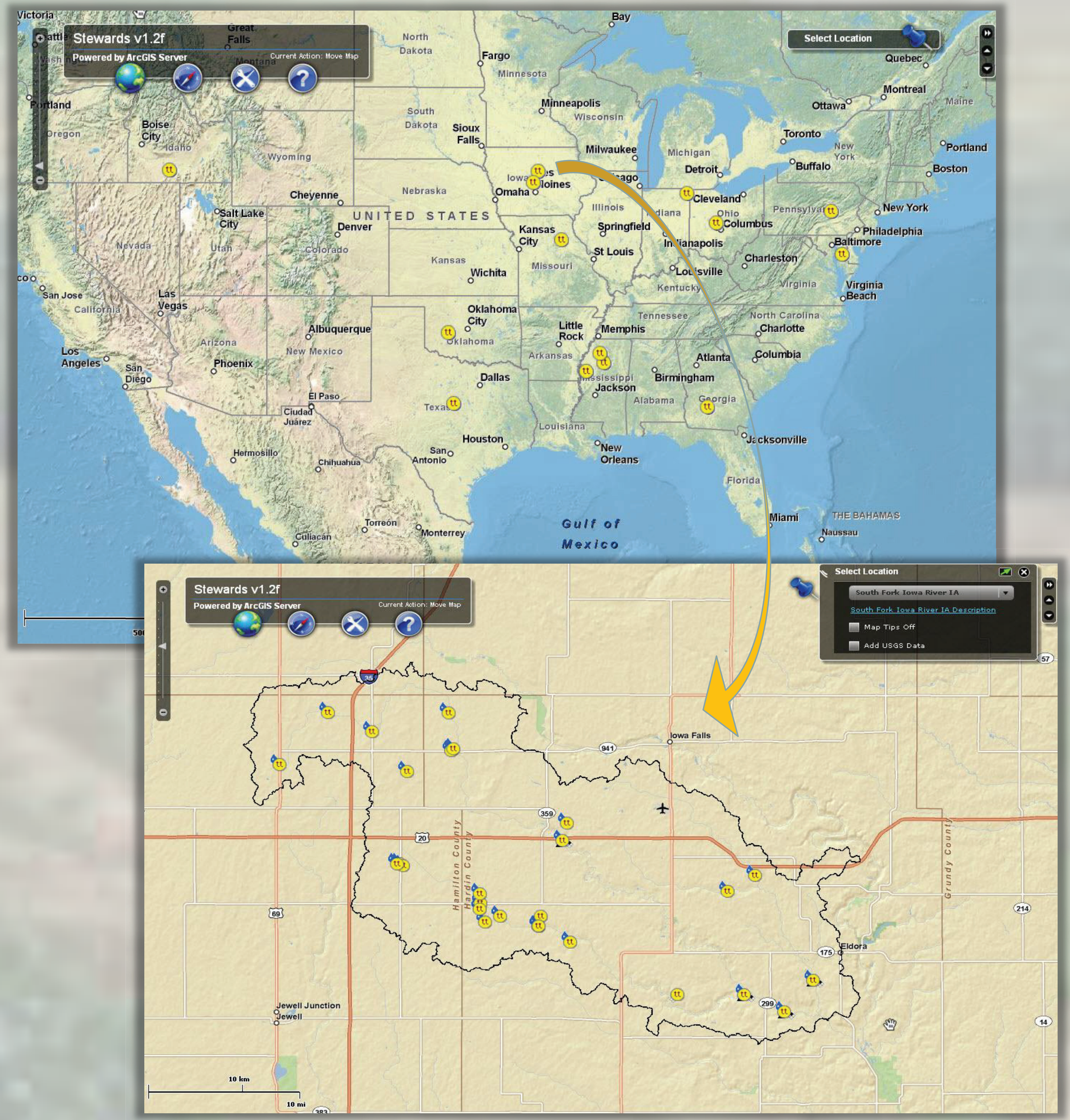

Figure 2. STEWARDS Application at opening screen and selected watershed displaying research sample locations.

Metadata

- Essential to describe disparate research methods.
- Used data schema from the National Environmental Methods Index for consistent metadata elements.
- Individual site descriptions with instrument histories are maintained.
- FGDC-compliant metadata for spatial database components are required for distribution.

Implementation Challenges

- Data provider reluctance to contribute due to limited resources and publishing concerns.
- Source data formats difficult to process.

Lessons Learned

- Strong leadership needed from local to national level.
- Data access and use policies need to be well defined and understood.
- Require modern data management products in research project plans.
- Make data contributions as easy as possible.

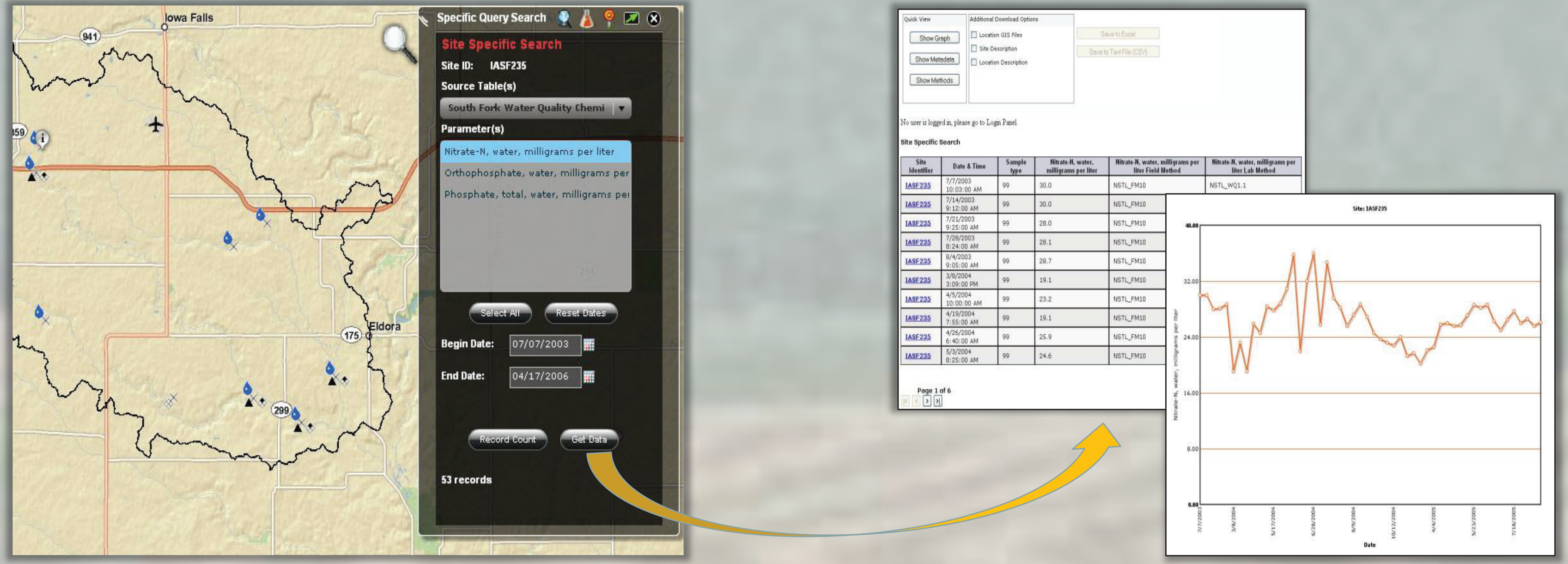

Figure 3. Example data selection and graphing.

Summary

ARS scientists collect watershed research data that vary spatially, thematically, and temporally. The STEWARDS application maintains a database that is easily accessible by the scientific community through an integrated environment using ArcGIS Server and SQL Server. This design represents the precursor to a future Agency-wide database that will provide access to ARS data holdings by the scientific community.

Using Remotely Sensed Brightness Temperatures to Infer Snowmelt Onset and Runoff in High Latitude Drainage Basins

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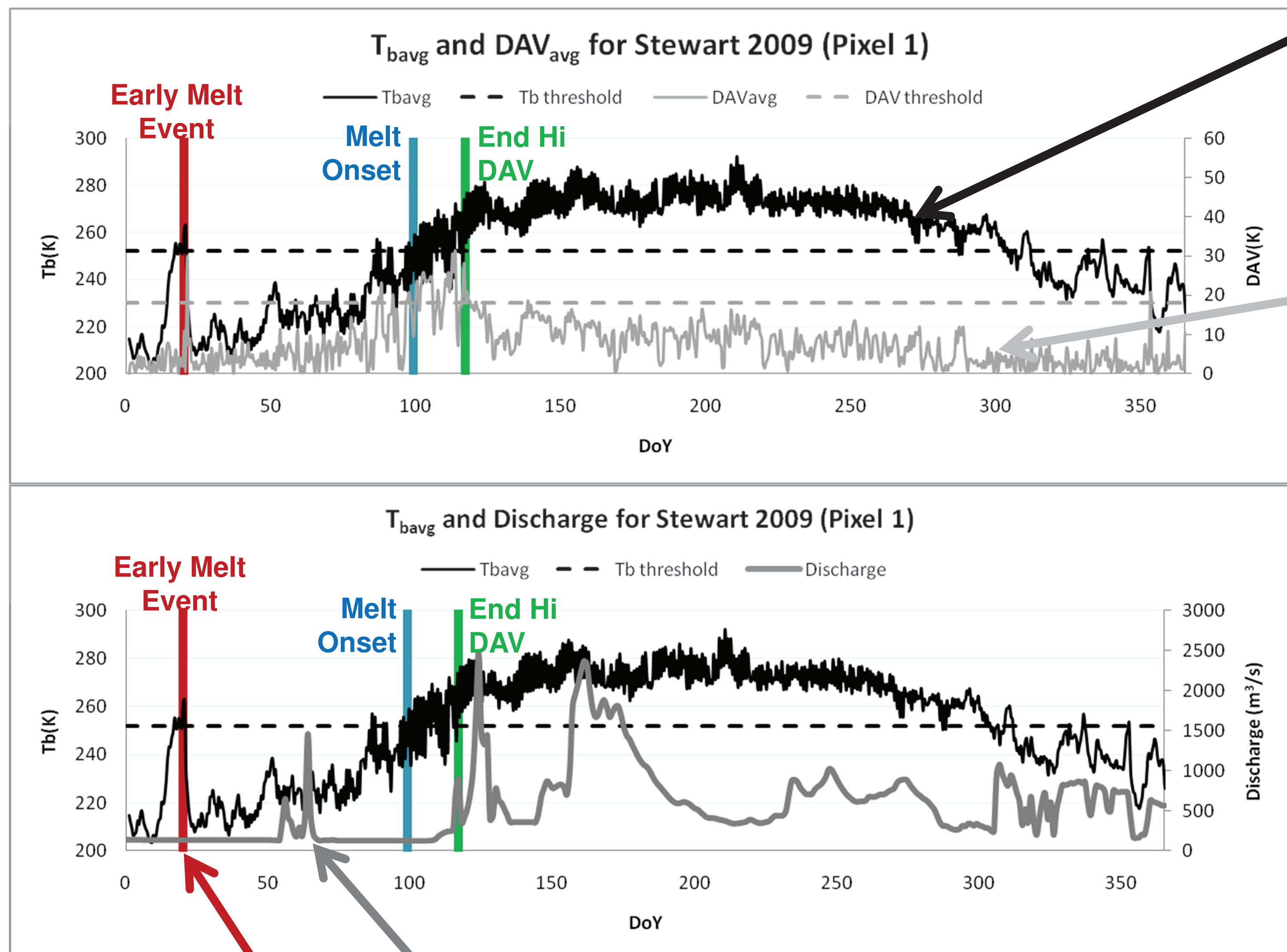
Detecting Snowmelt Onset and Duration

High latitude, snowmelt dominated basins are sensitive to increasing winter air temperatures with snowpacks' integrating these changes over the season, potentially influencing the timing of snowmelt onset and resulting in changes to spring runoff and associated flooding, often the most significant hydrologic events of the year. To understand and model these processes for ungauged basins with little meteorological data, passive microwave AMSR-E brightness temperature observations are used to determine snowmelt onset timing, melt-refreeze duration, and occurrence of early snowmelt events (short periods of melt before full spring melt onset). These data are then used as inputs into a simple flux based snowmelt runoff model (SWEHydro) to simulate peak discharge and timing.



T_b is the product of emissivity and physical temperature. Frozen snow has low emissivity compared to wet snow.

DAV represents the dynamism of the snowpack, when high the snowpack is melting during day and refreezing at night.

When Tb>252K & DAV>18K the snowpack is melting.

- When sustained for several days it is melt onset.
- All melt before melt onset are early melt events.
- When thresholds no longer met (end high DAV) snowpack is fully melting (not refreezing)

A significant early snowmelt event on January 20th may have contributed to the small peak in discharge on ~day 60.

Figure 1. Brightness temperatures (T_b) **and diurnal amplitude variation** (DAV) (from AMSR-E data) for 2009 for pixel near the mouth of the Stewart River.

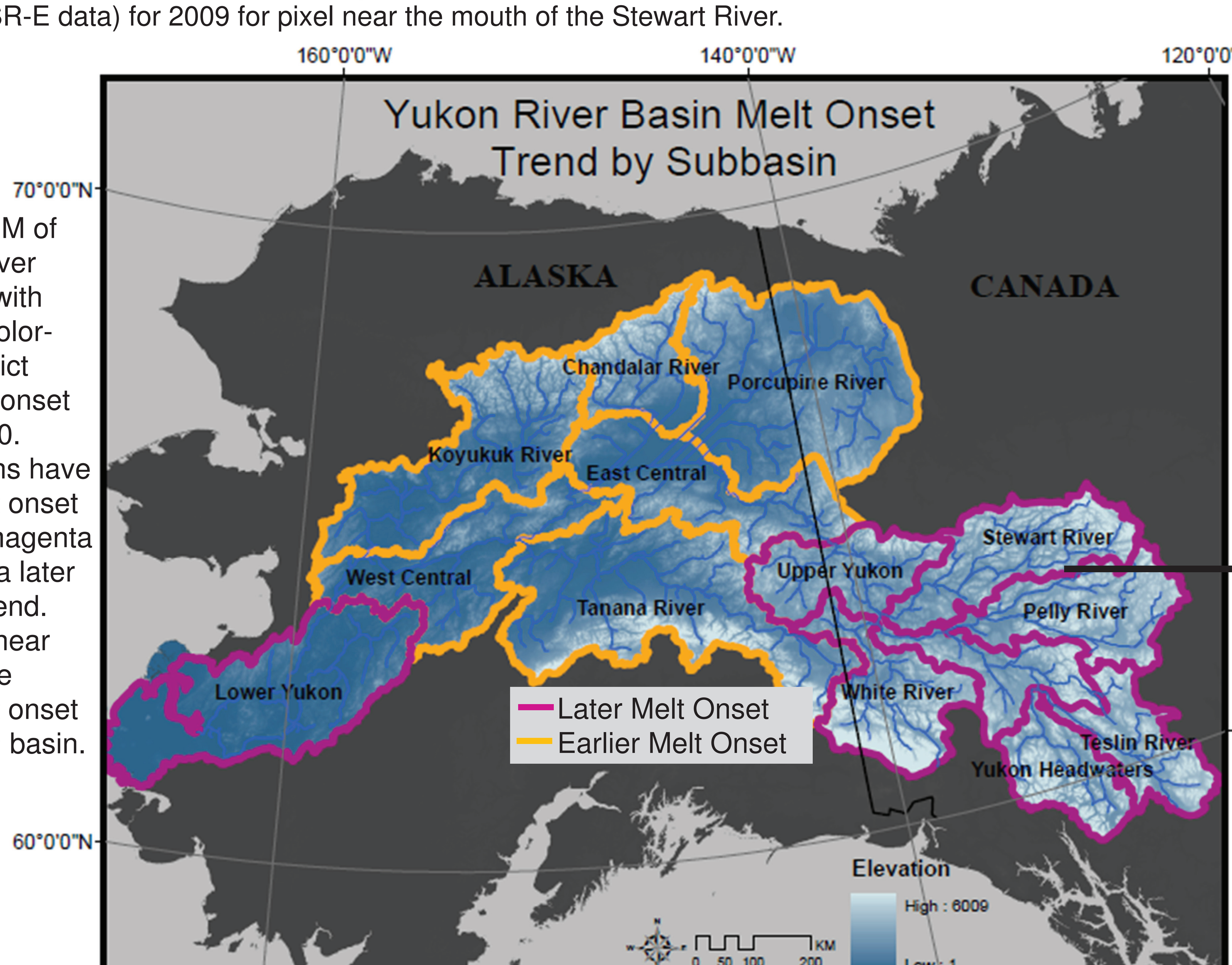


Figure 3. DEM of the Yukon River Basin(YRB) with sub-basins color-coded to depict trend in melt onset for 2003-2010. Orange basins have a earlier melt onset trend while magenta basins have a later melt onset trend. Trends are linear best fits to the average melt onset date for each basin.

Data provided by NSIDC.
Funding from NASA Earth &Space Science Fellowship

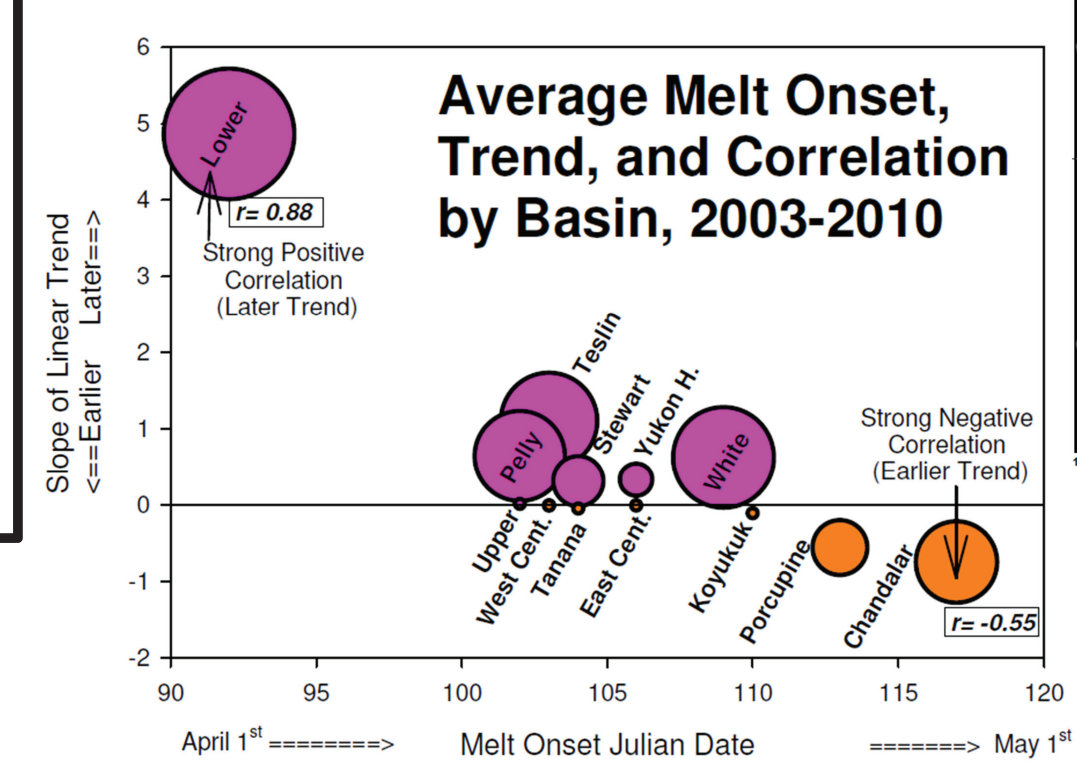


Figure 4. Average melt onset (2003-2010) for each basin (x axis) and linear trend slope (y axis). Bubble size corresponds to size of correlation coefficient with end members denoted (Lower and Chandalar). Note strong later melt onset trends for lower latitude headwater basins.

Snowmelt Runoff Modeling

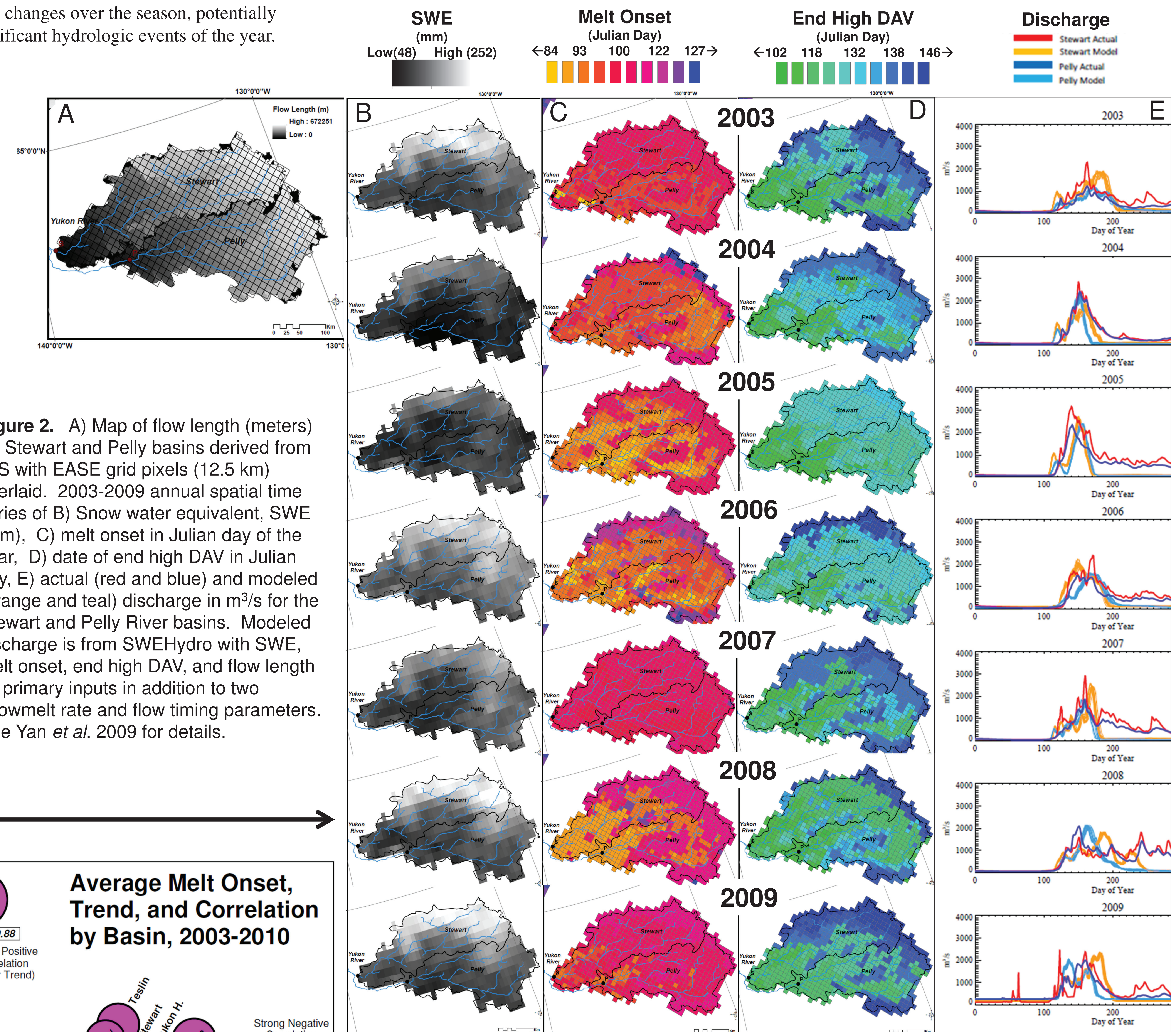


Figure 2. A) Map of flow length (meters) for Stewart and Pelly basins derived from GIS with EASE grid pixels (12.5 km) overlaid. 2003-2009 annual spatial time series of B) Snow water equivalent, SWE (mm), C) melt onset in Julian day of the year, D) date of end high DAV in Julian day, E) actual (red and blue) and modeled (orange and teal) discharge in m^3/s for the Stewart and Pelly River basins. Modeled discharge is from SWEHydro with SWE, melt onset, end high DAV, and flow length as primary inputs in addition to two snowmelt rate and flow timing parameters. See Yan *et al.* 2009 for details.

Conclusions

Remotely sensed brightness temperatures provide a viable means for detecting snowmelt timing in high latitude drainage basins. Such data can be used to drive a simple flux based snowmelt runoff model to effectively model peak discharge and timing. These techniques are important for understanding, studying, and modeling remote regions limited by data collection and instrumentation.

Future Objectives

1) Establish a longer time series using SSM/I Tb data from 1988 to 2010 for more robust trend analysis.
2) Conduct sensitivity analyses of snowmelt and flow parameters in SWEHydro to constrain estimates.
3) Incorporate elevation, permafrost, and land cover to improve and make the model transferable to other basins with varying characteristics.
4) Corroborate the occurrence and effects of early melt events with field data.

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Agency Presentations

A Long-Term Agro-Ecosystem Research (LTAR) Network for Agriculture

Mark R. Walbridge, Steven R. Shafer

Abstract

As the 21st century unfolds, agriculture will face a series of challenges—in the United States and globally—in providing sufficient food, fiber, and fuel to support a growing global population while our natural resources, environmental health, and available arable land decline and climate changes. The unprecedented nature of these challenges creates a growing sense of urgency for transformative changes in agriculture to accelerate progress towards achieving sustainable agricultural systems that maximize production and economic return for producers, minimize environmental degradation, and adapt to changing climate. Achieving such a transformation requires an improved understanding of the complexities of how agro-ecosystems function at multiple scales (i.e., fields to watersheds or landscapes). Long-term research and data collection are essential to achieving this understanding; at stake are the security and safety of our food production systems, our natural resources, and our environment.

Over the past 10 years, there have been frequent calls for the creation of a 'Long-Term Agro-Ecosystem Research' (LTAR) network, similar to the National Science Foundation's Long–Term Ecological Research (LTER) network, to provide a sophisticated platform for research on the sustainability of U.S. agricultural systems. The U.S. Department of Agriculture's Agricultural Research Service (ARS) currently maintains approximately 23 benchmark experimental watersheds and ranges that collect long-term data on agricultural sustainability, climate change, ecosystem services, and natural resource conservation at the watershed or landscape scale. Some of these sites have been collecting data for nearly a hundred years. Here we present a vision for how a subset of these sites could be used to form the core of an LTAR network. Eventually, such a network would link ARS sites with partner sites operated by universities, other research institutions, and (or) other Federal agencies to support multidisciplinary research and funding efforts addressing regional- and national-scale questions using shared research protocols. Such a long-term agro-ecosystem research network would provide the knowledge to substantially improve both agricultural sustainability and the delivery of ecosystem services to a society that increasingly demands that agriculture be safe, environmentally sound, and socially responsible, in addition to being productive and economically viable.

Keywords: agro-ecosystem, climate change, Long-Term Agro-Ecosystem Research (LTAR) network, sustainability, U.S. Department of Agriculture (USDA)

Introduction

During the last 30 years, new technologies, changing demographics and social values, and the globalization of markets, cultures, and competition have produced dramatic changes in the world's food and agricultural systems. Agriculture is shifting from a solely commodity-driven system to one at least partly driven by a global consumer population that values the quality, safety, and nutrition of their food and the way it is produced. Today, consumer concerns about food safety, nutrition, animal welfare, and the environmental footprint of agriculture are driving demands for more locally-produced, organic, humane, high quality, and sustainable agricultural products.

More recently, the global economic downturn, rising energy costs, spikes in food prices, and growing food insecurity, particularly in the developing world, have highlighted the challenges that agriculture faces to meet the food, feed, fiber, and biofuel needs of a growing global population. These and other factors have shifted public policy towards the creation of more

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sustainable agricultural systems, as evidenced by a renewed emphasis on the role of rural lands in providing essential ecosystem services and mitigating climate change and by a rededication towards managing natural resources at the landscape scale.

Recent Farm Bills have placed greater emphasis on conservation, organic agriculture, climate change, and sustainability, while the U.S. Department of Agriculture (USDA) has adopted an “all-lands” approach to conservation aimed at bringing public and private landowners together across landscapes and ecosystems to collaboratively address natural resource conservation issues (U.S. Department of Agriculture 2010).

Reflecting a growing sense of urgency, these policy shifts signal a transformative change in agricultural production, accelerating progress towards achieving the four goals of sustainability as defined by the USDA: (1) satisfying human needs; (2) enhancing environmental quality, the resource base, and ecosystem services; (3) sustaining the economic viability of agriculture; and (4) enhancing the quality of life for farmers, ranchers, forest managers, workers, and society as a whole (U.S. Department of Agriculture 2011).

In agricultural ecosystems, physical and biological processes are linked with social and economic processes. To achieve sustainability, it is essential that we understand how these processes interact and their effects on the environment through space and time (National Research Council 2003). Agricultural research must explicitly identify and address these linkages so that progress in one agricultural sector does not inadvertently create or exacerbate problems in another.

Creation of the scientific foundation necessary to transform the Nation’s agricultural production to meet our sustainability goals demands that the agricultural research, education, and extension communities take a strategic, long-term approach to understanding the aggregate effects of farming at the watershed/landscape scale. Such an approach calls for increasing integrative research by bringing together multidisciplinary teams of scientists from government, academic, and private sector research communities to increase synergies, accelerate progress, and improve cost effectiveness.

Long-term research is vital for agriculture to meet its sustainability goals, including profitability, environmental integrity, and the production of ecosystem services beyond food, fuel, and fiber. Building a sustainable agricultural production system requires a comprehensive, systems-level research approach that is both long-term and geographically scalable (Robertson et al. 2008).

Because of its existing infrastructure, the public research sector is uniquely positioned to undertake the long-term, large-scale, risky research necessary to transform agriculture. Conducting such research provides benefits that cannot be appropriated by individual firms. Long-term research often provides new “platforms” of discovery for multiple private and (or) local applications, with broad public benefits that address national needs and that are widely shared throughout the global community.

The Need for a Long-Term Agro-Ecosystem Research Network

Long-term research is essential to understanding how key agricultural system components interact at the whole system level. Environmental field research carried out over decades plays an important role in understanding the physical, chemical, and biological aspects of agro-ecosystems and can yield critical insights for fields such as agronomy, biogeochemistry, ecology, hydrology, and soil science. Long-term field studies are particularly important for anticipating the environmental effects of shifting agricultural practices, improving the effectiveness of conservation programs, and identifying the broader societal benefits of modern agriculture, such as bioenergy production, carbon sequestration, improved water quality and water-use efficiency, and wildlife habitat.

Developing a Long-Term Agro-Ecosystem Research Network

The scientific community has long recognized the value of maintaining long-term research sites focused on natural resources. Experimental watersheds, such as those maintained by the USDA Agricultural Research Service (ARS) and others, have proven essential for understanding regional hydrologic processes in the United States. The long-term data collected and maintained at these sites are critical for effectively managing water availability and water quality for multiple uses, determining flood effects, and developing, calibrating, and validating watershed models and decision support tools. Experimental watersheds also provide key data on sediment transport and flow needed for the optimal design of culverts, bridges, detention basins, and reservoirs. More recently, these sites are also providing data to help improve the effectiveness of USDA's conservation programs.

In addition to supporting high-quality, location-based research, including collaborations with the academic research community and other Federal agencies, long-term research sites provide the opportunity to take the pulse of agricultural ecosystems and the landscapes in which they exist, to help understand how changes in land management practices and the environment affect the status and trends of landscape condition. Long-term research sites allow us to answer important questions that take longer than the normal 2–5 year project cycle to formulate, allow questions to be addressed against a wide range of environmental conditions, allow the inclusion of episodic events, and enable the detection of important but slow-acting phenomena such as changes in soil carbon, climate, and the effect of land use changes. Long-term data collected at these sites also enable the development, calibration, and validation of models used to forecast these changes. It is well-known that diverse, nontraditional research collaborations form more readily around long-term research networks (National Research Council 2000, 2001, 2005; Robertson et al. 2004; Boody et al. 2005).

A network of long-term agro-ecosystem research sites would improve our understanding of agriculture from a systems perspective, facilitating the optimization of multiple management goals. A network would enable greater integration of biophysical and social sciences to provide solutions with acceptable economic and social costs, improve our knowledge of geographic scalability to ensure that solutions developed at one scale are also effective at larger scales, and allow processes that operate at larger scales to contribute to solutions at the field and farm scale. This scalability is critical to improving agricultural productivity for small landholders in the developing world, increasing their food security and improving incomes—both important U.S. foreign policy goals.

A network of long-term agro-ecosystem research sites would also strengthen ties between agricultural research, outreach, and education, helping to improve the relevance of agricultural research to stakeholders. Such a network would increase public understanding and awareness of agricultural ecosystems, including their social, environmental, and management implications.

Proposed Criteria for an LTAR Network

The critical mass needed to establish an inaugural LTAR program requires a capacity for field-scale experimentation at the site level as well as a capability for stakeholder involvement that exploits existing data sets and regional infrastructure. A reasonable minimum useful duration for an LTAR site is 30 years (Robertson et al. 2008). The Network must include multiple sites that will capture the breadth and diversity of U.S. agricultural production systems. Full value will only be realized when multiple sites function as a network, allowing more robust tests of common hypotheses and comparative analyses in and across different production systems, leading to a comprehensive understanding of agricultural issues.

Key elements for networking include common measurements at multiple sites that provide the foundation for scaling up to regional and national levels. These measurements provide the basis for cross-site syntheses, allowing analyses across gradients of climate change, management intensity, etc. (Robertson et al. 2008).

Criteria for LTAR candidacy could potentially include the length and breadth of existing data records and the availability, accessibility, and organization of datasets (e.g., data in STEWARDS or some other publically-accessible database). A location's infrastructure is also important. Of particular importance is the presence of an instrumented watershed or other long-term research facility (e.g., experimental range) of sufficient size to capture landscape-scale processes and heterogeneity and capable of integrating across small plot, watershed, and landscape scales. The availability of land to support crop and (or) livestock production is critical, as is the site's existing water/energy balance and (or) carbon flux/sequestration research (e.g., Ameriflux, GRACEnet, NEON), with the capacity to integrate across soil, water, and air processes. A formal association with other long term research networks (e.g., Ameriflux, LTER, NEON) is also desirable. Other criteria might include a location's knowledge base, extension capabilities, existing partnerships, and education/outreach potential.

A Proposal for an LTAR Network

As the Federal government's primary intramural agricultural research organization, with significant relevant infrastructure already in place, the Agricultural Research Service (ARS) accepts the challenge of forming an LTAR network for agriculture. Sites for inclusion in the LTAR network will be chosen to represent major U.S. agricultural regions (Northeast; Appalachia; Southeast; Delta States; Corn Belt; Southern Plains; Lake States, Northern Plains; Mountain; Pacific), with attention also paid to major U.S. hydrologic basins at the 2-digit Hydrologic Unit Code scale and National Ecological Observatory Network (NEON) Domains (Figure 1; Table 1).

A steering committee (SC) has already been formed to oversee the site-selection process. Later this year, the SC will solicit "Requests for Information" from the 23 ARS sites that represent candidates for initial inclusion in the emerging LTAR network (Figure 1; Table 1). While site criteria are still being finalized, they will likely include factors such as research productivity, infrastructure capacity, data availability and accessibility, geographic coverage, existing research partnerships, and institutional commitment. Our overall goal is to complete initial selection of ARS LTAR sites by the end of 2011. As we evaluate existing ARS sites for inclusion in the LTAR network, we will begin the process of reaching out to other Federal, academic, and private research organizations, customers, and stakeholders that may have sites suitable for inclusion in the developing LTAR network. Non-ARS sites will be evaluated using the same criteria established for ARS sites, with the overall goal of completing this second phase of LTAR network formation by the end of 2012. When complete, the LTAR network will bring together in a coordinated network experimental watersheds, rangelands, and other sites that address long-term agricultural research at the watershed/landscape scale. The LTAR network will provide an agro-ecosystem complement to the National Science Foundation (NSF) LTER and NEON networks of ecological sites.

Once developed, ARS will offer the LTAR network infrastructure for research and funding partnerships with other Federal agencies, universities, and the

ARS Benchmark Experimental Watershed and Range Research Sites

Code		Code	
BL	Beasley Lake Watershed	NA	North Appalachian Experimental Watershed
BW	Upper Big Walnut Creek Watershed	NP	Northern Great Plains Research Laboratory
CP	Central Plains Experimental Range	NS	Neil Smith National Wildlife Refuge/Walnut Creek South Watershed
CR	Choptank River Watershed	RC	Reynolds Creek Experimental Watershed
FC	Fort Cobb Reservoir Experimental Watershed	RI	Riesel Experimental Watershed
GCa	Goodwater Creek Experimental Watershed	SF	South Fork of the Iowa River Watershed
GCb	Goodwin Creek Experimental Watershed	SJ	St. Joseph River Watershed
JO	Jornada Experimental Range	US	Upper Snake River/Rock Creek Watershed
LR	Little River Experimental Watershed	WC	Walnut Creek Watershed
LW	Little Washita Experimental Watershed	WG	Walnut Gulch Experimental Watershed
MC	Mahantango Creek Experimental Watershed	YR	Yalobusha River/Topashaw Canal Watershed
MT	Mark Twain Lake Watershed		

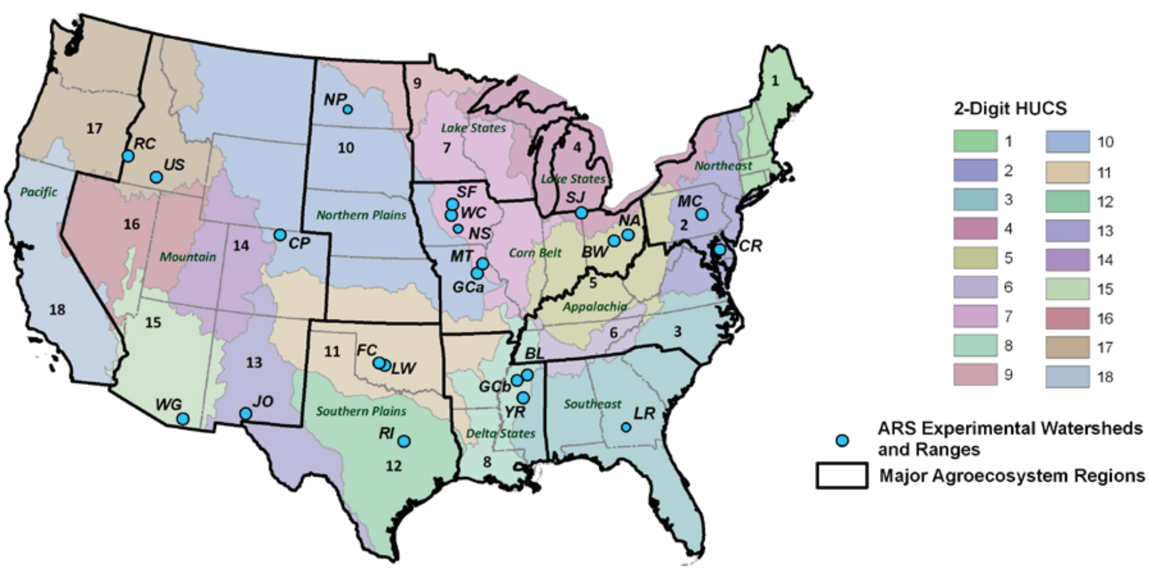

Figure 1. ARS Benchmark Experimental Watershed and Range Research Sites that would be candidates for inclusion in the proposed Long-Term Agro-Ecosystem Research (LTAR) network, plotted vs. 2-digit Hydrologic Unit Code (HUC) watersheds and Major Agro-Ecosystem Regions of the lower 48 United States.

private sector. Through these partnerships, funding will be sought from sources such as USDA's National Institute of Food and Agriculture (NIFA) and NSF to develop technologies and processes for standardized data collection, storage, access, and the development of LTAR Network-wide synthesis products to complement other long-term, multidisciplinary, large-scale Federal research investments (e.g., LTER, NEON).

Vision and Goals of the LTAR

Our vision for the LTAR network:

> *Transdisciplinary science conducted over decades on the land in different regions, geographically scalable, enhancing the sustainability of agro-ecosystem goods and services.*

Our goal in developing this network:

> *To sustain a land-based infrastructure for research, environmental management testing, and education that enables understanding and forecasting of the Nation's capacity to provide agricultural commodities and other ecosystem goods and services under changing environmental and resource-use conditions.*

LTAR Network Operating Principles

- Develop research questions that are shared and coordinated across sites.
- Provide the capacity to address large-scale questions across sites through shared research protocols.
- Collect compatible datasets across sites, and provide the capacity and infrastructure for cross-site data analysis.
- Facilitate and foster shared engagement in thinking and acting like a network.

Table 1. Potential ARS Experimental Watershed and Range LTAR Sites

Site code*	ARS lab location	Established	Record (years)	Area (km^2)	Network affiliations	NEON domain
BL	Oxford, MS	1995	17	9	CEAP	Ozarks Complex
BW	Columbus, OH	2004	8	492 (17)	CEAP	Appalachians/CP**
CP	Cheyenne, WY	1939	73	63	LTER/NEON	Central Plains
CR	Beltsville, MD	1985 (2004)	27 (8)	2057 (395)	CEAP	Mid Atlantic
FC	El Reno, OK	2004	8	786	CEAP	Southern Plains
GCa	Columbia, MO	1971 (1969)	41	73	CEAP	Prairie Peninsula
GCb	Oxford, MS	1981	31	21	CEAP	Ozarks Complex
JO	Las Cruces, NM	1912	100	780	CEAP/LTER/NEON	Desert Southwest
LR	Tifton, GA	1967 (2002)	42 (10)	334 (5208)	CEAP	Southeast
LW	El Reno, OK	1961	51	610	CEAP	Southern Plains
MC	University Park, PA	1967	45	420 (7)	CEAP	Northeast
MT	Columbia, MO	2005	7	6417	CEAP	Prairie Peninsula
NA	Coshocton, OH	1935	77	4		Appalachians/CP**
NP	Mandan, ND	1912	100	800	NEON	Northern Plains
NS	Ames, IA	1996	15	52		Prairie Peninsula
RC	Boise, ID	1960 (1962)	52 (50)	239	CEAP	Great Basin
RI	Temple, TX	1937	75	3	CEAP	Southern Plains
SF	Ames, IA	2002	9	780	CEAP	Prairie Peninsula
SJ	West Lafayette, IN	2002	10	205	CEAP	Great Lakes
US	Kimberly, ID	2005	7	820	CEAP	Great Basin
WC	Ames, IA	1992	19	51	CEAP	Prairie Peninsula
WG	Tucson, AZ	1954	58	150	CEAP	Desert Southwest
YR	Oxford, MS	1999	9	110 (5920)	CEAP	Ozarks Comples

*Site Codes follow Figure 1 **CP, Cumberland Plateau

Conclusion

Globally, agriculture is confronting tremendous challenges in providing for a growing global population under the constraints of dwindling natural resources, environmental degradation, and climate change. If agriculture is to meet the food, feed, fiber, and bioenergy needs of an estimated 9 billion people by 2050, the world's food and agricultural system must become more sustainable.

The National Academy of Science has called for transformative changes to agriculture, including *"...production systems and agricultural landscapes that are a significant departure from the dominant systems of present-day agriculture … that capitalize on synergies, efficiencies, and resilience characteristics associated with complex natural systems and their linked social, economic, and biophysical systems … integrating information about productivity, environmental, economic, and social aspects of farming systems to understand their interactions and address issues of resilience and vulnerability to changing climatic and economic conditions."* (National Research Council 2010, p.525–526.)

Achieving such a transformation requires a better understanding of the complex interactions of the biological, chemical, and physical processes of agro-ecosystems from a long-term systems perspective. The Long-Term Agro-Ecosystem Research network is thus urgently needed to answer some of the important large-scale questions posed by the challenges of the impending effects of climate change and water scarcity, including how episodic events such as floods, drought, and pest and pathogen outbreaks might affect an agro-ecosystem's ability to produce agricultural products or provide valuable ecosystem services. The LTAR network will also be invaluable for detecting important but slow-acting phenomena such as changes in soil carbon, climate, and the effect of land use changes. Such knowledge is vital if we are to achieve our goals for a sustainable agricultural future.

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U.S. Forest Service 100-Year History of Watershed Studies on Experimental Forests and Ranges

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Abstract

The U.S. Forest Service initiated its watershed research program in 1909 with the first paired watershed study at Wagon Wheel Gap, CO. It has since developed the Experimental Forests and Ranges Network, with over 80 long-term research study sites located across the contiguous United States, Alaska, Hawaii, and the Caribbean. This Network provides a unique, powerful continental research platform for study of complex environmental and societal problems at the local, regional, and landscape scales. The study of water and ecosystems through paired watershed studies has been an integral component of many of these studies with continual establishment of sites up to the mid 1960s. The U.S. Forest Service continues to maintain and develop a robust program emphasizing long-term research on watershed science. This paper gives an overview of U.S. Forest Service catchment research, emphasizing paired-watershed studies and highlighting important advances in ecosystem science, as well as lessons learned over the last century of research.

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