

Second Interagency Conference on Research in the Watersheds

May 16 – 18, 2006



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ABSTRACT

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The report comprises papers and abstracts from the Second Interagency Conference on Research in the Watersheds, May 16-18, Coweeta Hydrologic Laboratory, Otto, NC. Authors from several agencies (e.g., USDA ARS, US-EPA, USFS) and the private sector contributed papers on topics ranging from new technology and analytical procedures, modeling, and hydrologic responses to disturbance. To ensure timely distribution, the report has been reproduced essentially as submitted by the authors. Papers received a minor amount of editing for consistency and formatting, but they have not been peer reviewed. The authors' views are their own and do not necessarily reflect those of the U.S. Department of Agriculture, the USDA Forest Service, the publisher, or the Coweeta Hydrologic Laboratory. Mention of trade names or commercial products in this report is solely for the purpose of providing specific information and does not imply recommendation or endorsement by the U.S. Department of Agriculture.

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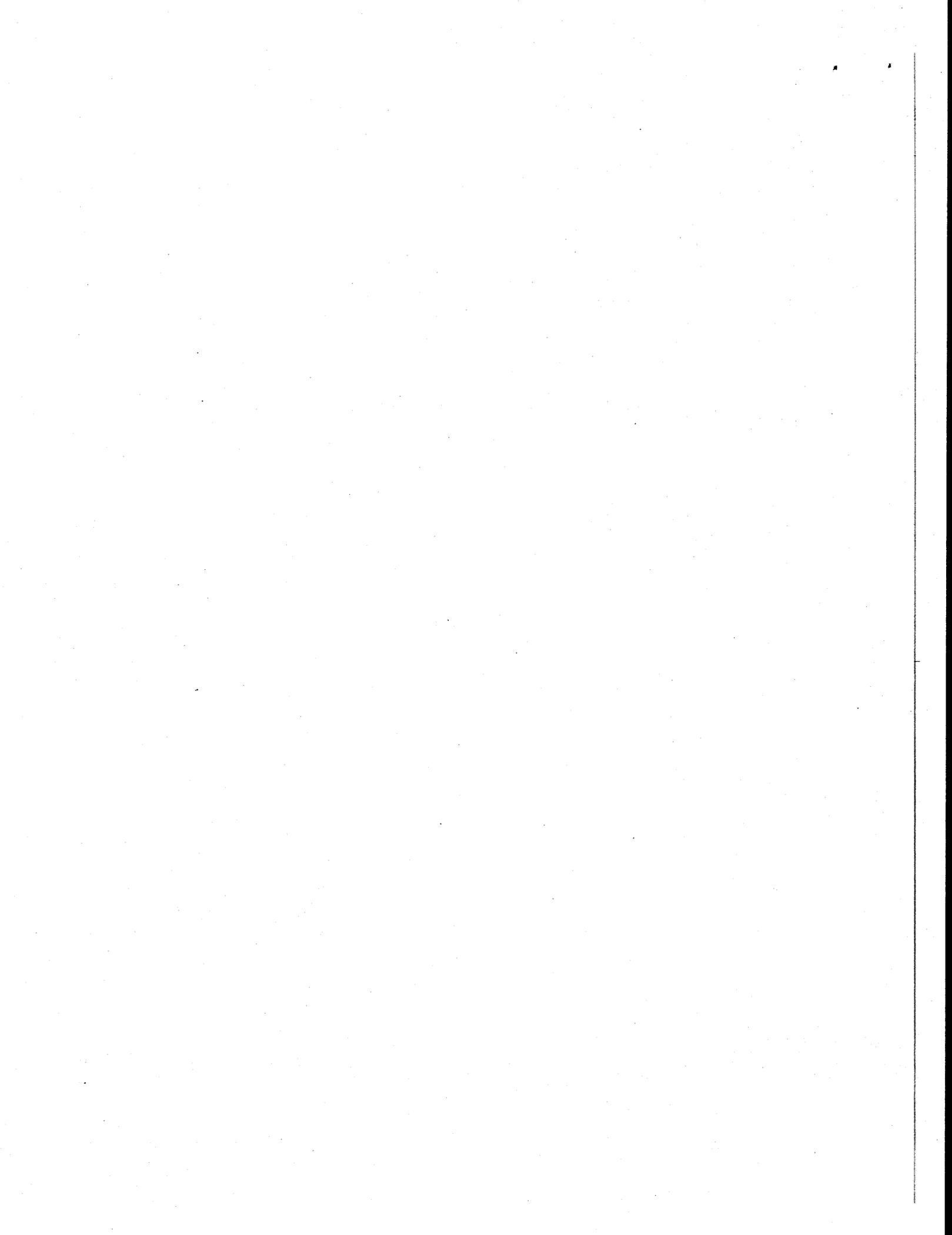
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PREFACE

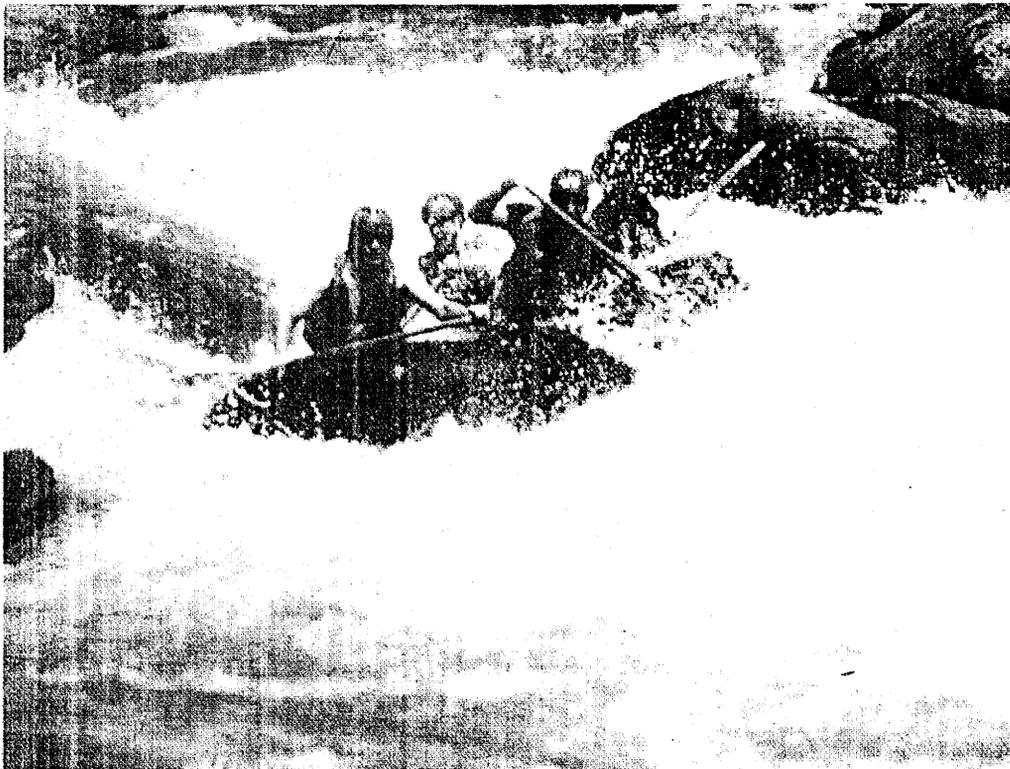
Watershed science will play a key role in providing natural resource managers and policymakers with the knowledge and tools required to address water-based challenges of the 21st century. The interplay among increasing demands for clean and dependable water, accelerated land use change, and climate change and variability requires an interdisciplinary approach that brings together social scientists, hydrologists, ecosystem ecologists, and physical scientists. Most importantly, watershed science needs to be at the forefront of informing policymakers and community leaders of the causes and consequences of contemporary and future decisions regarding land use and management. The 2nd Interagency Conference on Research in the Watersheds provided a forum for watershed scientists from federal agencies, state agencies, universities, and the private sector to share current research tools, field studies, and concepts addressing a wide array of watershed research questions.

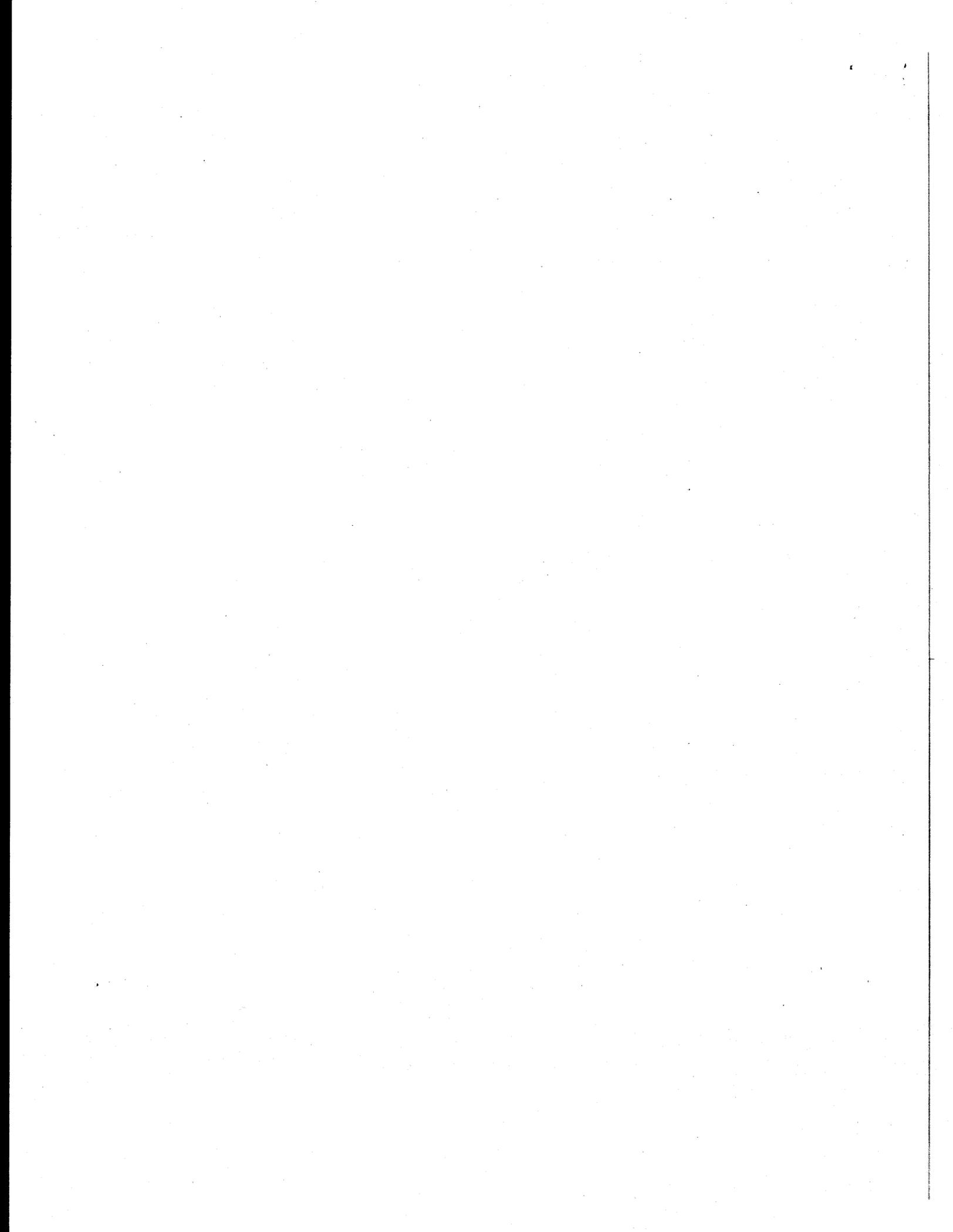


**Second Interagency Conference on
Research in the Watersheds**

May 16 – 18, 2006

**Challenges and Opportunities for
Watershed Research**





DISTURBANCE AND WATER RELATED RESEARCH IN THE WESTERN UNITED STATES

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Abstract—Water plays a critical role in agriculture, energy, recreation, conservation, transportation, and indeed, life in the western United States. Water supply and variability in supply was paramount even before European settlers moved into the region. As demands on water have increased with time, all water related issues have also increased in importance to civilization. Water quantity issues drive western economics, politics, and demographics. Disturbance affects water quantity because precipitation and water use are controlled in part by land cover. The biogeochemical controls on water quality are closely tied to water quantity, but are also affected by landscape variability and both natural and anthropogenic disturbance. Slow recovery of forest vegetation may prolong the disturbance impacts on water quality in arid western basins compared to their relatively moist eastern counterparts. Sedimentation is a common cause of water quality impairment in actively managed landscapes. Riparian areas provide critical buffers to aquatic ecosystems from upland disturbances and may be integrators, magnifiers, or filters depending on the state of both the upland and riparian systems. Fisheries attract more attention than other aquatic resources and management of western water systems has had widespread impacts on fish habitat and populations. A general review of western issues with a Rocky Mountain and Fraser Experimental Forest focus is provided for water quantity, quality, fluvial, riparian and native fisheries issues.

INTRODUCTION

Few would argue that water is the most critical natural resource in the western United States. For over a century water has been the foci of conflict and failure, survival and success on western lands. The Colorado River basin dominates the landscape of the western U.S. and shares characteristics with other major western water courses such as the Columbia, Missouri, and Rio Grande. These characteristics include high-elevation headwaters dominated by snow, long distances to ocean bodies flowing through arid regions, high demands on supply highlighted by over-appropriated water resources, and substantial agricultural requirements conflicting with increasing municipal and industrial demands. Western basins are also largely administered by government entities. Ownership of land area in the Colorado basin, for example, is only 19% private, with 56% federal, and 25% state and Indian (Weatherford and Brown 1991). These government lands, particularly the federally managed areas, are also the primary water source areas because they tend to be the upper-elevation forested regions that receive the greatest annual precipitation.

Headwaters of mid-latitude, western rivers systems are located in the Rocky Mountain cordillera, mountains of the basin and range region, Sierra Nevada, and coastal ranges. These physiographic features are particularly effective in producing the orographics that lead to higher precipitation rates than observed in nearby lowland regions. Northern hemisphere circulation produces moist westerly air mass flows from the Pacific that collide with these orographic barriers and the circulation is such that most of the precipitation comes as snowfall during the fall, winter and spring months. The end result is that the majority of annual precipitation for much of the region, as much as 75%, is stored in winter snowpacks in mountain regions. Potential evapotranspiration is also reduced at higher elevations due to cooler temperatures and limited growing seasons. These forested and alpine regions produce as much as 90% of the annual runoff as the snow melts in the spring and early summer. Because these source areas are largely federally managed, factors affecting the accumulation and subsequent ablation of the snowpack, as well as on-site consumptive use of water, place great pressure on agencies responsible for the source areas. As water resources become more limited due to increasing demand, the federal agencies managing the source areas are placed in a role of increasing conflict with multiple users with varied needs. Climate variability, recent prolonged droughts in particular, has accelerated both the demand and conflict in managing the resource. Management decisions must include consideration of downstream users' water rights, recreation, threatened and endangered species, in-stream flows, sedimentation, fire and fuels management, forest production, fisheries and wildlife, hydroelectric power generation, among others.

This paper briefly discusses western water issues from the perspective of research on streamflow generation from headwaters catchments, water quality, sediment production, riparian habitats, and native fisheries. We discuss the

broad western context of these issues and focus on Colorado and the Fraser Experimental Forest (FEF), where much of the authors' contemporary research is located. Past, current and future research on these issues are discussed.

WATER QUANTITY

Water supply in western watersheds follows natural cycles closely related to continental scale atmospheric forcings. In time of above-normal precipitation, the West enjoys ample supply in most regions. Low precipitation regimes affect almost all human endeavors negatively. Storage projects have ameliorated short-term deficits to some degree, but the time scale of many drought cycles exceeds storage capacity. In recent decades the construction of additional storage facilities has largely fallen out of favor for many reasons and efforts to increase runoff have centered on increasing wintertime snowpack storage and reducing summertime transpiration losses. Researchers have shown that runoff from headwater basins may be altered by changing the vegetation within the basin. Almost a century ago in southern Colorado, Bates and Henry (1928) conducted the first paired watershed study in the world at Wagon Wheel Gap and found that a reduction in forest cover produced an increase in local streamflow. Subsequent studies have refined our understanding of the processes that lead to increased streamflow with decreased forest cover. Bosch and Hewlett (1982) and MacDonald and Stednick (2003) summarize much of the research conducted over the last century on treated watersheds in the western U.S.

Much of our current understanding of the effects of forest management on runoff in the subalpine zone comes from studies conducted at the Fraser Experimental Forest since its establishment in 1937 near Fraser, Colorado (e.g. Wilm and Dunford 1948, Anderson and others 1976, Troendle 1983, Troendle and King 1985, Schmidt and Troendle 1989, Schmidt and others 1998). The FEF is a generally north-facing subalpine basin dominated by lodgepole (*Pinus contorta*), Engelmann spruce (*Picea engelmannii*), subalpine fir (*Abies lasiocarpa*), and some aspen (*Populus tremuloides*), with elevations ranging from about 2680 m a.s.l. (8800') to 3900 m (12,800'). The basin has a snow-dominated hydrological regime, with 70-80% of the annual precipitation falling as snow and about 90% of the annual runoff derived from melting snow.

Results from the Fool Creek paired watershed study within the Experimental Forest were first reported by Goodell (1958) who found that clear cutting 40% of the basin in alternate cut-leave patches produced a marked increase in annual flow when compared to the adjacent East St. Louis control watershed. Troendle and King (1985) later showed the long-term effect after revisiting the 28-year post harvest period. They observed a sustained average effect of increased streamflow of 82 mm (40%) resulting from increased snowpack accumulation of 28 mm (9%) water equivalent and reduced summertime losses. Increased snowpack results from reduced interception of snowfall by the canopy, of which a significant percentage is subsequently lost to sublimation before it can be incorporated into the snowpack. Removing the canopy allows snowfall to accumulate directly to the snowpack, which is then stored with only small losses until it melts and is available for runoff. Canopy removal also reduces summertime losses due to transpiration and plant depletion of soil moisture, resulting in increased runoff. Troendle and King also observed that peak flows occurred 7.5 days earlier with a 23% increase in the peak flow on average, over the period. Finally, they predicted a time to hydrologic recovery after treatment of approximately 70 years based on the regression relationship developed for the 28 year period.

Elder and others (in preparation) have examined the continuous long-term Fool Creek record through the 2005 water year, giving 49 years for post-treatment analyses (Figure 1). Results show that differences in snowpack water equivalent are no longer statistically significant. However, annual streamflow still shows a 60 mm (29%) increase on average over the 49 year period. Peak flows still arrive 7 days earlier than the pre-harvest period, but peak flow volume has been reduced to a 16% increase over the calibration period. The time to hydrologic recovery has dropped markedly from 70 years to slightly less than 60 years (Figure 2). This shorter predicted recovery time may be influenced in part by recent drought conditions observed in the last decade (Figure 1).

In addition to changes in the annual water yield, changes in the seasonal snowmelt hydrograph were documented following forest harvest. The primary change in the annual snowmelt hydrograph comes during the rising limb, with no significant change in the falling limb. The rising limb from the subalpine region occurs between the beginning of May and the middle of June. Increased yields following harvest are also three to four times greater on wet years than dry years when deficits are greatest. From a water manager's perspective, these factors show that increased flow occurs during the season of little need, which means that additional water storage must be constructed to realize the benefits of higher water yields.

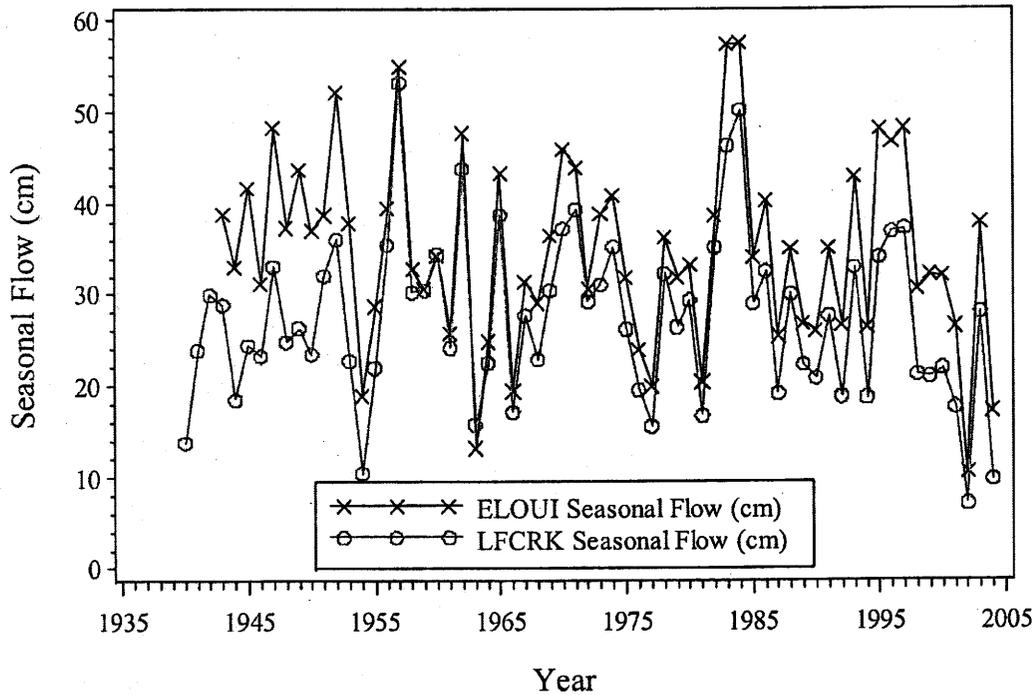


Figure 1. Unit area discharge record for the Fool Creek (1940-2004) and East St. Louis (1943-2004) paired watershed experiment. Treatment was completed in 1956.

There are still a number of open questions related to water supply in western watersheds. Because runoff is controlled by a snow-dominated hydrological regime, better understanding of snow-related hydrological processes offers the greatest potential for effectively managing watersheds. Additional studies are needed on spatial variability of snow accumulation and melt processes, vegetation effects on snow accumulation, climate variability and its effect on snow processes, and hydrologic pathways from the snowpack surface through the snowpack, the soil matrix and into the stream channel. The hydrologic consequences of extensive conifer mortality caused by bark beetle and other insects represent another critical unknown. Millions of acres are currently infested by mountain pine beetle (MPB) resulting in substantial overstory mortality in large watersheds. Tree mortality caused by mechanical harvesting, insect or disease all change the hydrologic cycle similarly, so one might expect a commensurate increase in water yield following the current outbreak. However, similarity between disturbance effects depends on many factors such as extent of impact, forest species composition, forest structure, residual vegetation, local climatic regime, etc. One important question is whether the residual stand will simply use excess water on site. Better understanding of tree water use, both in disturbed and undisturbed scenarios will help answer this question. Transpirational water use has traditionally been a weak area in our understanding of basic water balance at the stand, hillslope and watershed scale.

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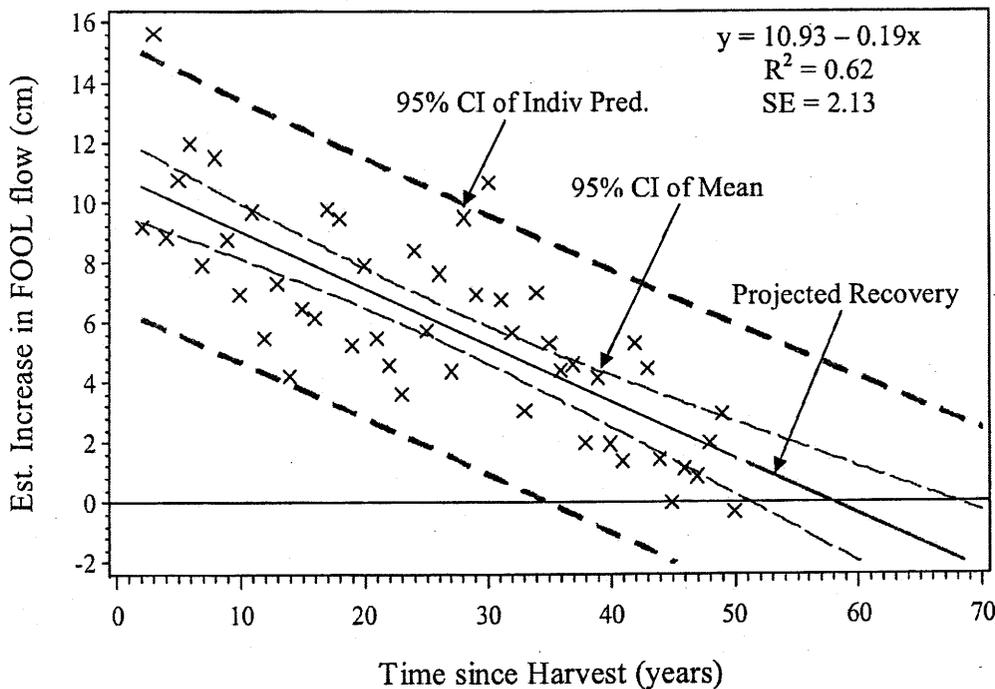


Figure 2. Regression of expected increase in Fool Creek yield based on the pretreatment calibration period versus observed yield since the clear cut treatment was completed in 1956.

Transpiration in Rocky Mountain Forests

Rocky Mountain Forests are characterized by relatively extreme environments ranging from cold, moist high elevation settings to lower elevation forests where persistent water deficits exist throughout much of the growing season. Consequently, water and temperature are the primary environmental factors limiting tree growth and water use in Rocky Mountain forests. Understanding how species respond to and cope with these environmental factors is critical for accurate models of site water balance and for developing mechanistic models of how forests will respond to disturbance, climate and management scenarios.

Plant water use and productivity are inextricably linked because under photosynthetic conditions, stomata operate to both enhance photosynthesis and avoid dehydration induced damage. As plants open stomata to acquire CO_2 for photosynthesis, water is pulled from the soil to the leaf via the water potential gradient that exists between the air at the leaf surface and the soil. As a result, plant water status during photosynthesis is regulated to permit the transpiration that necessarily accompanies stomatal opening, and is kept from falling below damaging levels to prevent disruption of function. A species' ability to balance the uptake of carbon for growth and metabolism with water loss via transpiration ultimately determines their productivity and survival, especially in arid and semi arid environments.

It is well recognized that stomata play a critical role in regulating plant water status in terrestrial vegetation but despite decades of research, the actual mechanism remains unclear. A large body of work has shown convincing, yet sometimes conflicting evidence that stomata respond to a variety of water relationship parameters under light-saturating conditions. Stomatal response has been linked to humidity (Mott and Parkhurst 1991), transpiration (Cowan 1995; Franks and others 1997; Mott and Buckley 1998), soil moisture (Loewenstein and Pallardy 1998; Tardieu and Simonneau 1998), and hydraulic conductance, and actual response mechanisms have been investigated for humidity, transpiration and soil moisture. Recent work suggests that the actual stomatal response mechanism is closely related to changes in the hydraulic conductance of the flow path (Hubbard and others 2001) and that stomatal control is achieved at the guard cells through a hydraulically-mediated feedback (Buckley 2005).

This type of control is consistent with most of the current and past research and remains the most plausible explanation for a universal mechanism.

Conifer species dominate most Rocky Mountains forest communities. Conifers are well suited to soil water deficits that prevail through much of the growing season in the region. In general, conifer species exhibit tight control of stomatal opening to prevent stem and leaf water potential from falling below critical levels leading to cavitation of the water column. Other factors that facilitate growth and survival of Rocky Mountain conifers include the physiological adjustments of hydraulic architecture and the ratio of transport tissues to leaf area.

Although most tree species in the Rocky Mountain region exhibit tight stomatal control of leaf gas exchange, the degree and timing varies significantly among sites and species. Pataki and others (2000) showed that four common Rocky Mountain species (*Pinus contorta*, *Abies lasiocarpa*, *Populus tremuloides* and *Pinus flexilis*) exhibited very different stomatal and transpiration responses to seasonal increasing soil water deficits at a subalpine forest site in Wyoming. A decrease in maximum transpiration was evident for all species with increasing soil water deficits but *Abies lasiocarpa* showed a 50% decrease in transpiration regardless of air saturation deficit (D). Conversely, transpiration was greater on low versus high D days for *Pinus contorta* while *Populus tremuloides* showed less stomatal sensitivity to soil moisture than any other species.

Past research on plant water relations at FEF has focused on development of heat pulse techniques to measure whole tree water use (Swanson 1962) and the response of stomata to light, relative humidity and moisture stress (Kaufmann 1982; Kaufmann 1985a). Swanson calculated sap velocities for *Pinus contorta* and *Picea engelmannii* and derived some of the first transpiration rates for these species at FEF (Swanson 1967). Kaufmann produced some of the first modeled estimates of stand level water use at FEF (Kaufmann 1984; Kaufmann 1985b). His model suggests that in pure stands of the four species, consumptive water use ranks from highest to lowest for *Picea engelmannii*, *Abies lasiocarpa*, *Pinus contorta* and *Populus tremuloides*, respectively.

Current water relations research at FEF is focused on understanding the contribution of the three dominant species (*Pinus contorta*, *Abies lasiocarpa*, and *Picea engelmannii*) to mixed-species subalpine forest water use. Initial research on a forested and regenerating hillslope at FEF suggests that canopy conductance and its response to vapor pressure deficit is different for old and young trees (Figure 3). Future work will focus on quantifying and scaling species and age class differences in canopy conductance and water use as we assess the impacts of MPB infestation and management on site water balance for watersheds at FEF.

Rocky Mountain forests occupy some of the highest elevation sites in the US. At these sites, temperature exerts significant control on productivity, water use, species composition, and tree line. For example, Jobaggy and Jackson (2000) showed that temperature explained 78% of the global variation in tree line elevations. Differences in

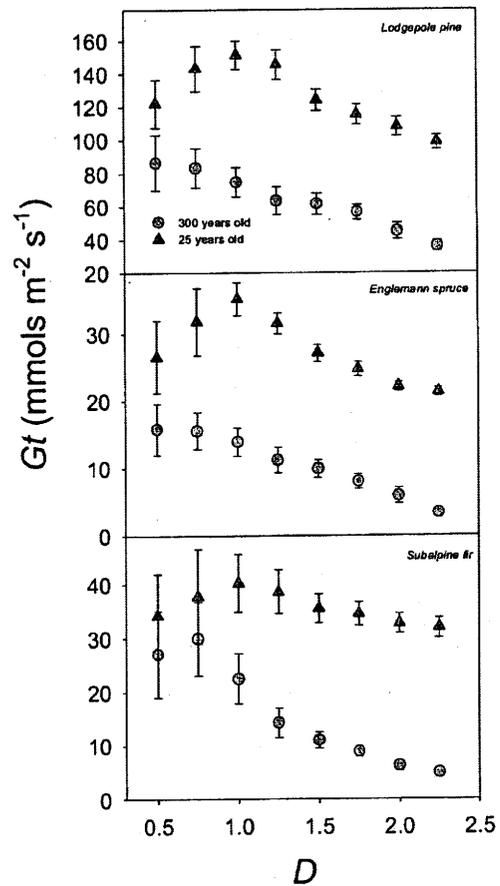


Figure 3. Canopy conductance versus air saturation deficit for the dominant tree species in 300 and 25 year old stands at FEF.

temperature during the summer months seem to control tree line elevation while temperatures during the winter months appear to control species composition.

Temperature may also exert considerable influence on tree water use. Research at FEF indicates freezing nighttime air temperatures limit leaf gas exchange the following day (Figure 4) for a range of age classes and species. The extent that freezing temperatures limit forest water use will depend on the duration and magnitude of these events in early spring and late fall. Because watersheds in the Rocky Mountain region typically span large elevation gradients, accurate scaling of vegetation water use to the watershed will likely need to account for air and soil temperature effects on canopy conductance and vegetation water use.

In addition to water availability and air saturation deficit, plant water status is dependent on the hydraulic architecture of the plant and the ratio of transpirational (e.g. leaves) and absorptive/transport (e.g. roots and sapwood) tissues. Consequently, in dryer environments, trees may allocate more resources to water uptake and transport tissues relative to leaf area and make physiologic adjustments in their hydraulic architecture. For example, *Pinus ponderosa* grows successfully along a range of xeric and mesic sites but exhibits distinct differences in hydraulic architecture and the ratio of leaves to absorptive and transport tissues. Callaway and others (1994) found *Pinus ponderosa* had higher ratios of leaf area to sapwood area in montane relative to desert environments and Maherali and DeLucia (2000) found increased xylem hydraulic conductivity in the same species growing in dry versus moist sites. Physiological adjustments are also likely important as trees grow larger and taller because increased height is accompanied by increased resistance to water flow. Recent research suggests that larger, older trees often allocate more resources to transport tissues relative to leaf area (Hubbard and others 2002; McDowell and others 2002). Increases in forest age and size have also been associated with increased sapwood permeability (Pothier and others 1989). Increased understanding of the physiological adjustments that trees make in response to changes in environment, age and size will be critical for better development of mechanistic models of how tree and forest water use will respond to disturbance and climate change.

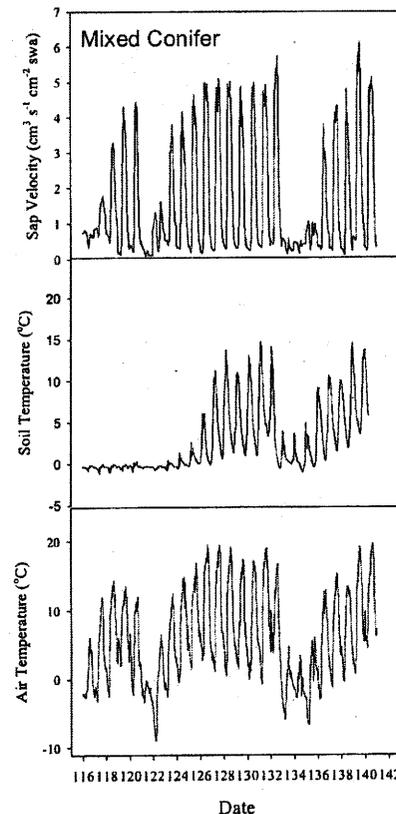


Figure 4. Effect of air and soil temperature on sap velocity in mixed conifer stands at FEF.

WATER QUALITY AND BIOGEOCHEMISTRY

Biogeochemical processes influence the water quality of headwater basins throughout the western US. There is growing concern that changing atmospheric inputs, most notably increased nitrogen deposition, have begun to alter watershed processes and threaten aquatic resources in high elevation basins (Fenn and others 2003; Galloway and others 2003). Rapid population growth in areas such as Colorado's Front Range has led to higher emissions from vehicle exhaust and coal-fired power generation and resulted in nitrogen loads that are 15 to 30 times above pre-industrial levels (Sievering and others 1996). During the past two decades, N inputs from precipitation have more than doubled at several high-elevation sites adjacent to Front Range population centers (NADP 2006; Figure 5). In some Rocky Mountain ecosystems, supply may exceed the capacity of plant, soil and microbial sinks to take up additional N and excess nitrogen may be exported from the system (Baron and others 1994; Fenn and others 1998; Burns 2003). In response to public concerns regarding increased N loading east of Colorado's continental divide, the director of Rocky Mountain National Park recently proposed a critical load (1.5 kg N / ha/yr) aimed at protecting the Park's terrestrial and aquatic ecosystems (Hartman 2006).

Current ability to set critical loads for high-elevation ecosystems is limited not only by regional and local differences in atmospheric deposition (Figure 5), but also by uncertainties relating to internal N cycling and storage (Williams and others 1996). The biogeochemical processes that regulate nutrient export differ with elevation, aspect,

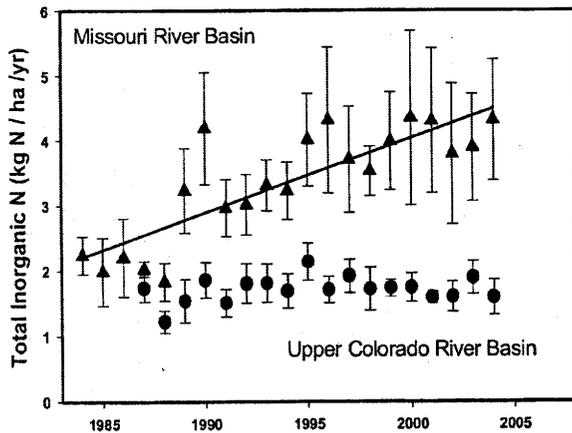


Figure 5. Annual nitrogen deposition in wetfall to high elevation sites (> 2500 m) east and west of Colorado's continental divide. Figure shows means and SE for 4 east and 2 west-side collection sites (NADP 2006).

watershed area, forest species composition and age, and edaphic and geologic attributes. Since soils represent the largest N sink in mountain ecosystems, the response of belowground processes to changes associated with land use practices, climate variability and landscape heterogeneity are critical to understanding the biogeochemical controls on water quality. For example, nitrate released in spring melt water is known to relate more closely to soil microbial activity (Brooks and others 1996) than to nitrate eluted from the snowpack. Like most terrestrial ecosystems, subalpine forests are limited by N supply (Fahey and others 1985) and they have the capacity to retain the bulk of N inputs (i.e. >90%; Stottlemeyer and others 1997). Fertilization studies demonstrate that nutrients released from N-amended forest soils result from excess supply relative to plant and microbial demand (Binkley and Hogberg 1997; Perakis and others 2005) as well as from enhanced nitrate production (Fenn and others 2005). Further, critical load estimates for high-elevation watersheds

must combine terrestrial nutrient uptake, transformation and storage processes with transfers between alpine and subalpine zones and terrestrial and aquatic ecosystems (Stottlemeyer and others 1997; Seastedt and others 2004). Landscape Patterns in High-Elevation Biogeochemistry

Research at the Fraser Experimental Forest has advanced understanding of the links between terrestrial processes and streamwater chemistry in mixed alpine/subalpine basins (Stottlemeyer and Troendle 1992; Stottlemeyer and others 1997). Water draining the alpine portion of the Lexen watershed contains 2 to 10-fold more nitrate than the subalpine portion of basin (Figure 6). At treeline and within the subalpine forest, streamwater nitrate concentration is highest during the winter and early spring when biological demand is low, then declines to near detection levels during the summer growing season. Elevated nitrate in soil solution (not shown) contrasts to the declining streamwater nitrate concentration at snowmelt and indicates that biological N immobilization by subalpine plants and microbes efficiently retains inorganic N (> 90%) before it reaches the stream (Stottlemeyer and others 1997). Low plant biomass and nutrient demand in alpine plant communities coupled with rapid leaching through coarse-textured alpine soils limit biological N retention and produce the more gradual decline in growing season streamwater nitrate.

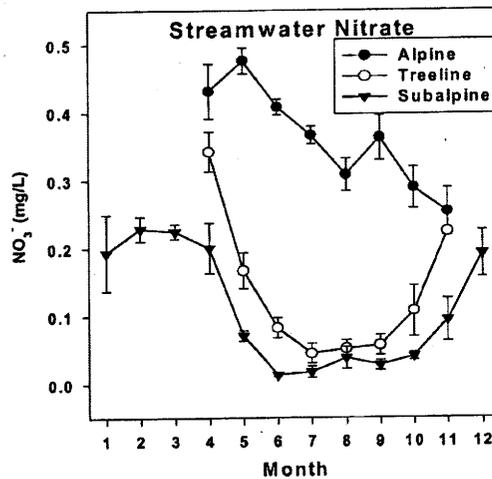


Figure 6. Streamwater nitrate concentrations across the alpine-subalpine ecotone at FEF (Reworked from Stottlemeyer and others 1997 and FEF unpublished data).

The influences of snow cover and snow redistribution on the soil abiotic environment are key factors controlling nitrate export to surface waters in high elevation landscapes (Brooks et al 1999; Burns 2004). For example, seasonal freeze-thaw cycles that disrupt the rhizosphere environment have been shown to alter nitrification rates and release nitrate from roots, microbes and soil compartments (Groffman and others 1999). Temperature-regulated fluctuations in soil N processes are also evident at the watershed-scale where streamwater nitrate has been shown to track extreme cold periods (Mitchell and others 1996; Goodale and others 2003). In high-elevation catchments, the short growing season, weakly-developed soils and limited residence time of snowmelt within groundwater flow paths restrict retention of additional N inputs (Fenn and others 1998). Nutrient movement from alpine soils and export in stream water is coupled to snow melt, shallow subsurface lateral flow and stream discharge.

The strong relationship between streamwater inorganic N and alpine land cover (Figure 7), as measured in 18 catchments at FEF, demonstrates the relative importance of alpine biogeochemistry on watershed N export (Seastedt and others 2004). Still, it remains unclear to what extent landscape-scale variability in snow accumulation and soil development contribute to the overall pattern of watershed N export.

Legacy of Forest Harvesting

Research conducted at FEF has shown that removal of the subalpine forest canopy increases snow accumulation, shallow subsurface flow and nutrient export (Troendle and Reuss 1997; Reuss et al 1997). Fraser's paired watershed studies (Fool and Deadhorse Creeks) document augmented streamflow (Elder and others, in prep) and subsurface nitrogen export (Starr 2004) for at least 50 and 20 years, respectively, following clear cut harvests. A recent comparison of soil processes in regenerating clear cut and old growth stands in these basins indicates that nitrate production and release remain significantly altered 25 years after harvesting and may require a half century to recover from canopy removal (Rhoades and Hubbard 2005).

We found that in both 25- and 50 year-old harvest studies, snowmelt ion exchange resin (IER) nitrate was more than two times higher in harvested areas (Figure 8) and that nitrate represented a greater proportion of total IER-N (nitrate plus ammonium). In 25-year-old harvest areas, there was 1.7-fold more total IER-N released compared to adjacent uncut stands during spring snowmelt. We also measured 1.7 and 0.3 kg / ha more N mineralized and nitrified annually in the mineral soil of 25-year-old regenerating forest compared to uncut stands. Significant differences occurred during summer months when N turnover in harvested areas was twice that in adjacent old growth. N transformations did not differ between the 50-year-old stands and adjacent old growth forest. When combined with greater snow accumulation and subsurface flow in regenerating harvests, our findings indicate the potential for elevated watershed N export for half a century after subalpine forest harvest.

FLUVIAL GEOMORPHOLOGY AND SEDIMENT

A primary issue facing western water managers is increasing pressure on water resources from human population growth occurring over substantial portions of the western US. This pressure has caused state water users to look to National Forests to tap additional unappropriated waters and to establish only minimal instream flow protections for purposes of channel maintenance. Hence, maintaining water and aquatic resources on National Forest lands has become increasingly contentious. In the legal arena, the Forest Service has asserted a need for flows that are capable of transporting all of the sediment load delivered to stream channels in order to maintain channel conveyance capacity and streamside vegetation as an essential part of "securing favorable conditions of water flows" consistent with the Organic Act of 1897 (Gordon 1995). However, the agency has been largely unsuccessful in securing water for purposes of maintaining the channels on National Forest lands. This was due in part to inherent difficulties in defining the range of flows required to support the physical form and function of channel. Channel process and aquatic function are strongly tied to the physical structure of the channel, which, in turn, is controlled by

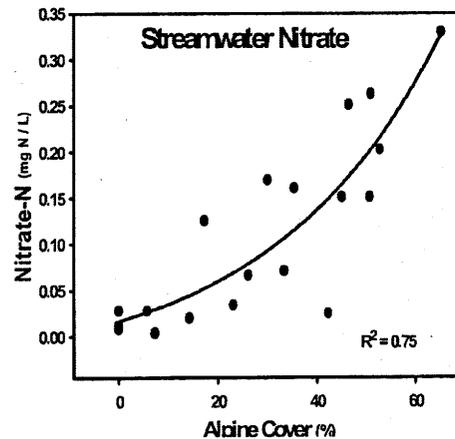


Figure 7. Streamwater nitrate in FEF catchments (Rhoades, unpublished data).

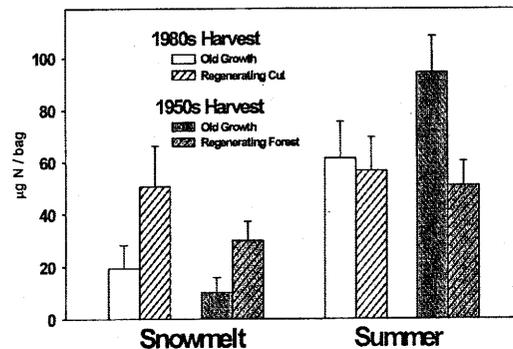


Figure 8. Soil nitrate in 25 and 50 yr old clear cuts at the FEF. Bars are seasonal means and SE for ion exchange resin bags placed in the upper 10 of mineral soil.

topographic, geologic, and vegetative features and land management practices and natural disturbances specific to the watershed. Channel type is largely dependent on the overall patterns of sediment supply and runoff (snowmelt or rainfall-dominated), constrained within local geologic setting (slope and valley characteristics). Channels on western forests typically range from steeper step-pool structures in the upper portions of the watersheds to meandering pool-riffle forms in the flatter valley bottoms (e.g., Montgomery and Buffington 1997). Changes in either sediment supply or stream flow are likely to cause alterations in channel form and unwanted changes in aquatic resources, particularly in flatter, more alluvial segments (Ryan 1997). However, identifying change in channel form due to anthropogenic influences has proven difficult because of the scarcity of unaltered reaches that provide suitable reference conditions (Wohl 2001), or that the magnitude of the impact does not exceed the range of natural variability.

Flow alteration, associated typically with damming and diversion, has had an impact on the channel form and function of many stream systems in Rocky Mountain region. However, the effects of dams and diversions are largely dependent on the nature of the change in flow associated with the operation of the structures and the physical characteristics of the affected channel. For instance, the effects of flow depletion on channels downstream of large storage dams is well documented (e.g., Ligon 1995) and typical responses include decreased channel size, aggradation at tributary junctions, loss of sediment from the system, and encroachment of riparian vegetation. However, it is difficult to extrapolate the results of studies from large dams to low head diversions that are typically found on streams in National Forests. Much of the research on dams has been done on larger, low gradient channels that are more responsive to flow alteration. In these situations, the annual flow remains about the same though the hydrograph is more irregular, peak flows are reduced, and sediment is trapped behind dam. By comparison, streams on National Forest lands tend to be smaller, headwater streams which are presumably less morphologically responsive. Under a typical diversion scenario, the annual flow is reduced but peak flows are less impacted and sediment may be by-passed. As a result, changes in channel form downstream of diversion structures are often subtle or absent. For example, at St. Louis Creek on the Fraser Experimental Forest, where an estimated 40% of the annual streamflow had been diverted off-site for over 40 years, changes in form (smaller channel, vegetation encroachment) were relatively minor and limited to segments in unconfined valley bottoms where the systems is more alluvial in nature (Ryan 1997). The absence of widespread response was attributed in part to the fact that high flows (bankfull and greater), which are responsible for forming the physical channel structure, are still passed downstream on a relatively frequent basis (Figure 9). Hence, it was difficult to demonstrate that flow diversion had caused harm to the physical channel conduit at this and other sites in Colorado. However, with increased demands on water resources from National Forests, there is likely to be an increased call for water during peak flows. Hence, there is greater potential for physical changes in the channel structure at a site like St. Louis Creek – that is, a smaller channel due to sediment accumulation and vegetation encroachment is more likely when peak discharges are further reduced. Hypothetically, a smaller channel is no longer able to pass large flows when they do occur and, as a result, the channel may scour and “blow-out,” resulting in a loss of channel function.

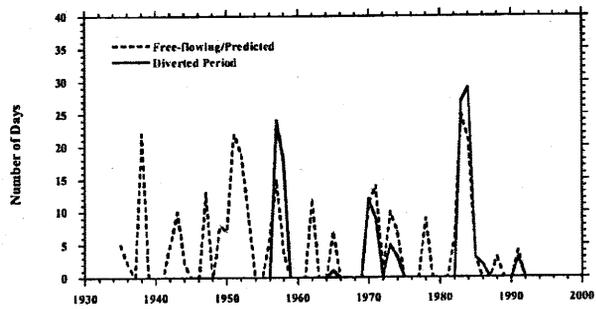


Figure 9. Difference in number of days for which greater than bankfull discharges were observed at St. Louis Creek after diversion began in 1956. In wetter years, the number of days was about the same as predicted had there been no diversion. Sustained bankfull discharges occurred every 1-2 years prior to flow diversion and about once every 5-7 years after diversion was initiated.

Sedimentation issues

Undisturbed forested areas in Colorado and Wyoming typically have low rates of erosion and sedimentation (i.e., Leaf 1970; Patric and others 1984), because much of the precipitation falls as snow, soil infiltration rates are relatively high, and rates of mass wasting are typically low. However, forest management activities have the potential to alter the nature of sediment delivery from hillslopes to channels, thus increasing the rate of sedimentation in channels. In particular, unpaved roads in forested environments present a major source of sediment affecting stream channels (Elliot 2000). Roads have lower infiltration rates and can generate increased runoff over

bare surfaces – factors that can lead to increased flow and more sediment from altered hillslopes reaching channels. The effectiveness of Best Management Practices (BMP's) in reducing the impacts of forestry activities is an area of on-going investigation at the Fraser Experimental Forest and other areas of northern Colorado.

Increases in coarse sediment (typically moved as bedload) may have less influential effects on water quality issues, but a more pervasive effect on the physical structure of the channel. Rates of bedload movement in streams in Colorado and Wyoming are relatively low (Ryan and others 2005) and channels overall are largely stable in form in undisturbed watersheds. However, large disturbances, such as wildfire, can pose substantial threats to water quality and sedimentation in channels downstream from the disturbed areas. Following fire, peak discharges can increase by an order of magnitude or more on severely burned areas and erosion rates can increase by 2-3 orders of magnitude (Moody and Martin 2001). While higher runoff and erosion rates often decline to near background levels within 3-4 years, the impacts of sedimentation on aquatic resources and structures such as reservoirs may be more long-lasting and pervasive. The potential fire risk in the face of continuing drought and increased fuel loads due to insect-caused canopy mortality over much of the western US remains an on-going area of research.

Sediment concentration in streambeds can have a strong and pervasive effect on aquatic organisms. However, the linkages between watershed disturbances and effects on instream habitat and fish in natural settings are poorly understood. The question as to how much sediment is too much in streams remains unanswered. Moreover, increases in sediment supply must often be quite large in order to be detectable outside of the range of natural variability. For instance, at Fraser Experimental Forest, there was little detectable increase in export associated with additional sediment derived from forestry activities in the Fool Creek watershed study (Troendle 1983). Instead, measured increases in sediment yield were attributed to additional flow generated from the removal of timber rather than substantial increases in hillslope sediment production. Repeat surveys using historical archive data from channels downstream of former and planned harvest units are in progress at FEF to assess the influence of chronic and pulse disturbances on in-stream sedimentation and channel form.

RIPARIAN ECOSYSTEMS

Riparian areas occupy only 0.5 to 2.0% of the landscape in the western U.S., yet they are disproportionately important for maintenance of water quality and quantity (water storage and aquifer recharge), habitat for aquatic and terrestrial biota, sediment retention, stream bank building and maintenance, and provision of services of economic and social value, such as livestock grazing and recreation (Gregory and others 1991, Naiman and Decamps 1997, Naiman and others 2005). Because stream-riparian corridors are located at the lowest point within drainage basins, they can act as integrators of entire watersheds and may be particularly vulnerable to effects of land use conducted upslope and upstream. In the past, undesirable changes in riparian areas have resulted from unsustainable management or the failure to recognize linkages among streams, riparian areas, and uplands. Ongoing issues surrounding the management of riparian areas in the western U.S. include the impacts of forest harvest, livestock grazing, road construction, inadequate road maintenance, and recreation on the structure and function of riparian ecosystems (NCASI 2005). More recently, increasing interest has focused on the influence of natural disturbances such as wildland fire and insect outbreaks, altered hydrologic regimes (dams and diversions), and fuel reduction treatments on valued riparian characteristics and functions.

On National Forest lands, management of riparian areas can generally be defined as custodial (Gregory 1997), and frequently includes establishment of buffers and implementation of best management practices (BMPs) (Belt and others 1992, Gregory 1997, Mosley and others 1997). Although BMPs and the establishment of riparian buffers have mitigated the effects of forest harvest activities on stream water temperature and quality, current BMPs may not be effective in protecting all valued riparian functions, particularly in watersheds that have undergone extensive anthropogenic or natural disturbance, such as severe wildfire or insect infestation. In harvested watersheds, riparian buffers may be susceptible to blowdown; where fires have been suppressed, they may contain unnaturally high fuel loads (Dwire and Kauffman 2003). Another difficulty in the management of many riparian areas is the definition of reference or target conditions. Current attributes and condition of riparian areas reflect the historically recent (approximately 100-200 years) physical conditions of the landscape, as well as land management activities (NCASI 2005). Although the lingering effects of land management prior to the riparian protection may influence the structure, function, and composition of riparian areas for decades to centuries (Young and others 1994), historical legacies for most watersheds are largely unknown. Attention to ecological context within drainage basin and the larger landscape is critical for effective management of riparian areas, as well as the connectivity between upslope and upstream management and condition of streams and riparian areas.

Despite several decades of focused research, riparian areas continue to be frontiers for the study of land/water interactions, ecosystem processes, impacts of watershed and landscape management, and the influence of natural disturbance on stream-riparian processes. The Fraser Experimental Forest (FEF) is well-positioned to contribute to the advancement of riparian research and improved management in the central Rocky Mountain region. Existing information on the management history, hydrologic and sediment regimes, and water chemistry for study basins within the FEF provide a strong baseline for examining the role of natural disturbance, such as large scale canopy mortality, and management activities, such as salvage logging, prescribed burning, and other fuel reduction treatments on subalpine riparian ecosystems. Stream-riparian corridors are dynamic, and the initiation of long-term studies allows for investigation of continuous change, including successional processes and responses to natural disturbance and management treatments. Past and ongoing research on stream-riparian ecosystems at FEF includes evaluation of BMP effectiveness, impacts of flow diversion on sediment transport, and riparian biota and processes, role of hillslope seeps and springs on streamwater, and effects of large-scale natural disturbance (insect-caused canopy mortality) on large wood dynamics and organic matter dynamics in headwater streams.

Diversion of water for municipal use and agricultural irrigation is common in many western watersheds administered by the USDA Forest Service. Although most of the diversions and dams in the western U.S. are built in mountainous areas, little is known about the influence of small dams and water withdrawals on aquatic and riparian biota and ecosystem processes along low order streams. As noted above, the influence of flow alteration on sediment transport and channel morphology depends on the timing of water diversion. Since most sediment moves during periods of high flow, stream channel condition may not be strongly impacted by water diversion occurring later in the season. Studies have been initiated to examine the impacts of flow diversion on riparian and aquatic biota, and the role of hillslope and groundwater inputs on spatial recovery gradients. Concurrent work is being conducted in the Medicine Bow National Forest, Wyoming. Results will assist in determining the amount of water needed during different seasons to maintain riparian and aquatic biota, and contribute to proposed water management strategies.

Springs and seeps are common throughout the Rocky Mountains. In many glaciated watersheds, they are important sources of stream water, provide critical riparian and aquatic habitat, and can exert strong controls on streamwater chemistry. Hydrologic and chemical characteristics of springs may be influenced by forest harvest and other management activities, climate change, and natural disturbance. Ongoing research in the Fool Creek watershed, FEF, is addressing the influence of past forest harvest, elevation, and landscape position (aspect, distance from stream) on spring water temperature and chemistry in relation to stream water characteristics (Dwire and others 2006). This research compliments concurrent studies on slope and riparian wetlands to improve understanding of hydrologic connectivity within basins and drivers of surface and subsurface biogeochemistry. Current research efforts also provide a baseline for detecting changes in springs and streams due to insect-caused canopy mortality, planned fuel reduction treatments, and climate change.

In forested landscapes, riparian areas are important sources of large wood for streams and floodplains. However, riparian forest stands are frequently patchy, and variation from different sources can lead to spatial variability in large wood distribution that is often not recognized in management prescriptions for a given amount of large wood per unit length of stream (Young and others 2006). Chronic inputs of large wood to stream channels occur as a result of bank cutting, windthrow, and mortality of individual trees from adjacent riparian areas (McDade and others 1990, Bragg and Kershner 2004). Large pulses of wood may originate from near-channel sources following fire, windthrow, or insect infestations, or be transported from distant sites by debris torrents, avalanches, or landslides (Bilby and Bisson 1998, Bragg 2000, Benda and others 2003). At FEF, considerable stand-level data exists for basins that are currently being impacted by severe infestation of mountain pine beetle. Studies are underway that examine chronic and pulse inputs and sources of large wood — including source (upland, riparian) and process (avalanche, insect-caused, windthrow) — relative to upland and riparian stand structure. Results will provide increased understanding of the role of natural disturbance (avalanches and insect infestations) in the delivery of large wood to headwater streams, and contribute to the development of in-stream targets for large wood volumes in the Rocky Mountain region.

FISHERIES

Among the issues that influence water management in the western U.S. is the status of the aquatic biota, particularly fish. The tenuous condition of many populations of Pacific salmon is well known, but perhaps less recognized is that all of the species of salmonines in the interior West—the 14 subspecies of cutthroat trout, the Apache trout and Gila trout, golden trout, bull trout, and some forms of redband trout—have either been petitioned for federal listing under

the Endangered Species Act, are currently listed as threatened or endangered, or are extinct (Young and Harig 2001). The majority of populations of these taxa are found in streams on federal lands, and the strongest populations tend to be found in streams draining basins afforded some additional protection, such as wilderness designation or status as a national park (Shepard and others 2005; Hirsch and others 2006). However, such designation is not sufficient to secure these populations; there are many examples of the extirpation of individual populations of these species in pristine habitats (Young 1995). This includes the Fraser Experimental Forest in Colorado, where indigenous Colorado River cutthroat trout have been lost from all streams in this area (Young and others 1996).

The threats facing these species are intimately linked to how water is managed in the West. As a consequence of water diversions for agricultural or municipal use, or of culverts that are impassable to upstream fish movement, many populations of these salmonids have been isolated in high-elevation stream segments that tend to be cold and unproductive. This isolation renders populations susceptible to environmental events, such as drought or fire-related debris torrents, that lead to local extinctions (Brown and others 2001; Morita and Yamamoto 2002) and prevent recolonization of these segments by mobile fish originating elsewhere in the basin. This has led to studies of how habitat quality and quantity affect population size, particularly of Colorado River cutthroat trout and greenback cutthroat trout in the Central Rocky Mountains (Young and others 2005). Ironically, these isolated segments also represent some of the best refuges for native salmonids, because barriers at their downstream ends prevent the invasion of nonnative species, such as brook, brown, or rainbow trout, that would otherwise replace or hybridize with the native taxa (Dunham and others 2002). Recognizing this dilemma and examining the consequences of habitat isolation or nonnative fish invasion are of critical importance to fish managers throughout the inland West (Fausch and others, in press).

The role of fire in altering fish habitat, population persistence, and nonnative fish invasion is also of great concern. Even today, the effect of fire on stream fish populations remains poorly understood, but isolated studies from different parts of the West are beginning to fill this gap. Occasionally, high-severity fires generate sufficient heat to lead to immediate fish kills (Rinne 1996), but more common are post-fire debris torrents and blackwater events resulting from summer thunderstorms that eliminate fish populations (Bozek and Young 1994; Brown and others 2001). Historically, such populations would have been immediately refounded by migrating fish that survived in local refugia or avoided these events by occupying other portions of a watershed, but as noted above, human-built barriers often make this impossible (Rieman and others 2003). A related concern is that habitat changes resulting from fire—increased water temperatures, greater sediment loads, and reduced channel stability—are often associated with conditions favoring nonnative trout. However, because western trout species evolved with such disturbances, they may be more resilient than nonnative species introduced from areas where fire-related habitat changes are uncommon (Dunham and others 2003). Preliminary results from some Montana basins suggest that native cutthroat trout recover more rapidly after fire than do introduced brook trout (Sestrich 2005).

Many questions remain to be answered with regard to native fish species and water management in the Rocky Mountain West. Continued research on fish response to forest disturbances such as fire, large-scale insect-related tree mortality, and drought is a priority. Also, formal systematic conservation plans (Margules and Pressey 2000) for restoration of rare fish in the West have rarely been developed, and it may be possible to construct such a plan for the stream network on the Fraser Experimental Forest. Similarly, existing diversions in this area may present opportunities to measure the ability of nonnative fish to move through structures, with the objective of engineering more effective barriers to nonnative fish passage.

A NATURAL DISTURBANCE RESEARCH OPPORTUNITY

Populations of mountain pine bark beetle (MPB) and other forest insects have increased rapidly in western North America during the past decade. The impacts of MPB-induced mortality on watershed processes and aquatic resource conditions in Rocky Mountain forest ecosystems are poorly understood. Widespread MPB-related forest mortality has created public anxiety over human safety and property loss, associated with perceptions of heightened wildfire risk. Such concern led to enactment of the Healthy Forest Restoration Act of 2003, legislation that gives federal land managers the administrative tools to address hazardous fuel loads and other forest health issues rapidly. Unfortunately, federal resource managers working in much of the Rocky Mountain West currently lack adequate information to evaluate the influence of fuel reduction treatments on forest productivity, water quality, streamflow and other watershed resources.

FEF is located near the epicenter of a large MPB outbreak that is affecting lodgepole pine in subalpine forests of central Colorado. Significant MPB-related lodgepole pine mortality was first observed at FEF in 2003; by 2005 most mature pine stands had become infested. This outbreak places FEF in a unique position to address critical gaps in the understanding of the watershed consequences and management responses to the outbreak. FEF researchers are focusing on two fundamental questions regarding the effects of bark beetles on subalpine forest watersheds:

- 1) How does the present bark beetle outbreak influence watershed processes and forest conditions in managed and unmanaged basins?
- 2) What are the consequences of forest management manipulations associated with the insect outbreaks and other forest health management activities?

To evaluate biogeochemical and hydrologic changes we will utilize more than fifty years of snowpack and streamflow measurements and two decades of precipitation and streamwater chemistry. Our assessment of management activities will:

- 1) quantify the influence of salvage operations on nutrient, carbon, sediment and large wood retention within riparian buffers and validate the effectiveness of watershed best management practices for protecting water quality and aquatic resources;
- 2) evaluate how mechanical fuel reduction treatments (chipping and mastication) and post-harvest site preparation impact tree seedling establishment and growth, plant nutrient and moisture relations, and biogeochemical and hydrologic processes;
- 3) assess the impacts of forest road construction and retirement on hillslope hydrology and nutrient and sediment fluxes.

Greater understanding of these management practices will assist national forest managers protect and sustain upland, riparian and aquatic resources.

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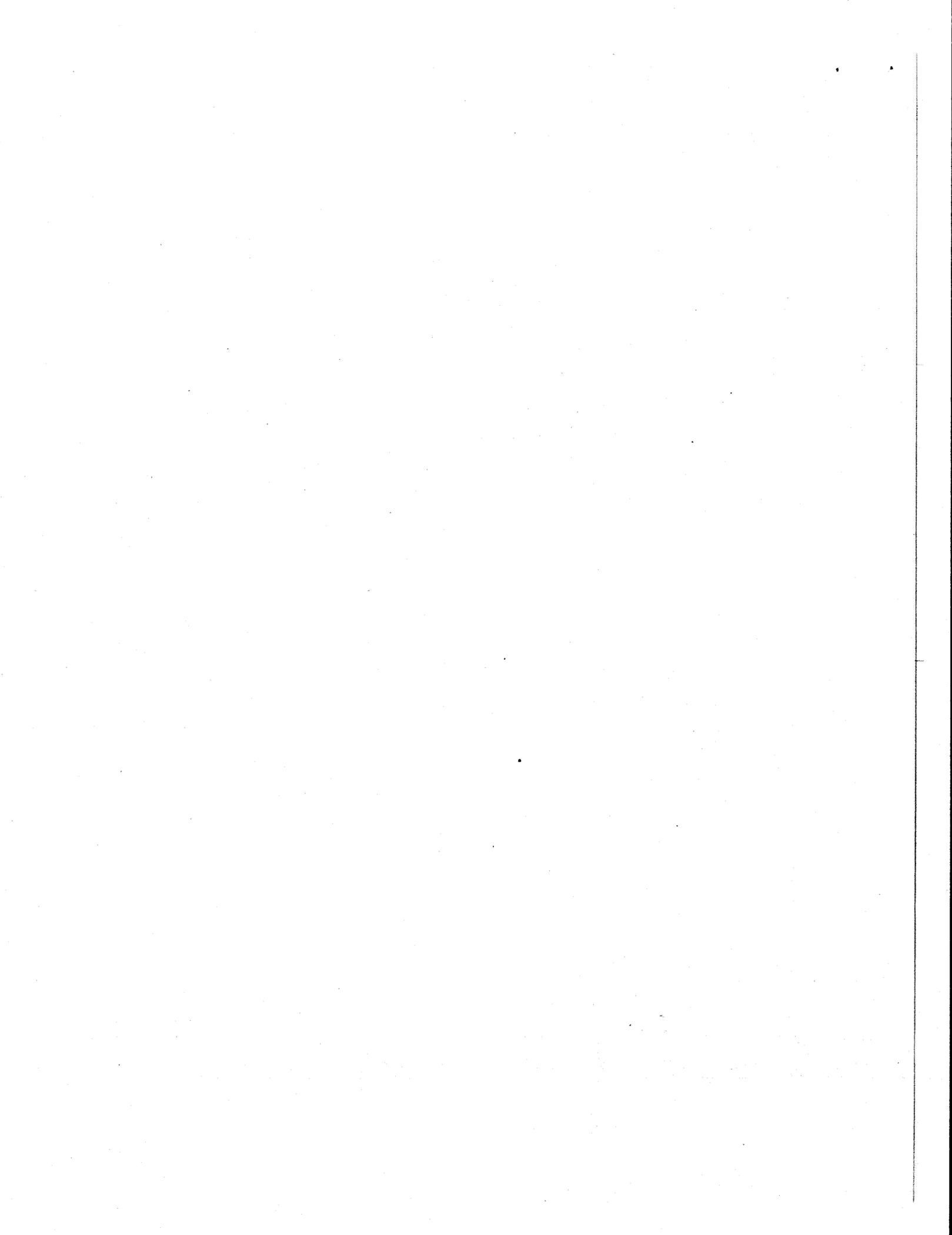
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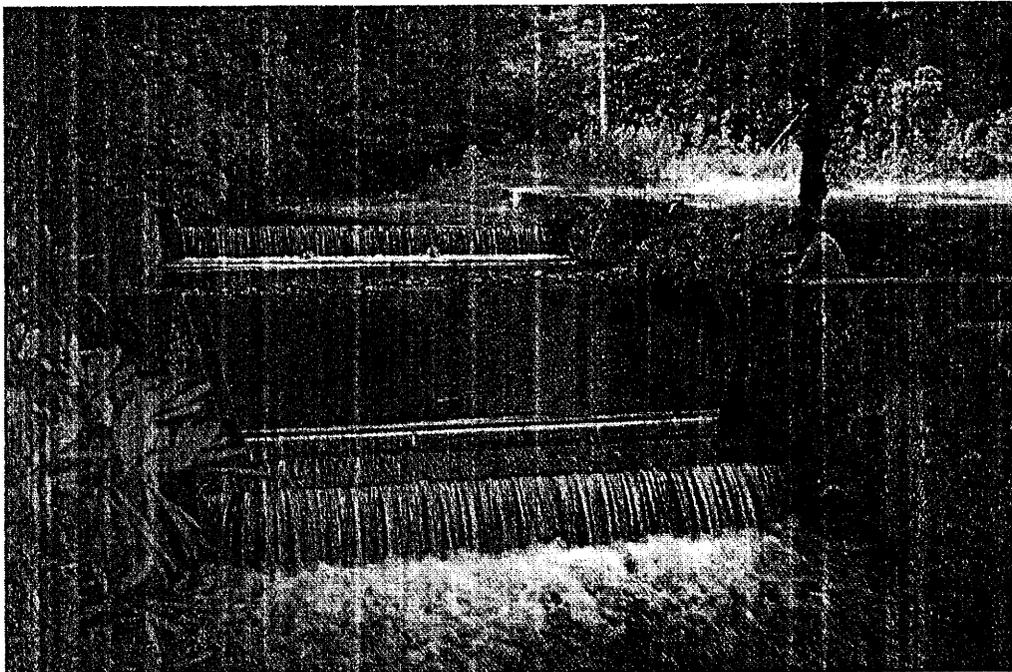
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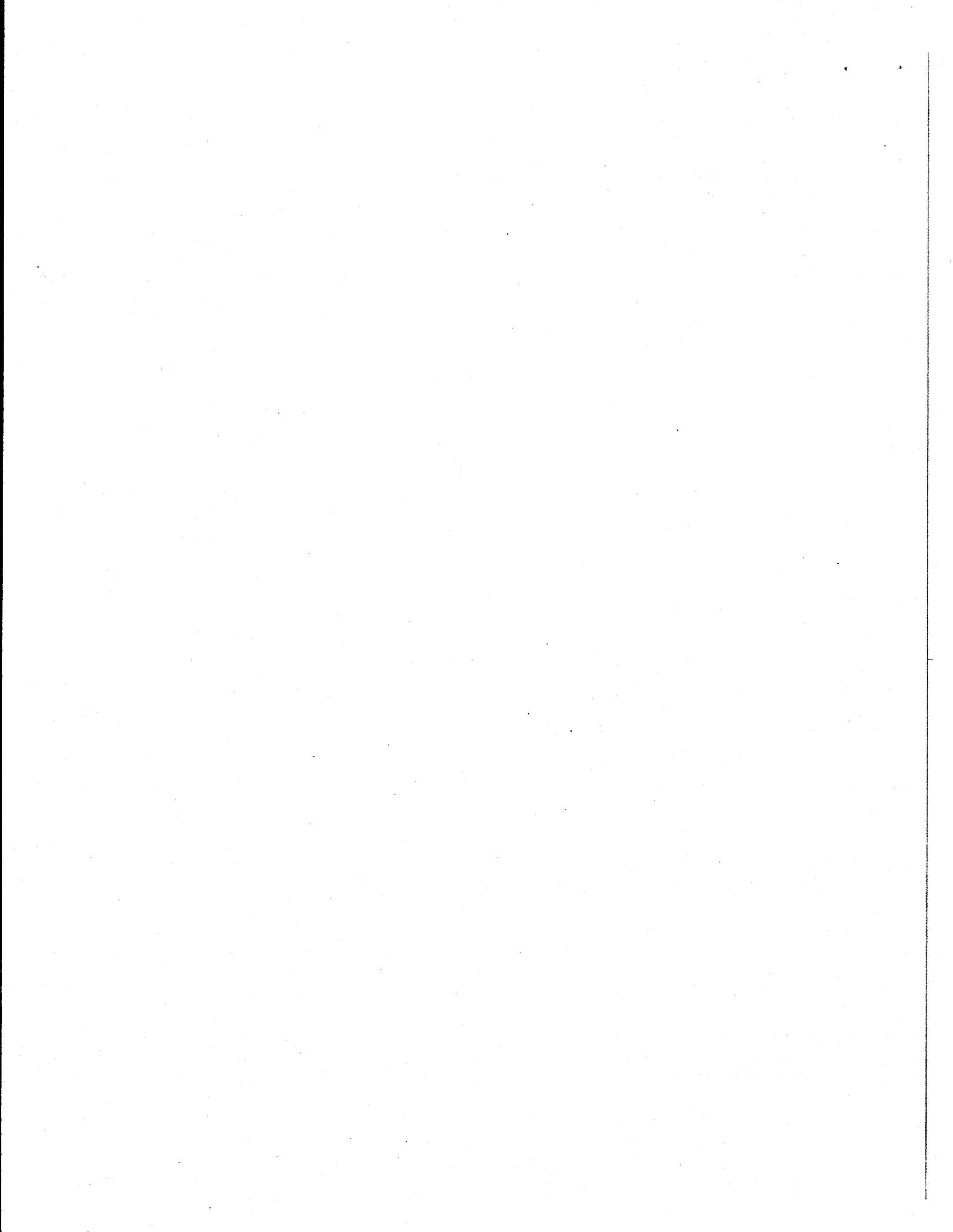


**Second Interagency Conference on
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**Hydrologic Data Collection
and Modeling**





AN ARS DATA SYSTEM FOR ASSESSING CONSERVATION PRACTICES IN WATERSHED STUDY

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Abstract--To support ARS's recently established Conservation Effects Assessment Project (CEAP) in assessing USDA conservation programs and practices on soil and water quality, a team located at several ARS locations is developing a web-based watershed data system named STEWARDS. The data system consists four components: databases (metadata, measured data, and GIS layers), servers, clients (users) and ARS watershed sites (data sources). The system requirements document has been completed and system design is underway. The fully implemented data system will organize and document soil, water, climate, land-management, and socio-economic data from twelve benchmark ARS research watersheds and provide data search, visualization and downloads to users. A case study using a prototype is presented here to illustrate the functionalities of STEWARDS in facilitating water quality data visualization.

INTRODUCTION

To effectively retrieve embedded information/knowledge from watershed data and apply to national scale hydrological research, a web-based and user friendly data system is needed. This is especially true for the newly implemented USDA-ARS Conservation Effects Assessment Project (CEAP) which was designed to quantify environmental effects of USDA conservation programs and practices (Mausbach and Dedrick 2004). Using the sampling and modeling approach, the CEAP studies require a variety of data that describe hydrology, soils, climate, topography, management practices, and land use to assess the impacts of conservation practices on soil and water quality, and ecological/environmental health. The Soil Water Assessment Tool (SWAT) and Annualized Agricultural Non-Point Source (AnnAGNPS) models are the primary models to be used in the modeling effort.

Although the USDA and the Agricultural Research Service (ARS) have conducted watershed research since early in the 20th century, the data have been managed to address location-specific research needs and are managed and disseminated independently from each research location. Such practices greatly reduce the accessibility and utility of these data for policy-relevant, multi-site analyses. Development of a centralized web-based data system would simultaneously increase the longevity and usage of the ARS watershed data and meet the data needs in CEAP. To meet these concerns, an ARS watershed data system named STEWARDS (Sustaining the Earth's Watersheds, Agricultural Research Data System) has been initiated (Steiner and others 2003).

Existing Data Systems

In the last two decades, progress in the study of information (informatics) and its manipulation via computer-based tools has stimulated the development of data systems in geology (geoinformatics), hydrology (hydroinformatics), and ecology (ecoinformatics). For examples GEON Portal (<http://www.geongrid.org/>) is based on a service-oriented architecture (SOA) with support for "intelligent" search, semantic data integration, visualization of 4D scientific datasets, and access to high performance computing platforms for data analysis and model execution. Likewise, Hydrologic Information System (HIS) (<http://www.cuahsi.org/>), which is under development by the Consortium of Universities for the Advancement of Hydrologic Science, Inc. (CUAHSI) (<http://geo.sdsc.edu/cuahsi/>) is a SOA based system. It relies on a collection of loosely coupled self-contained services that communicate with each other and will integrate hydrologic data, tools and simulation models to support hydrologic science, education and practice. The CLIMDB/HYDRODB data system sponsored by the Long Term Ecological Research (LTER) Network is a centralized server which provides open access to long-term meteorological and streamflow records from a collection of research sites. The LTER Network has established a network-wide information system to facilitate data exchange and integration for long-term ecological research (<http://www.lternet.edu/informatics>). The USGS' NWISWeb data system (<http://waterdata.usgs.gov/nwis>) provides access to water-resources data collected in the US, and also provides visualization tools for viewing surface and groundwater data. What these data systems have in common is that they were created for general use in providing data storage, data access, visualization, with or without analysis/modeling tools, and non-specific data downloads. The uniqueness of STEWARDS is that this data system will not only be built for general data access but also will be designed to support CEAP watershed studies.

Objectives

The objectives of this paper are

- 1) to outline STEWARDS' system architecture and functionality,
- 2) to present the approach of STEWARDS/prototype development with an emphasis on system design,
- 3) to illustrate the system functionality and usage of STEWARDS by using a prototype to explore the water quality data in Oklahoma.

FEATURES of STEWARDS

STEWARDS is a web-based, client-server data system designed to organize, document, manipulate, and compile climate, water, soil, land-management, and socio-economic data derived from ARS research watersheds. The server processes requests/queries from the client (user), and enables data visualization or data downloads. The client issues a queries or data request interactively through a user interface.

The architecture of STEWARDS consists of four major components (Fig. 1):

1. Server: a centralized site running a database management system (DBMS), which consists of Web/SQL/ArcIMS servers and application software,
2. Data: includes metadata, measurement data, and images/GIS layers etc.,
3. Clients: users who search, access, visualize and download data from remote sites, and
4. Research Watershed Sites: data sources for STEWARDS.

Architecture of STEWARDS

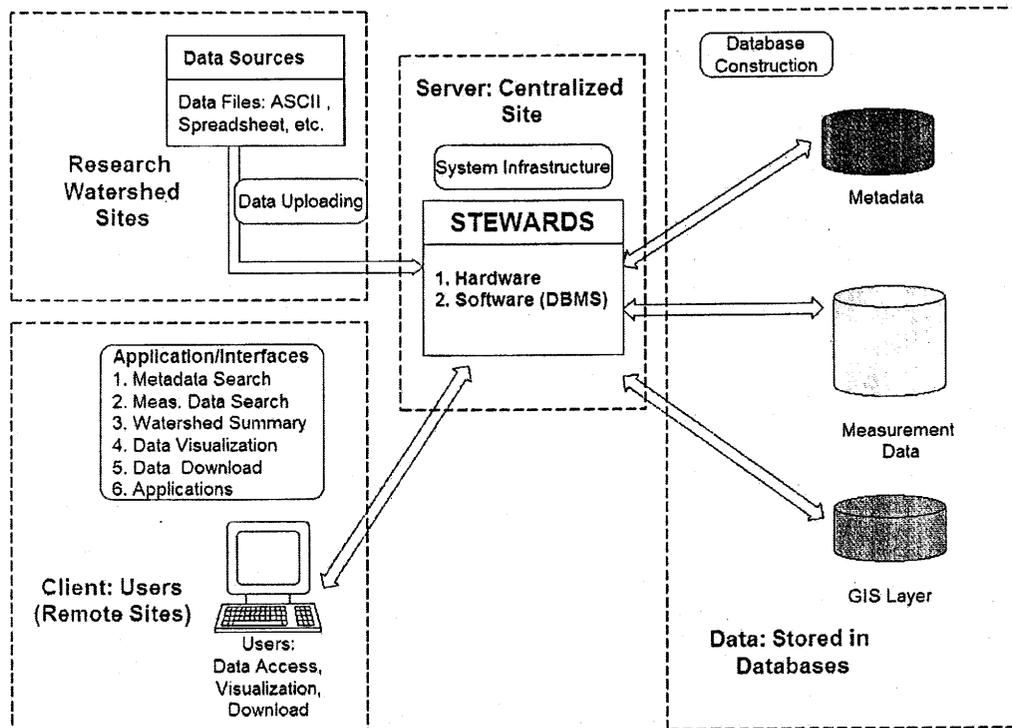


Figure 1--Architecture of the ARS STEWARDS data system. This diagram can be taken as a conceptual data system. There are four identified system design components. Each component is bounded a dash line. The contents in the rectangles with the corners clipped off represent the tasks of each design component.

Status of STEWARDS Development
 Development of STEWARDS involves a team of ARS researchers from numerous locations. Steiner and others (2005) documented the system lay-out/requirements and the approach of the system implementation for STEWARDS (Fig. 1). The implementation of the data system consists of five tasks. The system requirements document (task 1), which details the overall system architecture, features and interface specifications, has been peer-reviewed internally and is in the external review process. System design (task 2) is nearly completed and will be described in details below. Parallel to these ongoing tasks, a prototype data system (task 2a) has been developed in El Reno, Oklahoma, using data from the Fort Cobb Reservoir Experimental Watershed (FCREW) and Little Washita River Experimental Watershed (LWREW). Metadata development (task 1a) will be implemented during the system design and system construction (task 3) phases followed with system testing (task 4) and deployment (task 5).

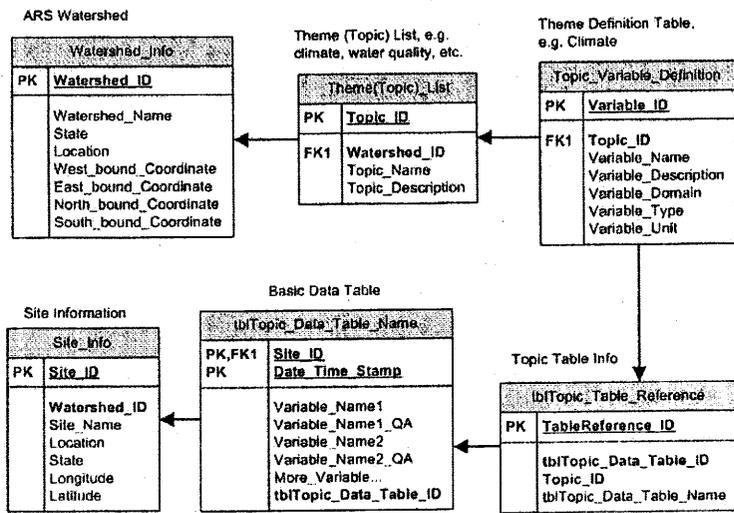


Figure 2--ER diagram represents the Type 2 data which time series data are collected at the same time.

Approach in System Design

The purpose of system design is to translate the system requirements into more technical specifications. System design begins with building a conceptual data system and then a logical data system and ends by producing a physical data system design. Based on the system requirements (system architecture), a conceptual data system (Fig. 1) is developed and four design tasks/components are identified. These four components are system infrastructure (hardware/software), database construction, user interface/application development, and data uploading/translation tool development, and are represented by four blocks in Fig. 1.

During the logical system development phase, the four system components/tasks and their relationships are analyzed and evaluated through various levels. For example, Entity-Relationship (ER) diagrams (Figs. 2 and 3), which provide pictorial representation of the

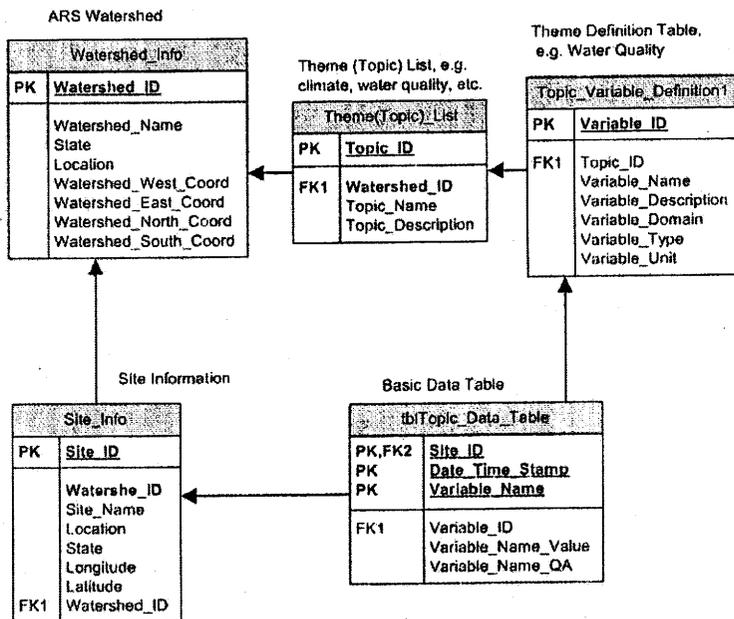


Figure 3--ER diagram represents for Type 3 data which time series data are usually more sparse than Type 2 data with an identification (ID) field that explains what the parameter field contains.

logical structure of the database design, are developed and evaluated. Similarly, flow charts (Fig. 4), in which user interfaces are outlined, are used to examine and validate the logic of data/task flow. The resulting products of system design will be: 1) a physical data system design to be used for system construction, 2) a work plan which implements the action of physical design with the considerations of hardware, software tools and skill requirements, and other resources, and 3) formation of a system development team to proceed to the construction phase. The system design specified the following.

1. **System Infrastructure:** A computer with a Pentium 4 CPU (3.0GHZ or greater) with storage devices of at least 2.0 tera bytes capacity; Microsoft Windows Server 2005; Microsoft Internet Information Services (IIS); and Microsoft SQL (Structured Query Language) 2005 server. Other software include Microsoft Visual Studio .NET package which is used to develop user interfaces and other application tools.
2. **Database Construction:** Metadata sets will conform to the requirements of the Federal Geographic Data Committee (FGDC) standards. The data managers of the local ARS watershed sites will be responsible for constructing the metadata sets. FGDC's metadata structure and guidelines will be provided to the watershed locations to assist them in developing the required metadata sets. For measurement data sets, a data schema (i.e. a list of data elements) and Entity-Relationship (ER) diagrams will be developed, validated, and tested. There are three types of empirical/measurement data in STEWARDS: 1) GIS layers (Type 1 data) which are data mostly generated by using ESRI's ArcGIS or ArcView GIS, 2) time series data (Type 2 data) with columns for different data collected at the same time, and 3) time series data (Type 3 data), usually more sparse than 2) with an identification (ID) field that explains what the parameter field contains. The ER diagram of Type 2 data is presented in Fig. 2 while the ER diagram for Type 3 data is presented in Fig. 3. Note that the generalized data entities and their ER diagrams in Figs. 2 and 3 cover all the possible data objects and their relationships in the CEAP watershed studies.
3. **User Interfaces/Applications:** Consist of three major interface components (Fig. 4): Three components are overview/site summary access, system/data searches and data access. Microsoft Visual Studio .NET package is used for developing STEWARDS interfaces/tools because under .NET environment, the HTML and scripts can be separated and script maintenance is easier compared to other interface developing package such as PHP in which the scripts are embedded in the HTML. Additionally, Microsoft Visual Studio .NET provides a rich amount of library routines and there are many online scripts/examples for .NET available, which are useful for the development of STEWARDS interfaces.
4. **Data Uploading/Translation:** A generic data translation module has been developed to identify data fields, retrieve the data values on selected fields, filter out unnecessary data records (e.g. blank, comment lines etc.) and data fields based on specific data selection criteria, perform data transformation/unit conversion, and build SQL scripts for creating a database.

STEWARDS Data System

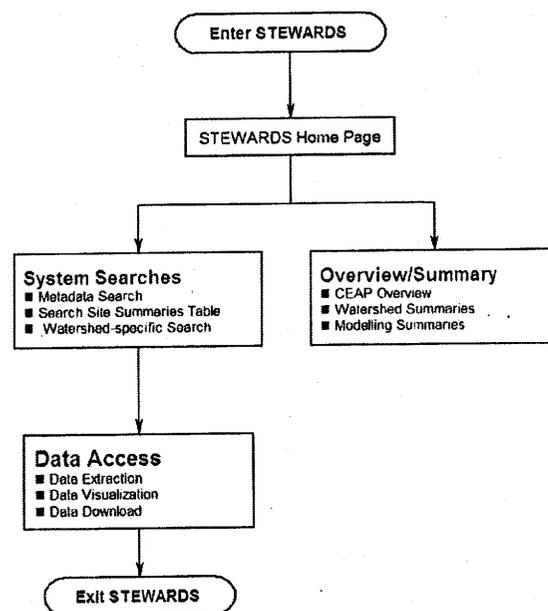


Figure 4--User interfaces in STEWARDS data system. The system menus show data accessing flow.

Time Series Data Structure

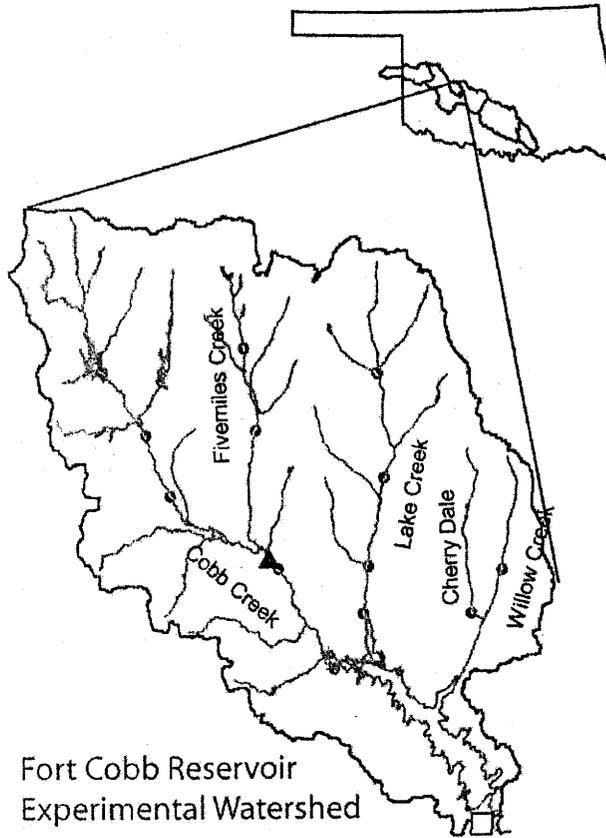
One of the major concerns on the data structure of time series data is how to create temporal relational databases. Classical database technology can handle time instants (times at which events occur) reasonably well; however, time intervals (periods of time during which states persist, e.g. break point data for precipitation) are not accommodated well (Date, 2000). In STEWARDS, the time in break point data can be stored by a beginning timestamp and an ending timestamp reasonably well.

Another issue on time series data is the time representation. That is, a composite-field representation, which treats year, month, day, hour, and minutes as individual fields, versus a "datetime" representation, in which year, month, day, hour, and minute are stored as a whole. The datetime type is used because it takes less disk space less overhead on disk space; data query using one single "datetime" field is generally simpler than using composite-field representation; and accessing time using "datetime" data type in query or web scripts is less. For example, the SQL access time to the composite-field representation takes about 40% longer than "datetime" type.

Experiences gained from developing a prototype

One of the merits of developing a prototype is to test the feasibility of potential technical solutions for the final data system. It was found that developing a generic module (e.g. the data translation tool, a general-purpose graphics routine) which complies with industrial standards is beneficial in terms of software development cost. "Generic" means reusable and general purpose. A generic module allows repeated use of the cripts. The web pages produced by a generic module have similar functionality and appearance throughout the system, which will foster intuitive use of other system modules once the user gains familiarity with one system module. Compliance with the industrial standards means that users familiar with web browsing can enter and browse STEWARDS web pages with minimal

difficulty. Additionally, the system software can be better maintained and the system usability will increase. Furthermore, following the industrial standards in error handling, naming conventions and programming styles would make the system to robust, easy-to-use and easy-to-maintain.



Fort Cobb Reservoir
Experimental Watershed

Figure 5--Site map of Fort Cobb Reservoir Experimental Watershed in Oklahoma. The solid circles are water quality sampling sites. The solid triangle represents the location of USGS stream gauge (7325800) at Cobb Creek near Eakly, OK. The open rectangle represents the Oklahoma Mesonet climate station at Fort Cobb.

CEAP CASE STUDY USING THE STEWARDS PROTOTYPE

STEWARDS allows the users to examine the spatial and temporal trends of measured data , to examine the relationships among the measured parameters, and retrieve/download the data for preparing input data files for the SWAT and AnnAGNPS models. The following case study is used to illustrate these STEWARDS functionalities using the prototype and data from the Fort Cobb Reservoir Experimental Watershed (Fig. 5) in the Upper Washita River hydrologic unit of Oklahoma.

Water Quality Data Time Series Visualization
Biweekly water quality samples have been taken at FCREW since December 1, 2004. Water quality parameters measured are pH, conductivity, dissolved oxygen, turbidity, total phosphorus, bio-available phosphorus, sediment concentration, nitrate and total nitrogen. Fig. 6 shows the time changes of total-P, sediment concentration, daily averaged discharge, and daily total rainfall at FCREW. The sharp increases of total-P concentration and sediment concentration observed on Jun 15, 2005 were associated with a peak discharge immediately after a big storm. A high total-P concentration is usually associated with a high sediment amount during a big storm event which results in a high discharge in general. For example, a significant linear relationship (with $R=0.67$) was observed between sediment concentration and total-P was observed for the water samples with higher sediment concentrations (> 200 mg/L) for FCREW data.

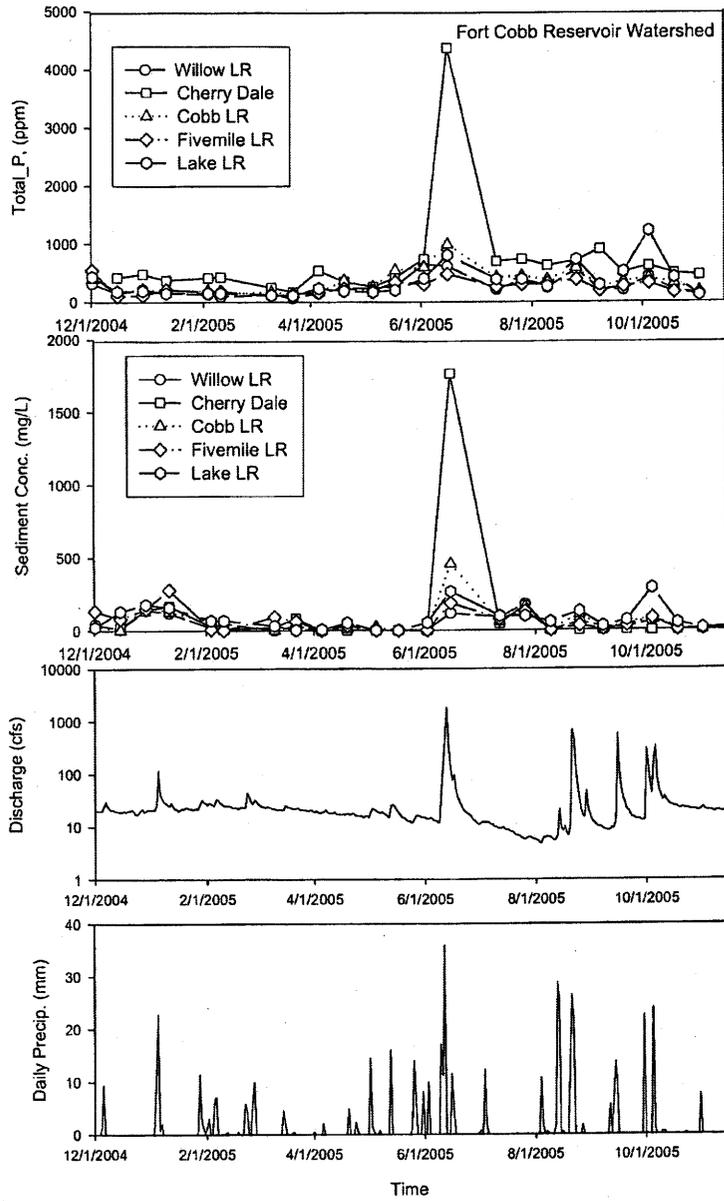


Figure 6--Time changes of total P (a), sediment concentration (b), daily average discharge (c), and daily total rainfall (d) at FCREW. The sharp peaks in sediment concentration and in total-P concentration were observed on 6/15/2005 right after the storm event of 6/9-6/13, 2005.

CONCLUSION

The example demonstrated using a prototype presented here illustrate how STEWARDS can provide a useful means to help users to see data clearly, analyze relationship among the water quality parameters STEWARDS, when completed, will provide data to scientists, resource managers, and public to support natural resource decision-making and to assess conservation practices and other hydrologic analyses.

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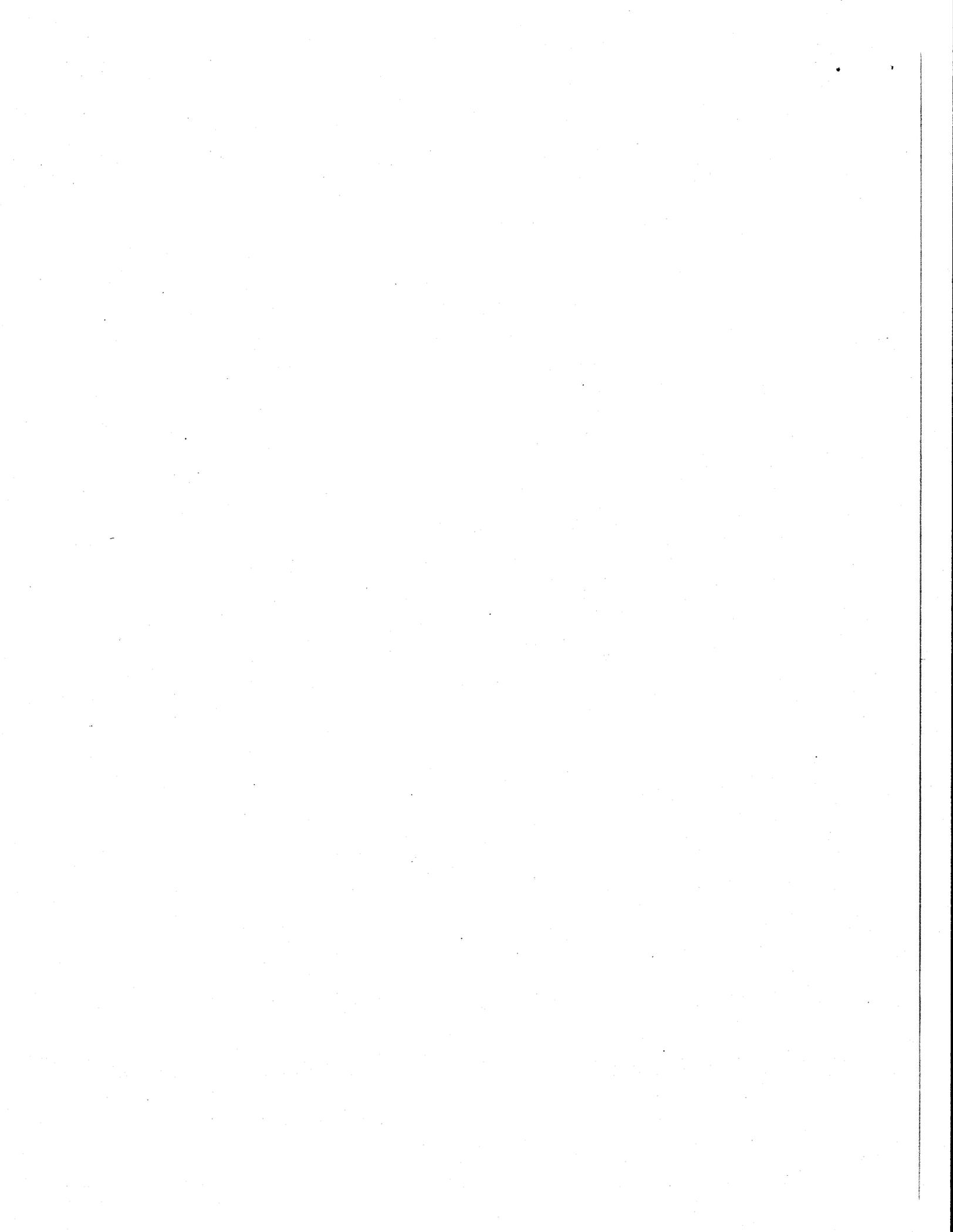
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HYDROLOGIC MODEL UNCERTAINTY ASSOCIATED WITH SIMULATING FUTURE LAND-COVER/USE SCENARIOS: A RETROSPECTIVE ANALYSIS

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Abstract--GIS-based hydrologic modeling offers a convenient means of assessing the impacts associated with land-cover/use change for environmental planning efforts. Alternative future scenarios can be used as input to hydrologic models and compared with existing conditions to evaluate potential environmental impacts as part of this process. Model error, however, can be significant and potentially compounded when projecting future land-cover/use change and management conditions. To address this problem we have utilized repeat observations of land cover/use as a proxy for projected future conditions. A systematic analysis of model efficiency during simulations based on observed land-cover/use change is used to quantify error associated with simulations for a series of known "future" landscape conditions over a 24-year period. Calibrated and uncalibrated assessments of relative change over different lengths of time are also presented to determine the types of information that can reliably be used in planning efforts for which calibration is not possible.

INTRODUCTION

Integrated, regional planning efforts have begun to use an innovative GIS-based simulation modeling strategy that considers the demographic, economic, physical, and environmental processes of an area and projects the consequences to that area of various land-use planning and management decisions (e.g. Steinitz and others 2003). The results of such projections, and the approach itself, are known as "alternative futures", and are being used with increasing frequency to favorably guide efforts to shape future landscape change. The results of an alternative futures exercise are represented in terms of a land-cover/use grid that embodies the issues, user groups, and management choices associated with a particular option. This map serves as the primary means of relating the consequences of management alternatives to biophysical and socioeconomic systems.

Geographic Information Systems (GIS) have been widely used to facilitate the parameterization of hydrologic models and visualization of model results through the development of GIS-based model interfaces (e.g. Ogden and others 2001, Miller and others In press). Land-cover/use grids are a principal input to watershed hydrologic models and the primary means of incorporating anthropogenic impacts into distributed hydrologic assessments. Alternative future land-cover/use grids thus provide a means of incorporating projected growth and development into hydrologic assessments for the purpose of exploring potential environmental impacts associated with future scenarios (e.g. Kepner and others 2004). This technique holds great promise as a means of providing decision support for planning efforts, but a significant concern is the lack of available information on the uncertainty and appropriate use of physically based hydrologic models in a forecasting mode.

The Automated Geospatial Watershed Assessment (AGWA) tool is a GIS-based interface for watershed modeling and assessment that has been developed jointly by the U.S. EPA Office of Research and Development, USDA Agricultural Research Service, and University of Arizona. AGWA provides the functionality to conduct all phases of a watershed assessment for two widely used watershed hydrologic models: the Soil & Water Assessment Tool (SWAT; Arnold and others 1994); and the Kinematic Runoff and Erosion Model (KINEROS2; Smith and others 1995, Semmens and others In press). SWAT is a continuous simulation model for use in large (river-basin scale) watersheds. KINEROS2 is an event-driven model designed for watersheds characterized predominantly by overland flow. The AGWA tool combines these models in an intuitive interface for performing multi-scale change assessment (Hernandez and others 2005). Data requirements include elevation, land cover, soils, and precipitation data, all of which are available at no cost over the Internet. AGWA is available at no cost via the Internet as a modular, open-source suite of programs (www.tucson.ars.ag.gov/agwa or www.epa.gov/nerlesd1/land-sci/agwa/).

The use of watershed hydrologic models to forecast impacts associated with land-cover/use change is fraught with uncertainty. Even in data rich locations it is not possible to calibrate a model to future conditions, and it is thus beneficial to have some idea of how model performance varies with time from a baseline, calibrated period. In

locations lacking sufficient data for initial model calibration, two additional pieces of information are needed to determine the reliability of information derived from distributed hydrologic models. First, it is beneficial to know how well a model performs when it is parameterized entirely by automated processes and default values without calibration. Second, in cases where the objective is to predict hydrologic response to projected future conditions it is necessary to determine if assessments of relative change, derived from a comparison of uncalibrated simulation results, can provide reliable information.

The present study does not attempt to distinguish between the various sources of uncertainty that occur in deterministic hydrologic modeling. Instead it presents their cumulative effects and implications for using model projections as a basis for environmental planning. Specifically, we address the hypothesis that differencing results from two simulations to derive the relative change can minimize uncertainty associated with individual simulations and provide a useful means of qualitatively evaluating hydrologic response to landscape change when calibration is not possible. The study focuses on the SWAT model as implemented through the AGWA modeling interface.

STUDY AREA

AGWA-SWAT was applied to the Upper San Pedro River Basin above the USGS Charleston gauge (Figure 1). The San Pedro River flows north from Sonora, Mexico into southeastern Arizona. The area is a transition zone between the Chihuahuan and Sonoran deserts and has a highly variable climate with significant biodiversity. The study watershed is approximately 3196 km² and is dominated by desert shrub-steppe, riparian, grasslands, agriculture, oak and mesquite woodlands, and pine forests. Large changes in the socio-economic framework of the basin have occurred over the past 30 years, with a shift from a rural ranching economy to considerably greater urbanization. As the human population has grown, so too has groundwater withdrawal, which threatens the riparian corridor and the long-term economic, hydrologic, and ecological stability of the basin.

Satellite data were acquired for the San Pedro basin for a series of dates over a period of 25 years: 1973, 1986, 1992, and 1997. Landsat Multi-Spectral Scanner (MSS) and Thematic Mapper (TM) satellite images have been reclassified into 10 land-cover types ranging from high altitude forested areas to lowland grasslands and agricultural communities, with 60-meter resolution. The most significant changes were large increases in urbanized area, mesquite woodlands, and agricultural communities, and commensurate decreases in grasslands and desert scrub (Kepner and others 2002).

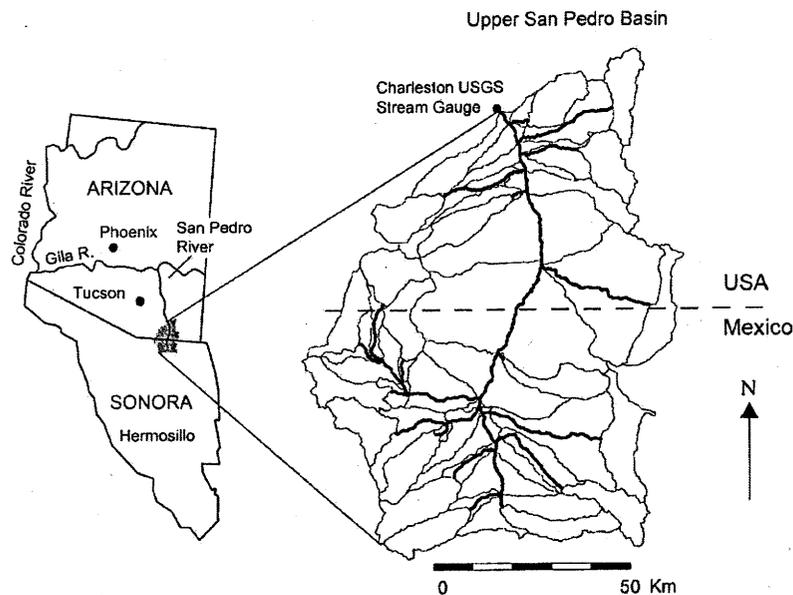


Figure 1-- Map showing the location of the Upper San Pedro River Basin and the watershed discretization for SWAT, with 53 subwatersheds.

METHODS

AGWA was used to delineate the Upper San Pedro watershed, subdivide it into model elements (subwatersheds and stream reaches), and derive an initial parameter set and input files for SWAT using the 1973 land cover. Precipitation inputs were derived from a total of seven National Weather Service gauges within the basin, and distributed across the subwatersheds using a Thiessen polygon weighting scheme (Semmens and others 2002). Agricultural withdrawals and diversions were incorporated into this default parameter set, and model performance was noted for the period from 1966-1975. The year of 1966 was defined on the basis of utilizing the maximum number of rain gauges with continuous daily rainfall records. A calibration exercise was then carried out for the same period. SWAT was calibrated for base flow, surface runoff, and water yield. Results from the automated base flow separation program (Arnold et al., 1995) were used to identify the groundwater contribution to the total water yield.

Additional "future" simulations based on the 1986, 1992, and 1997 land-cover/use grids were carried out for the equal-length periods of 1979-1988, 1985-1994, and 1990-1999, respectively. They incorporated all known changes to agricultural water withdrawals and diversions. Management actions such as these are a common component of alternative future scenarios, and considered to be something that can reasonably be projected along with land use. Simulations for the future conditions were carried out in two ways: once with no calibration using the default parameter set derived from AGWA, and once using optimized parameters from the baseline conditions. For the latter, parameters not derived from land-cover datasets were retained in all simulations; parameters derived from land cover were first estimated using each data set and then adjusted in the same way they were during the calibration (e.g. 10% reduction in Curve Number).

Climate, and in particular precipitation, is a major source of uncertainty in hydrologic modeling. Changing climate and its associated impacts on basin hydrology, however, are a significant concern associated with future predictions. In this exercise climatic inputs for all simulations were treated in two ways. Simulations were run first using the observed daily precipitation and temperature associated with each simulation period. Figure 2 provides a summary of the total annual precipitation during each of the four 10-year simulation periods. The simulations were then repeated using precipitation and temperature from the 10-year calibration period for the three "future" scenarios (i.e. 1986, 1992, and 1997). The latter treatment of climate is the most practical means of deriving climatic inputs for future simulations, and has the added benefit of eliminating climatic variation from assessments of hydrologic response to landscape change. Together, the two treatments of climatic inputs provide a means of estimating the proportion of model uncertainty in future simulations that is associated with unknown climatic conditions.

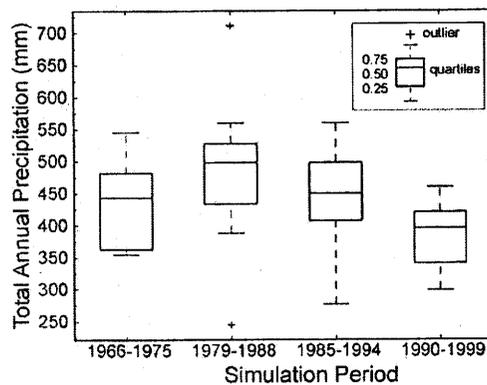


Figure 2-- Box plot showing the spread and distribution of total annual precipitation for each simulation period.

Four simulations were thus carried out for each of the four simulation periods to yield a total of 16 simulations. Results were compared in terms of Nash Sutcliffe model efficiencies relative to observed water yield for each simulation. A second set of comparisons was then carried out to determine how the model fared in terms of predicting change relative to the 1973 baseline condition. Simulation results for the baseline condition were subtracted from those for the three future conditions to compute relative percent change in the average annual water yield according to equation 1:

$$[(\text{future}) - (\text{baseline})] / (\text{baseline}) * 100$$

(1)

AGWA incorporates the functionality to do this automatically for each pair of simulation results and for each model element, producing what is effectively a new set of results (percent change) for each model element that can be mapped over the watershed. Although insufficient data are available to evaluate predicted change across the watershed, a visual comparison of distributed results was made to evaluate whether the model was able to qualitatively predict the spatial patterns of hydrologic response when calibration-period (historic) climate inputs were used, and when the model was not initially calibrated. In addition, all results were compared with observed change at the watershed outlet.

RESULTS AND DISCUSSION

Model Performance

As expected, results from the initially calibrated simulations using observed climate data best reproduced the observed conditions for all simulation periods (Figure 3). Uncalibrated simulations using observed climate data capture the trends quite well, and although they over predicted water yield they did so consistently. Also as expected, simulations based on the historic climate inputs vary considerably from those based on observed inputs. Predicted water yield increases with time reflecting land-cover/use change in the basin, but the much stronger influence of climate renders meaningless projections of water yield at any point in the future. Uncalibrated simulations produce an almost identical trend of increasing water yield with time, consistently over predicting the initially calibrated simulations.

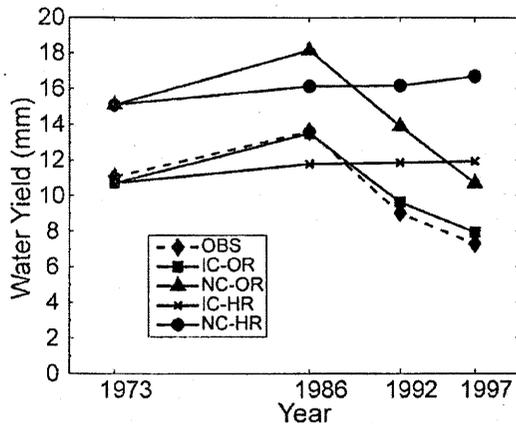


Figure 3-- Observed and simulated average annual water yield (mm) for the simulation period around each land-cover dataset. Simulations are abbreviated as: initial calibration (IC), no calibration (NC), observed rainfall (OR), and historic rainfall (HR).

Model efficiencies for the initially calibrated simulations using observed climate inputs were quite good, although they declined somewhat for the most distant "future" simulation (Table 1). Uncalibrated simulations using observed climate inputs had much lower efficiencies that declined slightly with time. For simulations based on historic climate inputs, model efficiencies were lower and declined over time.

Table 1-- Nash Sutcliffe model efficiencies for the simulation periods around each land-cover dataset

Simulation	1973	1986	1992	1997
Initially calibrated, observed climate	0.89	0.72	0.94	0.5
No calibration, observed climate	0.04	0.25	-1.21	-1.4
Initially calibrated, '66-'75 climate	0.89	0.21	-1.2	-4.69
No calibration, '66-'75 climate	0.04	0.12	-4.65	-13.9

Relative Change Assessment

Percent change in average annual water yield was also dominated by climatic inputs (Table 2). Both sets of simulations using observed climate inputs did reasonably well at predicting the observed change in water yield. Interestingly, the initially calibrated simulations did not yield better predictions of change, and although not statistically significant the uncalibrated simulations more closely predicted the observed changes on average. With climatic variability removed, simulations based on historic climate inputs were not able to predict the magnitude of observed changes in water yield.

Table 2-- Observed and simulated percent change in average annual water yield from the 1973 baseline condition

Simulation	1986	1992	1997
Observed change	22.85	-18.52	-34.15
Initially calibrated, observed climate	25.93	-10.07	-25.93
No calibration, observed climate	20.2	-7.95	-29.07
Initially calibrated, '66-'75 climate	9.98	10.63	11.38
No calibration, '66-'75 climate	6.85	7.15	10.6

Having established that it is inappropriate to use simulations based on historic climate inputs to predict future conditions, one additional comparison is necessary to evaluate how uncalibrated simulations predict the spatial pattern of changes in water yield. Although no data are available to confirm these patterns, changes predicted by the initially calibrated simulations using observed climate data are the best available means of estimating them. Figure 4 presents maps of the change in water yield predicted by the initially calibrated simulations using observed climate data (A) for comparison with those derived from: (B) uncalibrated simulations using observed climate, (C) initially calibrated using historic climate, and (D) uncalibrated simulations using historic climate data. Although the values in the legends indicate a substantially different range of values between the sets of simulations, it is noted that some of the major spatial patterns of predicted change are reasonably similar despite the fact that different, distributed precipitation inputs were used. Subwatersheds exhibiting the greatest change (positive and negative – lightest and darkest colors) in water yield match well between all four sets of simulations. One way of interpreting this result is that although the overall changes in watershed response are dominated by climate, within the watershed there are areas where the land-cover changes are significant enough to dominate hydrologic response, and climate is of secondary importance in determining relative water-yield changes. This effect is more pronounced (i.e. change assessments match more closely) during the drier periods around the 1992 and 1997 simulations.

A quantitative comparison of the predicted hydrologic response to landscape change was performed by computing the correlation coefficients between the water-yield change results for individual subwatersheds in each simulation period (Table 3). Results show a strong correlation between the change predicted by the A simulations and that predicted by the others (B, C, and D). Correlations are highest between the A and B simulations based on observed climate data, and lowest between the A and D simulations, which differ in terms of calibration and climate inputs. Correlations are lowest for 1986, which is the wettest of the four simulation periods. This result agrees with the interpretation of the previous paragraph; during periods characterized by higher precipitation climate is more likely to dominate hydrologic response.

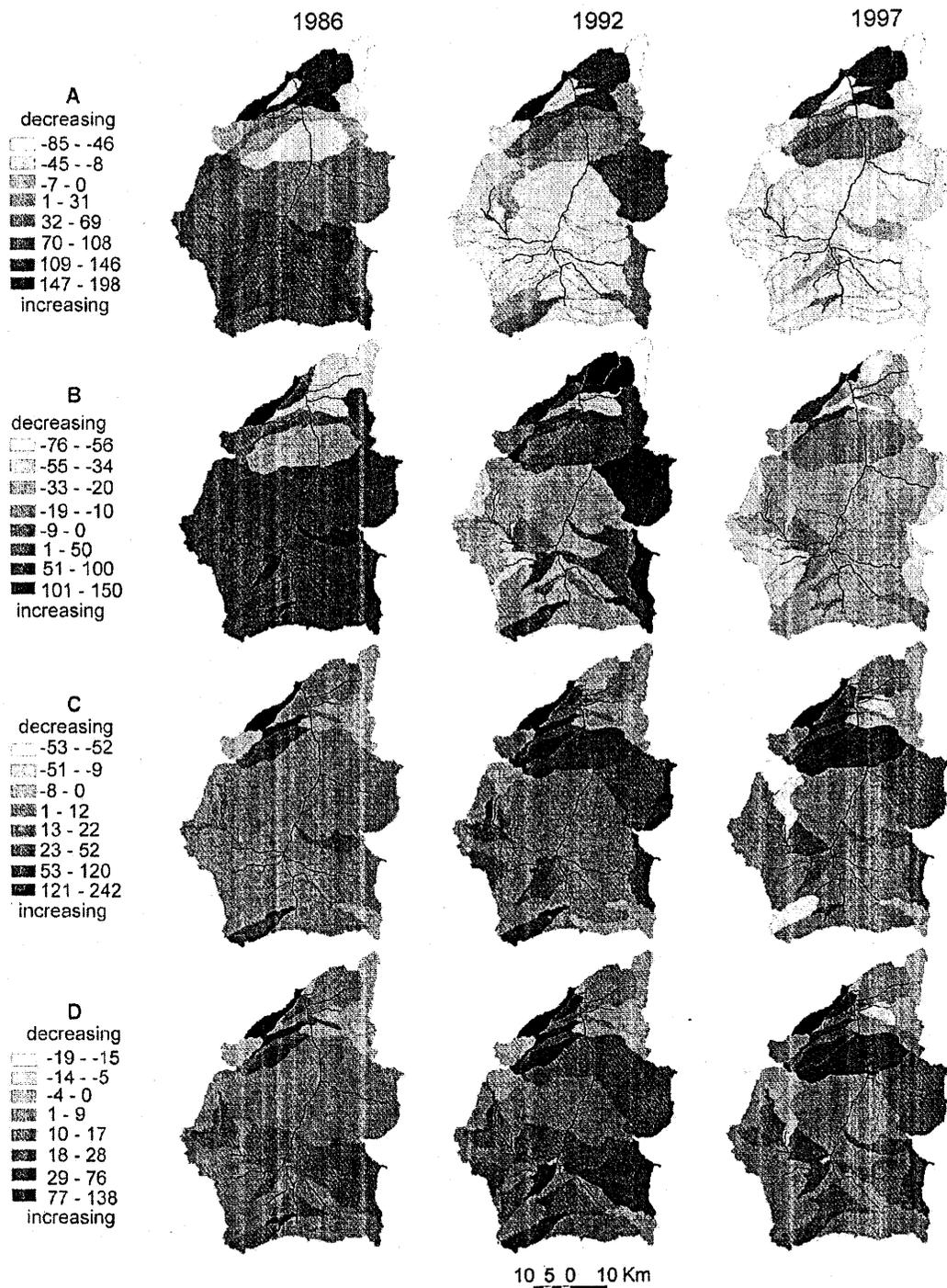


Figure 4-- Maps showing change in water yield relative to 1973 baseline for: A) initially calibrated simulations using observed climate, B) uncalibrated simulations using observed climate, C) initially calibrated simulations using historic climate, and D) uncalibrated simulations using historic climate.

Table 3-- Correlation coefficients relating the spatial predictions of water yield change from the A simulations to those from B, C, and D for each "future" landscape condition (see text for simulation descriptions)

Correlation	1986	1992	1997
A-B	0.812	0.834	0.830
A-C	0.481	0.784	0.815
A-D	0.478	0.732	0.715

CONCLUSIONS

Predicting hydrologic response to projected future land-use/cover conditions is a particularly challenging task because it is not independently verifiable. This paper explores some of the difficulties associated with the use of a GIS-based hydrologic model as a predictive tool to guide planning-related decision support. Results demonstrate that if future land-use/cover and climate conditions are known, then the model does a pretty good job of predicting observed conditions almost 25 years into the future. With initial calibration to baseline conditions, the model was able to provide reliable, quantitative estimates of average annual water yield. Significant performance declines were not observed until somewhere between 19 and 24 years into the future. Without calibration to a baseline condition the model was unable to provide quantitative estimates of average annual water yield, but was able to predict changes over time just as well as it did with initial calibration.

Unfortunately, future land-use/cover and climate conditions can never be known with certitude. The goal of regional planning efforts is to explore desired outcomes, and it is assumed that policy can be used to shape future change and guide it towards a particular outcome. As a result, climate conditions are the primary unknown in projecting future hydrologic response. Results of the present study indicate that by holding climate constant, it is possible to evaluate qualitatively the broad spatial patterns of hydrologic response to landscape change within a basin. Even when calibration for baseline conditions is not possible (i.e. for an ungauged basin), it is still possible to identify a significant portion of areas that are likely to experience the greatest amount of change, both positive and negative. The methodology of using historic rainfall and automated, GIS-based parameter estimation to evaluate hydrologic response to future land-use change can thus provide useful, qualitative information for planning-related decision support.

Given the sensitivity of hydrologic response to climatic conditions, future research will focus more attention on the use of climate scenarios to characterize hydrologic response for a range of climatic conditions. With a suitable weather generator it would be possible to hold land-use constant and explore model performance using observed and generated future climate. In this manner it may be possible to partition predicted hydrologic response into the portions derived from land-cover/use and climate change for a range of climatic conditions. If successful, this methodology could provide a means of deriving quantitative estimates of hydrologic response for various future land-cover/use scenarios.

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USING THE HYDROLOGIC MODEL MIKE SHE TO ASSESS DISTURBANCE IMPACTS ON WATERSHED PROCESSES AND RESPONSES ACROSS THE SOUTHEASTERN U.S.

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Abstract-- A clear understanding of the basic hydrologic processes is needed to restore and manage watersheds across the diverse physiologic gradients in the Southeastern U.S. We evaluated a physically based, spatially distributed watershed hydrologic model called MIKE SHE/MIKE 11 to evaluate disturbance impacts on water use and yield across the region. Long-term forest hydrologic data from a southern Appalachian Mountain and a lower coastal plain watershed in South Carolina were used as model inputs. The model captured the temporal and spatial dynamics of shallow groundwater table movement and streamflow. Results suggest climate change and tree removal would have pronounced hydrologic effects; especially during dry periods. We also found that the data parameterization for even small scale distributed watershed-scale modeling remains challenging where spatial subsurface characteristics are often not known. The global change implications on hydrologic processes and response to in the two landscapes are discussed.

INTRODUCTION

Over half the land mass of the Southeastern U.S. is forested. The region has high biodiversity, and favorable climate for plant and animal growth, and human habitation. However, forest ecosystem services are threatened by global changes that include population growth, urban sprawl, climate change, and other natural and human stressors (Wear and Greis, 2004). These current and future biotic and abiotic changes will have long-term impacts on watershed ecosystems through their direct effects on the water cycle within the region (McNulty and others 1998). Although forested watersheds provide the best water, potential water quantity and water quality degradation from intensive forest management practices, landuse changes, wildfires and other disturbances is of regional concern (Swank and others 2001). Watershed management and restoration practices, such as Best Management Practices (BMPs) require an accurate understanding of the basic controlling factors of hydrologic processes at a watershed-scale across the diverse physiologic gradients in the Southeastern U.S. (Sun and others 2004)

The southeastern U.S. has a long history of forest hydrologic research (Jackson and others 2004; Amayta and others 2005). Over the past 50 years, numerous watershed manipulation experiments were conducted in strategic locations representing the three major physiographic regions across the southeast (i.e., coastal plain, piedmont, and mountain). The paired watershed experiments developed by those studies provided much of our current knowledge about the hydrologic processes and how watershed responds to disturbances and alternative land management practices. Past studies suggest that forest harvesting an increase in water yield and elevates groundwater tables due to the reduction of total ecosystem evapotranspiration. The increase in stream run-off has also been associated with elevated nutrient and sediment loading to streams (Swank and others 2001; Sun and others, 2001). Water quality effects diminish with vegetation regrowth and forest canopy cover restoration. The time required for canopy restoration to pre-disturbance levels is relative short compared to other part of the nation, and varies from a few years to several decades (Sun and others 2004). Synthesis studies in the southeast region (Sun and others 2002; Sun and others 2005) and worldwide literature (Andreassian 2004; Sun and others 2006) suggest that climate, soil, and topographic class (e.g., wetlands vs. uplands) control the hydrologic processes and responses to disturbance or land management. For example, shallow groundwater tables dictate the slow moving streamflow processes in forested watersheds on the flat coastal plains (Riekerk, 1989; Amayta and Skaggs, 2001) while hillslope processes and gravity (both saturated and unsaturated subsurface flows) control the water flow in steep mountain watersheds (Hewlett and Hibbert, 1967). Over 70% of precipitation returns to the atmosphere as evapotranspiration in the coastal watersheds due to high temperature, but upland watersheds in the piedmont and Mountains have a lower proportion of the total precipitation returned to the atmosphere as ET (i.e., 30-70% of precipitation) due to lower temperature and higher

precipitation (Lu and others 2003; Lu and others 2005). A larger fraction of the precipitation is therefore removed from the watershed as higher stream flow peaks and volumes in piedmont and mountain regions.

Hydrologic modeling has become an essential and powerful tool in watershed studies (Graham and Butts, 2005), and perhaps the only way to extrapolate hydrologic experimental finding from small watersheds to large basins and the region. Process-based, spatially distributed models are best suited for understanding how different types of watersheds respond to disturbance.

Traditional small watershed experiment used a 'black box' approach that focuses on the effects of land management on streamflow measured at the watershed outlets. Modern forest hydrologic studies focus on the processes and interactions between the hydrologic cycle and other biological processes under a changing environment at multiple temporal and spatial scales. Computer simulation models are useful in this technology transition in hydrological research.

It is important to understand how climate and topography influence hydrologic responses to disturbance at the regional scale across the southeast U.S. for regional water supply and forest management and policy purposes. Empirical studies in the past century have provided valuable location-based data to develop mathematical models to 'scale up' hydrologic findings to the region and examine how forested watersheds respond to global change. McNulty and others (1996) and Liang and others (2002) examined potential climate change impacts on regional forest water yield using the monthly time step, stand level forest ecosystem models, PnET-II and PnET-3SL, respectively. Both models linked forest growth, productivity, and water use (ET). Both models proved useful in modeling regional ET from forests. Sun and others (2005) applied a simpler annual-scale ET model and examined the potential water yield changes due to deforestation across the region. Those types of region-orientated modeling studies show a strong spatial variability in predicted forest water use and yield due to variations of climate, topography, and forest types across the region.

In this paper, we hypothesized that different regions would have different hydrologic responses to forest management practices and climate change due to differences in topographic, climatic, and vegetation conditions. Models that are developed on physical principles should be applicable to physiographic regions. Thus, the objectives of this study were: 1) to test the process-based, groundwater-surface integrated watershed hydrologic model, MIKE SHE, to accurately predict ground water movement at multiple sites with long-term forest hydrologic monitoring data across the southeastern U.S., and 2) to apply the validated model to examine hydrologic responses to forest harvesting and climate changes across a physiographic gradient across the region.

METHODS

Study Sites

MIKE SHE model evaluation and application was conducted at two research sites representing the coastal plain and the Appalachian Mountain physiographic regions - in the southeastern U.S. (Figure 1). These are two intensively studied small experimental forested watersheds with varying land cover and soil types (Table 1). One watershed is located on the mountainous upland of North Carolina and the other is located on the lower coastal plain in South Carolina. Streamflow, baseflow and peakflow rates, and spatial distributions of groundwater table were the major hydrologic variables used in the evaluation of model performance. Detailed results for two additional model testing sites are found in a dissertation by Lu (2006).

Table 1. Characteristics of two watersheds used for model evaluation in the Southeastern U.S.

Site	Landscape characteristics	Area (ha)	Average Precipitation (mm/year)	Soil	Vegetation Coverage	Data for calibration and validation
Santcc Watershed 80	Coastal plain, SC	160	1370	Sandy loam	Mixed hardwood and pine	2003 model calibration; 2004 model validation (streamflow and water table)
Coweeta Watershed 2	Appalachian Mountains, NC	12	1772	Loamy soil	Mixed hardwood	1987-1988 model calibration; 1985-86; 1989-1990 model validation (streamflow)

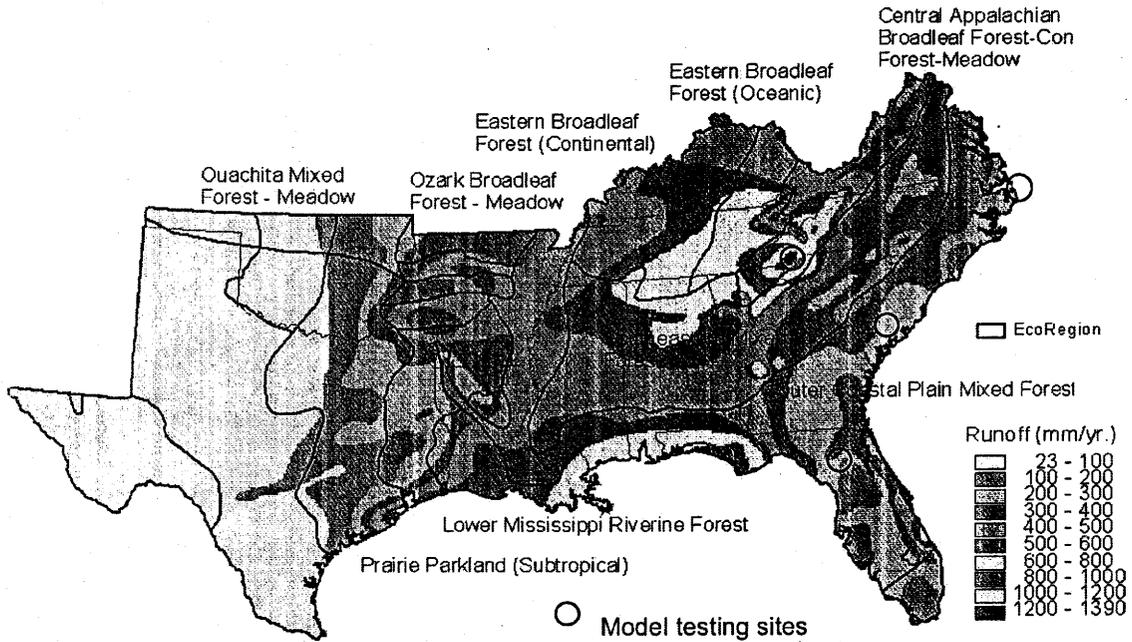


Figure 1 - Provinces of the Bailey ecoregion, annual runoff, and locations of model testing sites in the southeastern U.S.

The Santee Watershed 80 is located in the Santee Experimental Forest, part of the Francis Marion National Forest, on the lower Atlantic Coastal Plain, eastern South Carolina (33.15°N, 79.80°W) (Figure 2). As the control watershed for a paired watershed study, it was installed in the mid-1960s by the USDA Forest Service for studying forest management for water quality and quantity in the coastal plain geographic region (Amatya and others 2005; Harder, 2004). The watershed has a low topographic relief (< 4%) with surface elevation ranging from 3 - 10 m above mean sea level and consists of an ephemeral stream as the main drainage pathway.

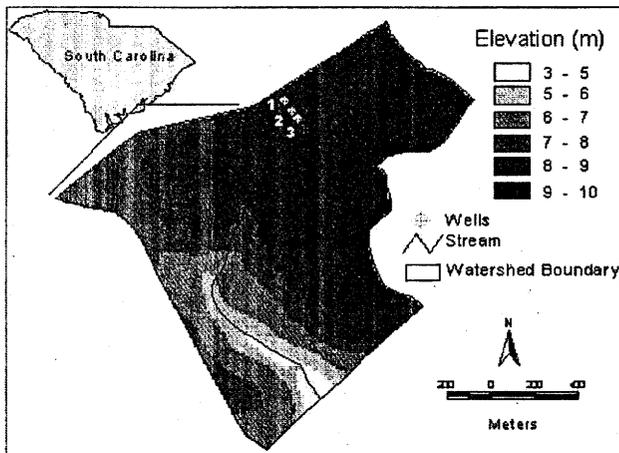


Figure 2 - Santee Watershed 80 topography and instrumentation.

The climate of the study site is classified as humid subtropical with long hot summers and short mild winters. Mean annual precipitation is about 1370 mm. July and August are the wettest months (receiving 28% of total annual precipitation) and April and November are the driest months (receiving 10% of total annual precipitation). January is the coldest month with a maximum average low air temperature of 10 °C, and July is the hottest month with a maximum average high air temperature of 28 °C. The mean annual air temperature is 19.1 °C. Approximately 23% of the watershed is classified as wetlands (Sun and others 2000). The forest coverage is mainly composed of pine-hardwood (39%), hardwood pine (28%) and mixed hardwoods (33%). Dominated tree species include loblolly pine (*Pinus taeda L.*), sweetgum (*Liquidambar styraciflua*), and a variety of oak species typical of the Atlantic Coastal Plain. Most of the trees are 17 years

old, and regeneration after damage caused by Hurricane Hugo in 1989. The study site consists primarily of sandy loam soils with clay subsoils. Much of the soil is part of the Wahee-Lenoir- Duplin association (SCS, 1980) that are somewhat poorly drained to poorly drained (SCS, 1980). Soils are influenced by seasonally high water tables and argillic horizons 1.5 meters below ground surface.

Coweeta Watershed 2

Coweeta's 12 ha, Watershed 2 was selected for model evaluation and application of the southern Appalachian Mountain upland conditions. This watershed is located at the USDA Forest Service Coweeta Hydrologic Laboratory, a Long Term Ecological Research (LTER) Center site in western North Carolina (35°03'N, 83°25'W) (Figure 3). The Coweeta Hydrologic Laboratory was established for forest hydrologic research in 1934 and has been a National Science Foundation (NSF) LTER site since 1980 (Swank and Crossley 1988). Large amount of classic forest hydrologic research has been conducted in this facility (Hibbert 1967; Swank and Douglas, 1974; Swift and Swank 1975; Swank and others 1988; Vose and Swank 1992).

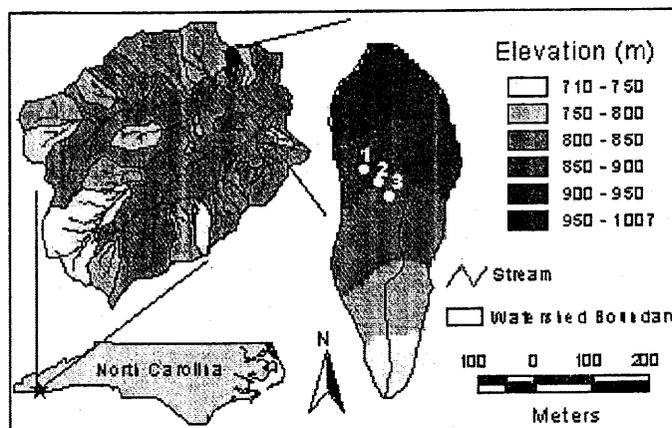


Figure 3 - Location of the study watershed (Watershed 2) at Coweeta Hydrologic Laboratory in southern Appalachian Mountains, western NC.

The climate in Coweeta is classified as marine, humid temperate with a water surplus in all seasons. Watershed 2 is one of Coweeta's control watersheds, and has not been disturbed since it was clear-cut in 1927. The average of 1772 mm of precipitation is evenly distributed through out the year. The mean annual streamflow is around 854 mm, which is 48% of precipitation. Long-term average monthly air temperature is as low as 3.3 °C in January and is as high as 21.6 °C in July (Swank and Crossley 1988). The watershed has an average slope of 23° with a maximum of 49°. The elevation ranges from 710 m at the watershed outlet to 1007 m at the ridge top. The south-southeast facing watershed is covered by mixed hardwoods with scattered Pitch Pine (*Pinus rigida*) on the ridge top. Tree species mainly include eastern hemlock (*Tsuga canadensis*), tulip (*Liriodendron tulipifera*), sweet birch (*Betula lenta*), white oak (*Quercus alba*), and red oak (*Quercus rubra*) with great rhododendron (*Rhododendron maximum*), flame azalea (*Rhododendron calendulaceum*), laurel (*kalmia latifolia*), and blueberry (*Vaccinium pallidum*) as the understorey. The soil series are Chandler and Fannin (Swank and Crossley, 1988; Rosenfeld, 2003). The soil physical parameters are derived from Vose and Swank (1992). Although Watershed 2 is a much smaller 12-ha watershed, it has a first order perennial stream flowing year round.

The MIKE SHE/MIKE 11 Model

Numerous watershed-scale hydrologic models are available in the literature. The choice of models should be based on the objectives of use. Literature review suggests that the MIKE SHE/MIKE 11 modeling package (DHI, 2004) (Figure 4) has several advantages for achieving our objectives: 1) it is a distributed model and most of the algorithms in describing the full water cycles are physically based, 2) it simulates explicitly groundwater-surface water interactions, so it is ideal for wetland dominated systems as well as storage-based systems commonly found in humid regions, 3) it has been commercialized and a GIS user interface was built in the system that can directly use spatial GIS databases for model inputs. Also, the model has a strong visualization facility that makes interpretation of modeling outputs much easier.

MIKE SHE is a first generation of spatially distributed, physically based, hydrologic model (Abbott et al., 1986a, 1986b). MIKE SHE simulates the terrestrial water cycle including evapotranspiration (ET), overland flow, unsaturated soil moisture and groundwater movement (Figure 3). Evapotranspiration is modeled as a function of potential ET, leaf area index, and soil moisture content using the Kristensen and Jesen (1975) method. The

unsaturated soil water infiltration and redistribution processes are modeled using Richard's equation or a simple wetland soil water balance equation. Saturated water flow (groundwater) is simulated by a 3-D groundwater flow model similar to the MODFLOW model (McDonald and Harbaugh, 1988). Channel flows and channel surface water and upland groundwater interactions are handled by the MIKE 11 model and coupling of MIKE SHE and MIKE 11. MIKE 11 is a one-dimensional model that tracks channel water levels using a fully dynamic wave version of the Saint Venant equations. The coupling of MIKE SHE and MIKE 11 is especially important for simulating the dynamics of variable source areas in both wetland and upland watersheds. Detailed descriptions of the modeling procedures and mathematical formulation can be found in the MIKE SHE user's manual (DHI, 2004) and associated publications (Abbot and others 1986a, 1986b; Graham and Butts 2005).

Identical graphical and statistical methods were used to evaluate models performance for the two watersheds. The statistical measures included mean estimation error (ME), Correlation Coefficient (R) and the Nash-Sutcliffe (1970) coefficient of efficiency (E). The model was first calibrated with data from 2003 for the Santee and for data from for each site for 2003 with data from 1988 to 1989 for the Coweeta watershed. The models were validated with 2004 data from Santee and with from 1985-1987 and 1990 for Coweeta (Table 1).

Model Application Scenarios

After model calibration and validation were conducted, the MIKE SHE model was applied to four scenarios for both watersheds. These scenarios included: 1) Base line (BL); 2) Clear Cutting (CC); 3) a average annual temperature increase of 2 °C (TI); and 4) a average annual precipitation decrease (PD) of 10%. The purpose of the scenarios were to examine watershed hydrologic response land management and climate change.

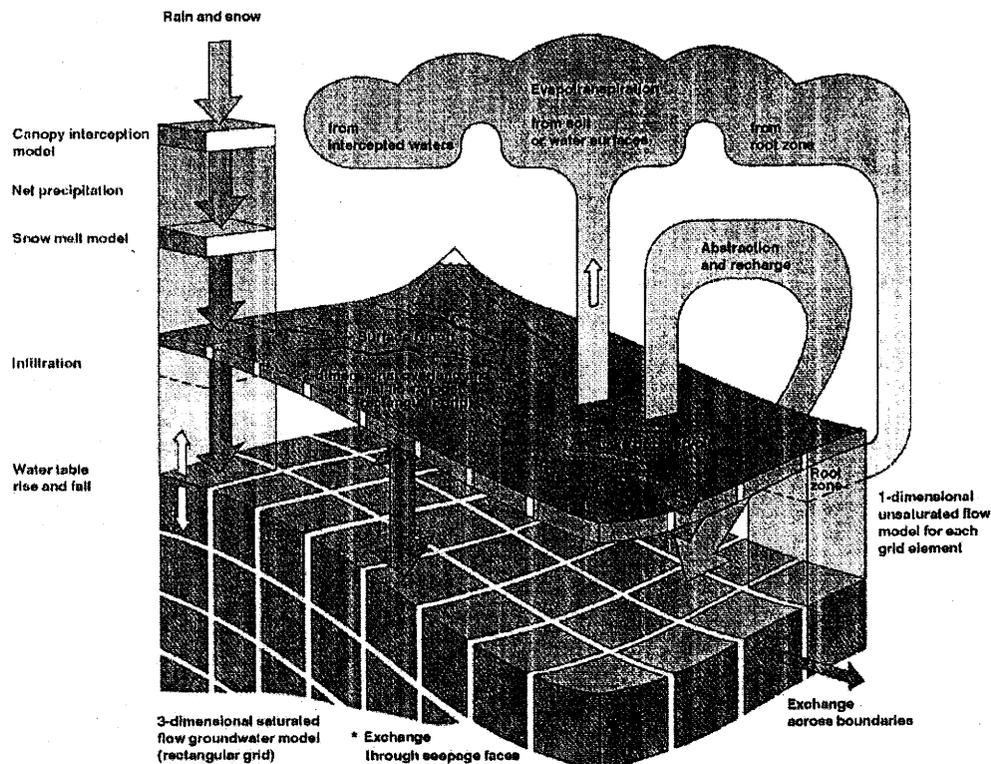


Figure 4 – MIKE SHE Model structure and hydrologic components(DHI, 2004)

RESULTS

Model Calibration and Validation

Santee Watershed 80

The MIKE SHE model was calibrated against the daily streamflow data from 2003. Compared to the long-term annual average precipitation at the study site, 2003 was a wet year with a 300 mm surplus in precipitation (Harder, 2004). Model parameters were finalized after calibrations resulted in the best match between simulated and measured daily streamflow as gauged by the established model performance criteria.

Generally, the model could simulate the daily variations of streamflow with $R = 0.75$, $ME = 0.10 \text{ mm day}^{-1}$ and $E = 0.56$ during the calibration (Figure 5). However, the model did not catch all the peak flows, especially for one large mid-September storm event (i.e., Hurricane Isabelle). The simulated peakflow rate of $0.37 \text{ m}^3 \text{ sec}^{-1}$ was much lower than the measurement peakflow rate of $1.44 \text{ m}^3 \text{ sec}^{-1}$ for that storm event.

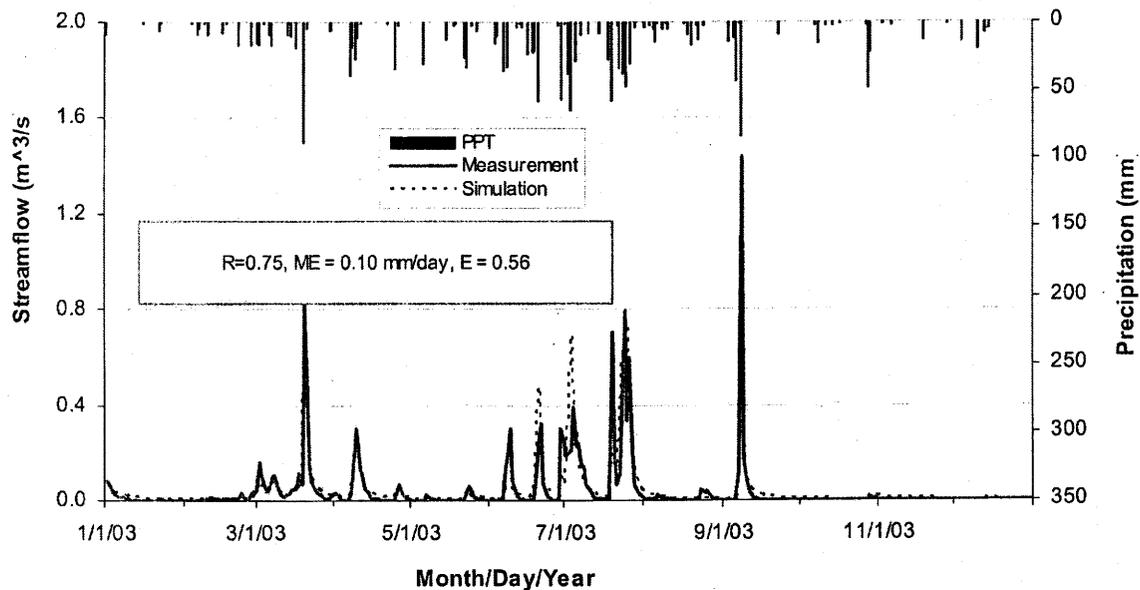


Figure 5 – MIKE SHE model calibration with daily streamflow in 2003 at Santee Watershed 80.

The MIKE SHE model was validated with daily streamflow data in 2004 and water table depth measured from the end of 2003 through 2004. 2004 was a dry year with 409 mm less rainfall than the long-term annual average. There were only three stormflow events in the entire year, and there was no streamflow observed during June, July, October, November, and December (Figure 6). Overall, MIKE SHE simulated the streamflow dynamics with a $R = 0.75$ under these extremely dry conditions, but it over-predicted a peakflow rate in late August. Overall, the model over-predicted streamflow with $ME = -0.31 \text{ mm day}^{-1}$ and $E = -3.61$. Simulated water table depths compared reasonably with Well #1 (Figure 3, and Figure 7) but failed to match measured stream flow depths in two other wells located on the watershed. (not shown).

Coweeta Watershed 2

The MIKE SHE model was calibrated with the daily streamflow data in 1988 and 1989. Compared to the long-term annual average precipitation of 1770 mm, 1988 was a dry year (i.e., 1267 mm), and 1989 was a wet year (i.e., 2341 mm). Generally, the model performed reasonably well with $R = 0.88$, $ME = -0.04 \text{ mm/day}$ and $E = 0.74$ (Figure 8). On the annual basis, the model over predicted streamflow in 1988 by only 31 mm, and almost exactly predicted measured 1989 streamflow of 1019 mm.

At the daily basis, the biggest differences in streamflow were found in January of 1988 and during the summer of 1989. The approximately 7 mm day^{-1} over estimate of streamflow in the early 1988 may have been partially caused by poor estimates of initial conditions. The base line simulation run from 1985 to 1990 showed that the difference in predicted and measured streamflow was reduced to 5 mm day^{-1} for that particular date. The largest discrepancy in the summer of 1989 may have been due to the relatively shallow soil depth used in this modeling study. We used the same soil parameters to a depth of 3 m since there was no data available for soil properties below the 1.8 m depth. The soil properties were distributed uniformly across the entire watershed. In reality, the soil depth is likely to be highly variable across the watershed (Yeakley 1993; Miner 1968).

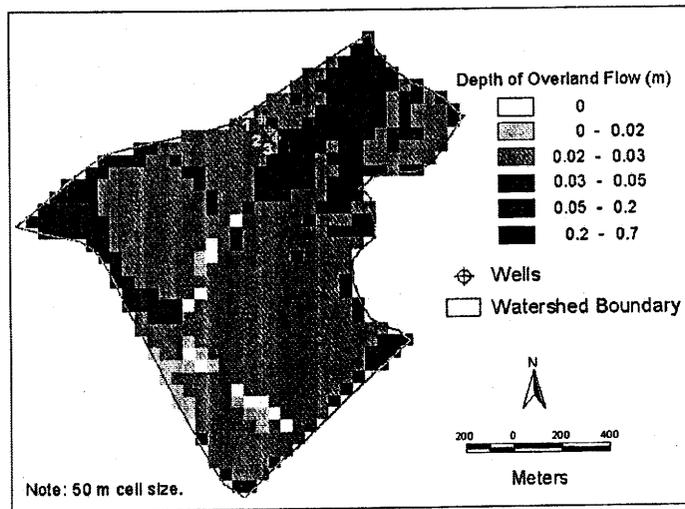


Figure 6 – Simulated spatial distribution of overland flow depth on 06-20-2003 at Santee watershed. Spatial resolution is 50 m.

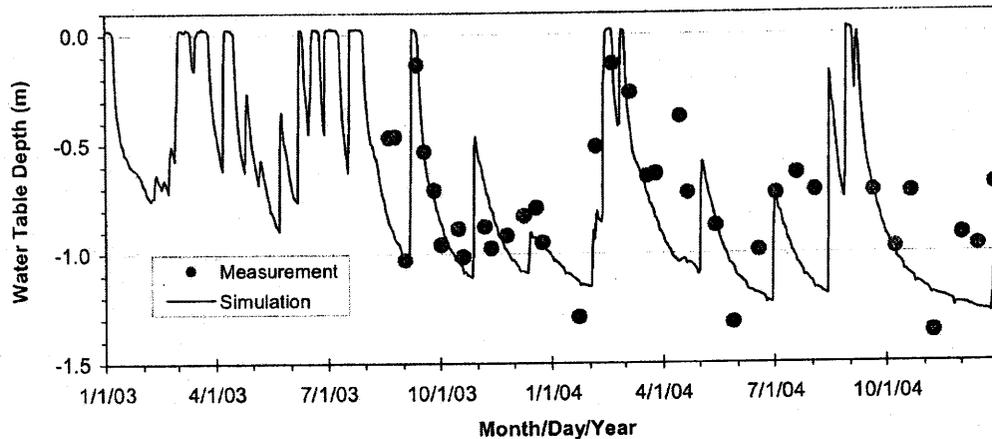


Figure 7 - Validation of MIKE SHE model with water table depth at Well 1 during October 2003 - December 2004 at Santee Watershed 80.

The MIKE SHE model was validated with the daily streamflow data recorded in 1985-1987 and 1990 (Figure 9). Compared to the long-term average precipitation at the watershed, 1985-1987 were extreme dry years and 1990 was a wet year. The model generally could match the streamflow dynamics with $R = 0.85$, $ME = 0.04 \text{ mm day}^{-1}$ and $E = 0.72$. Simulated streamflow values were close to measured except for the big storms in February and March of 1990 when the model overestimated daily streamflow values up to 10 mm day^{-1} . On an annual basis, the model had a tendency to over-predict streamflow in a dry year and under-predict streamflow in a wet year.

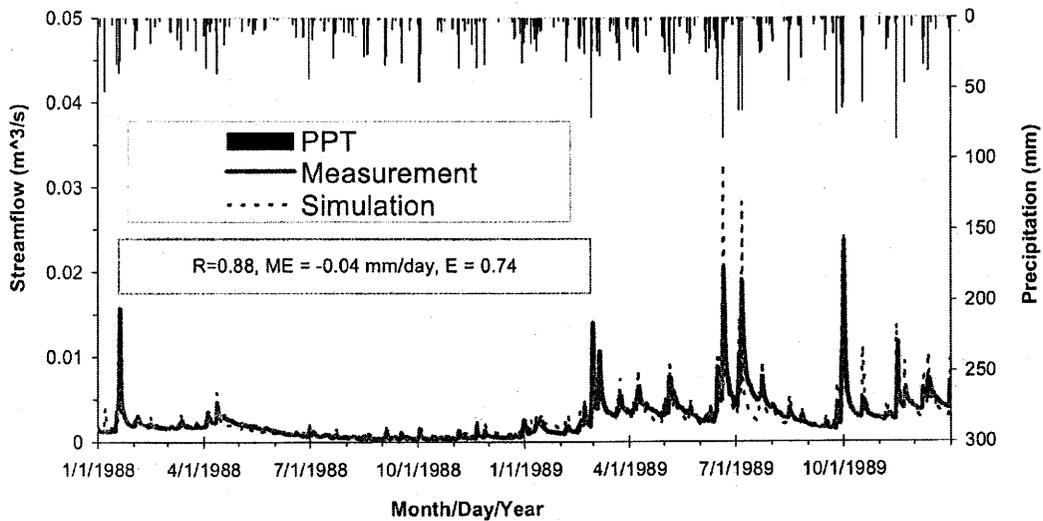


Figure 8. MIKE SHE model calibration with daily streamflow in 2003 at Coweeta Watershed 2.

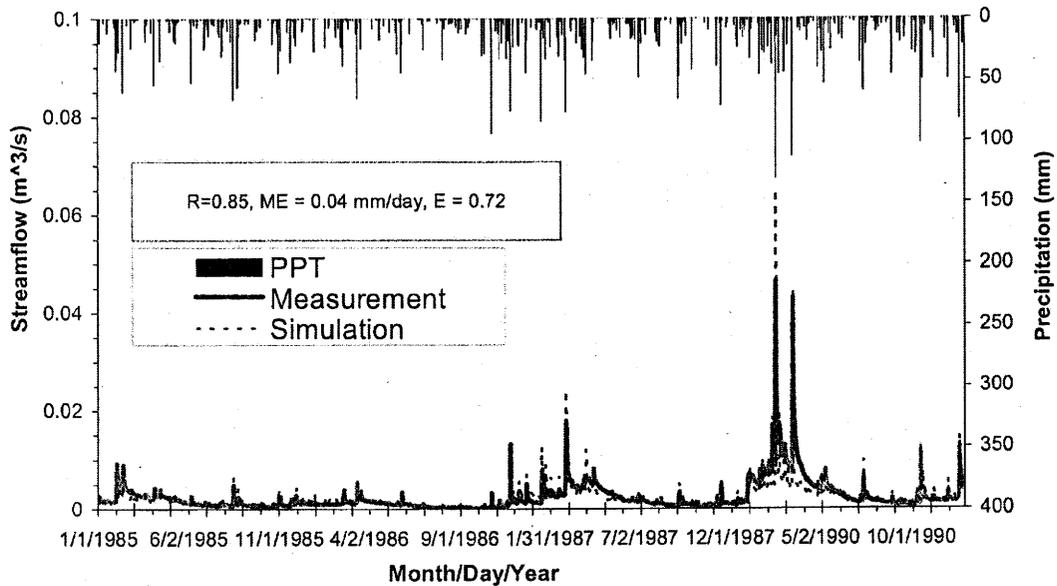


Figure 9. Model validation with daily streamflow during 1985-1987 and 1990.

Model Applications

We evaluated the potential effects of three hypothetical scenarios on the ground water table and annual water yield during 2003 and 2004 at the Santee Watershed 80, and from 1985-1990 for Coweeta Watershed 2 using the validated model from both watersheds (Figure 10). Our simulation results suggested that clear-cut would decrease ET, elevate the groundwater table level, and increase water yield. The effect is especially pronounced during dry

periods when the ET differences between the baseline (BL) and disturbed scenarios were largest. Harvesting reduces leaf area and will result in a decrease in potential ET. Plant transpiration capacity and total ecosystem ET will decrease, and therefore soil water recharge for streamflow generation will increase. Increase in temperature by 2°C caused increase in PET, while decrease 10% precipitation caused direct soil water recharge. Both climate change scenarios will result in lower water table level.

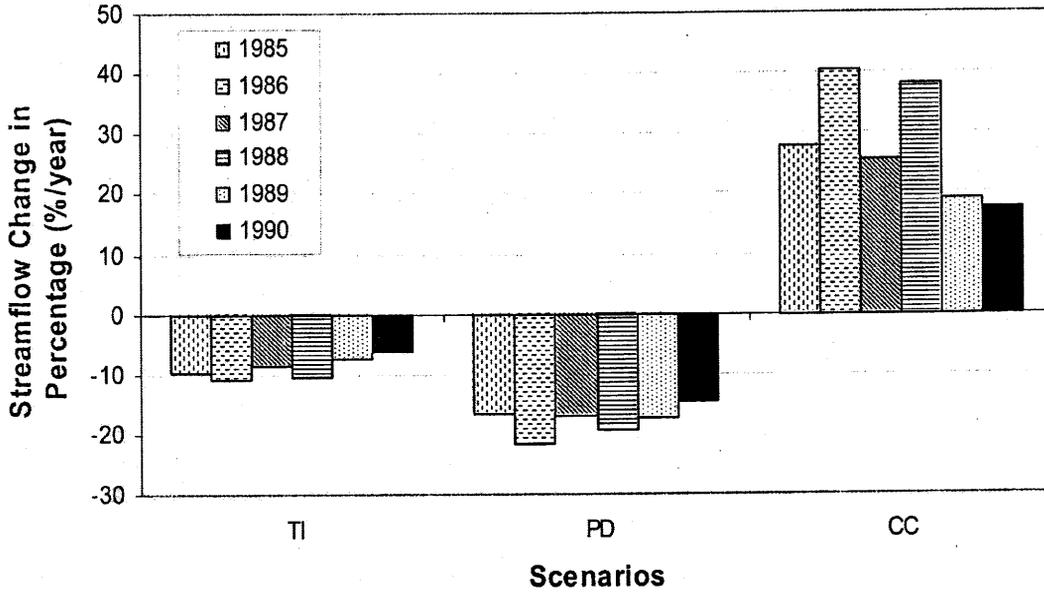
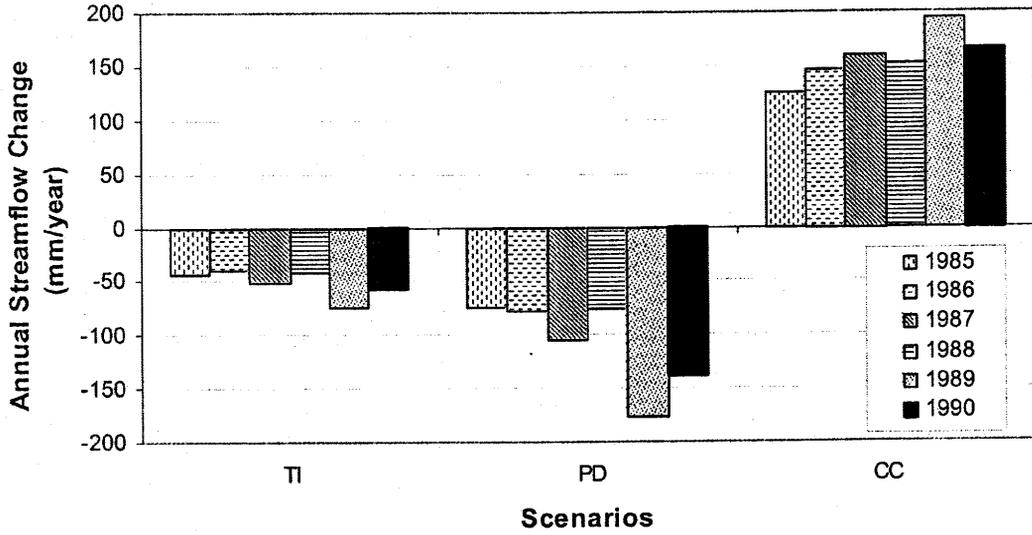


Figure 10. Simulated effects of clearcutting (CC), increase of air temperature by 2°C (TI), decrease of precipitation by 10% on streamflow as expressed by: a) change in absolute annual streamflow amount, b) change in percentage of annual streamflow at the Coweeta Watershed 2. For the CC case, a 30% reduction of potential ET was assumed (Grace and Skaggs, 2006; Sun, G. Unpublished data).

DISCUSSION

Computer simulation models are a powerful tool for data syntheses, understanding the hydrologic processes, and for predicting potential future conditions. Watershed-scale experiments are expensive to conduct, and a modeling approach can be cost-effective, especially for answering large-scale hydrologic questions. The MIKE SHE model was evaluated with hydrologic data from two small headwater watersheds on two separate contrasting landscapes in the humid southeastern U.S. In general, the model performed reasonably well for simulating daily streamflow measured at the watershed outlet, and for estimating the spatial distribution patterns of the shallow groundwater table depth. However, parameterizing the physically based, distributed watershed-scale model was a challenging task, even for small watershed with a size in the tens of ha. For example, data on the spatial distribution of soil water storage, the depth until bed rock, or on restricting soil layers is rarely available. Thus, model calibration is still needed produce reasonable model results. Climate and landuse change will have a major impact on ET. Thus process-based ET algorithms are needed in the MIKE SHE model. Validating distributed hydrologic models requires detailed measurements of internal processes, such as streamflow at subwatersheds, spatial distribution of water table over the landscape, soil moisture distribution, and hillslope processes. Most of those measurements are rarely available in one watershed.

Streamflow in the flat coastal watershed (Santee Watershed 80) is controlled by the dynamic shallow groundwater table that reflects the water balances of rainfall and ET. Saturation-overland flow contributes the majority of the total flow in the first order, ephemeral stream. In contrast, saturation overland flow is not common in the steep mountain watershed (Coweeta Watershed 2). The MIKE SHE model could simulate the variable source areas that contribute directly to stormflows during large rainfall events. It appears that the Santee Watershed 80 had higher variability of storage capacity as characterized by large peakflows and discontinuous flow patterns when compared to the upland watershed. The flushness of the wetland watershed reflects large extent of overland flow during large storm events. The upland watershed suggested to have higher water 'turnover' rates (low residence time) because of the frequent rainfall events and steeper hillslopes. The frequent rainfall and lower ET in the upland watershed result in continuous streamflow in this mountain watershed. Accurately simulating the narrow saturated variable source areas in the upland watershed remains elusive because the model data is restricted to the coarse 10 m digital elevation model spatial resolution.

CONCLUSIONS

This study concluded that the hydrologic response to disturbance in the two watersheds varies with climate. Soil moisture is normally unrestricting for plant growth on both the upland and wetland watersheds in the humid southeast region. Hydrologic responses are most pronounced during dry years when surface soil evaporation is minor, but forest transpiration is usually not severely reduced even during dry years.

Findings from this study have important implications to forest management practices at the regional scale. Best Management Practices (BMPs) for protecting water quality from harvesting or wildfires should consider a much larger extent than just the riparian zones as the practices do for the hilly piedmont and mountain regions because watershed-wide overland flows in the lowland watersheds are the sources of surface waters in the coastal region. In contrast, overland flow is rare and saturated subsurface flows are the sources of streamflow for the upland watersheds. Thus, forest roads that are often cut into bed rocks will likely alter water flowpaths, resulting in increase in sediment and peakflow rates. Best management practices should include on the entire hillslope for the protection of riparian zones.

Findings from this study also have implications to water yield under the projected global climate change. If global warming results in an increase in air temperature and drought, the role of forests in affecting streamflow, especially baseflow (lowflows) will increase. On another extreme climate change scenario, when rainfall intensity increases, the role of forest cover in soil protection will be most important.

The role of soil depth on hydrologic response to disturbances has been well examined in the literature. Also, mechanistically understanding watershed responses to forest management and climate change need focus on the changes in forest evapotranspiration processes. Improvements to the ET algorithms in MIKE SHE model are needed for its application in global change studies.

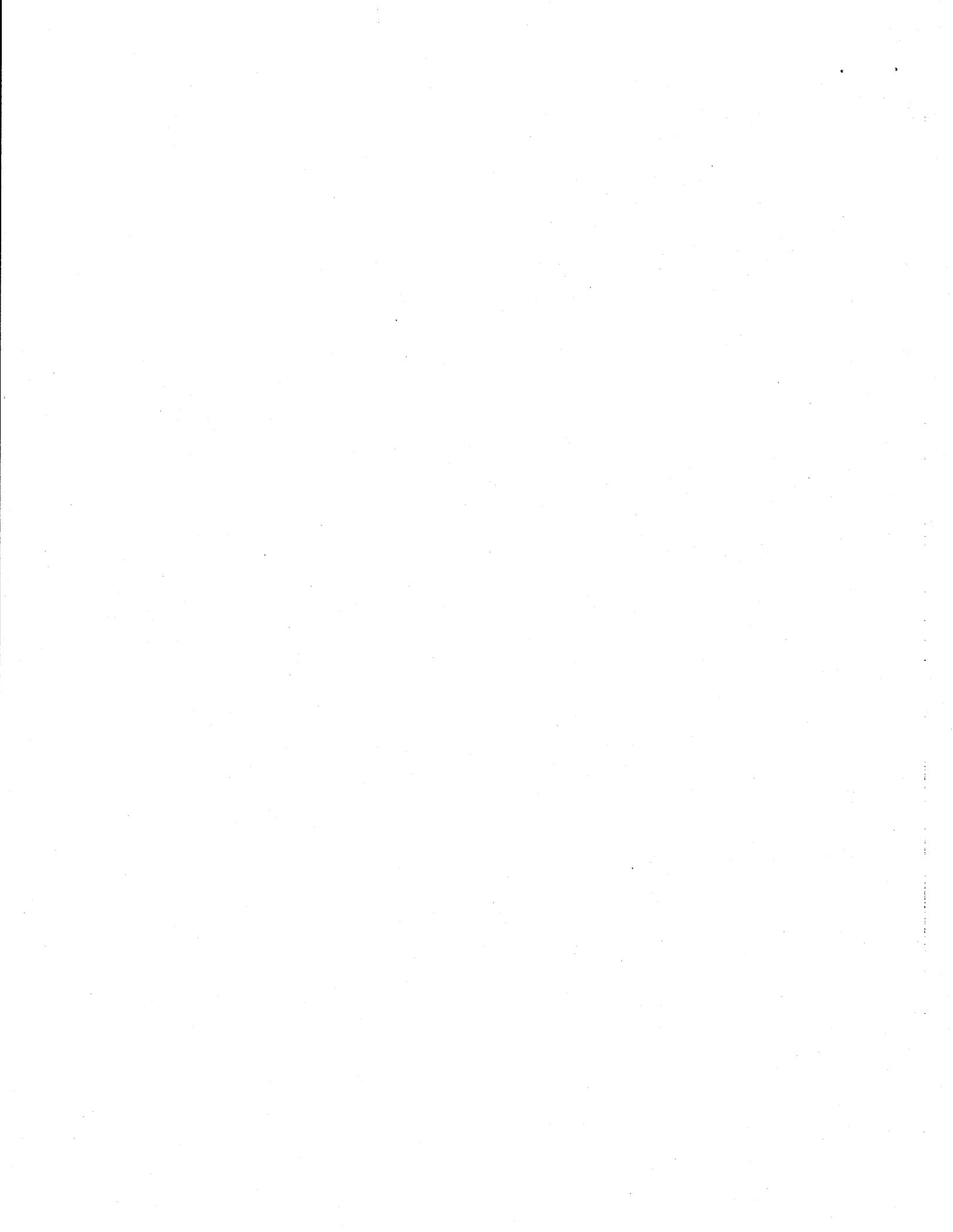
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PARTITIONING EVAPOTRANSPIRATION USING DIURNAL SURFACE TEMPERATURE VARIATION

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Abstract--An approach is proposed in which daily ET, measured with a conventional Bowen ratio technique, is partitioned into E and T using coincident measurements of diurnal soil surface temperature. Sites dominated by woody and herbaceous vegetation in the USDA ARS Walnut Gulch Experimental Watershed were instrumented with automated sensors as part of the NASA/USDA Soil Moisture Experiment SMEX04. At each site, surface temperature, soil moisture, soil temperature and meteorological data (including solar radiation, precipitation and evapotranspiration) were measured at 1 to 20-minute intervals over an eighteen-month period in 2004 and 2005 encompassing the dry/hot season, the North American monsoon and the dry/cool season. Results showed that apparent thermal inertia (I_A), defined as the difference in soil surface temperature at 2:00 pm and 5:00 am, was related to soil evaporation. The I_A values exceeding a nominal threshold were used to identify days when E was negligible, and consequently, $ET \approx T$. Further work is planned to derive a total annual T/ET ratio to better understand the ecohydrological consequences of woody plant encroachment in semiarid grasslands.

INTRODUCTION

Encroachment of woody plants in grasslands is becoming a common phenomenon across the Western U.S. This transformation is of particular interest in semiarid regions because woody-dominated vegetation has a different water demand than that of herbaceous vegetation. This is manifest in the water loss from evapotranspiration (ET) across the semiarid landscape. Though woody plant encroachment may not impact the total ET, it can alter the relative contributions of soil evaporation (E) and plant transpiration (T) to ET. In turn, these shifts in E versus T related to vegetation change can impact net ecosystem production and carbon cycling. In a landmark analysis of vegetation dynamics in drylands, Huxman et al. (2005) identified the partitioning of E and T as one of the most important ecohydrological challenges.

An approach is proposed in which daily ET, measured with a conventional eddy covariance or Bowen ratio technique, is partitioned into E and T using coincident measurements of diurnal soil surface temperature. The difference between the mid-afternoon and pre-dawn soil surface temperature, termed Apparent Thermal Inertia (I_A), was used to identify days when E was negligible. It is demonstrated herein that when I_A reached a seasonal maximum, E approached zero. With this set of measurements at a given site, ET can be measured; dates for which $E \approx 0$ can be identified; and T can be determined as the residual. Furthermore, instrumentation for these measurements can be maintained in place continuously for years, as demonstrated in this and other studies.

APPARENT THERMAL INERTIA

By definition, soil thermal inertia (I) represents the ability of soil to conduct and store heat, where

$$I = (k\rho c)^{1/2} [J m^{-2} K^{-1} s^{-1/2}]. \quad (1)$$

In Eq. (1), k = thermal conductivity [$W m^{-1} K^{-1}$]; ρ = density [$kg m^{-3}$]; and c = heat capacity [$J kg^{-1} K^{-1}$]. Like I, apparent thermal inertia (I_A) also represents the resistance of soil to temperature change. However, it is derived instead from the difference between mid-afternoon and pre-dawn (or soil) temperatures, where

$$I_A = (t_{2pm} - t_{5am}) [^{\circ}C]. \quad (2)$$

The terms t_{2pm} and t_{5am} represent soil surface temperatures measured with a down-looking infrared thermometer (IRT) at times 2:00 pm and 5:00 am, respectively.

In early studies, I_A was loosely related to regional soil moisture (Kahle et al., 1987; Pratt and Ellyett, 1979). Though introduced in the early 1980s based on satellite images of surface temperature (Price, 1977), it was not easily interpreted over a heterogeneous terrain (Price, 1985). That is, I_A responds to changes in soil moisture and mineralogy, but it is also highly sensitive to changes in incoming solar radiation, as well as wind speed, air temperature and vapor pressure. In this application, these fundamental limitations in application of I_A are overcome by 1) computing I_A at one site and interpreting the signal over time rather than space, and 2) combining I_A with an on-site measurements of the surface (in this case, ET) to account for atmospheric conditions.

STUDY SITE, MATERIALS AND METHODS

The Soil Moisture Experiments 2004 (SMEX04) was conducted during the summer of 2004 in Arizona and Mexico to address overlapping science issues of the North American Monsoon Experiment (NAME) and soil moisture remote sensing programs. As part of SMEX04, two sites in the USDA ARS Walnut Gulch Experimental Watershed (WGEW) were instrumented with automated sensors to measure surface and atmospheric conditions. The Kendall site, dominated by herbaceous vegetation, is only 9 km from the Lucky Hills site, which is dominated by woody vegetation. At each site, surface temperature was measured with an IRT at 5-minute intervals and precipitation was measured at 1-minute intervals. Volumetric soil moisture (θ) was measured at 3 depths (5, 15 and 30 cm) with Vitel capacitance sensors at 5-minute intervals. Soil temperature (T_s) was measured at 1-, 2-, 5-, 6-, 15- and 30-cm depths with thermocouples at 20-minute intervals. Meteorological data (including incoming solar radiation and soil heat flux) were measured at 5- and/or 20-minute intervals. These sites were also equipped with flux stations to measure evapotranspiration using a Bowen ratio technique at 20-minute intervals (Emmerich, 2003). These data were analyzed over an eighteen-month period in 2004 and 2005 encompassing the dry/hot season, the North American monsoon and the dry/cool season, with particular attention to drying periods after storm events.

During the growing season in 2003, measurements of ET and T were made at the Lucky Hills shrub-dominated site. ET was monitored every twenty minutes using the flux Bowen ratio method (Emmerich, 2003), and shrub transpiration was measured every thirty minutes using the constant heat balance sapflow technique (Scott et al., in press). This shorter, but more comprehensive, data set was used to supplement and clarify the analysis of the 2004/2005 study at Kendall and Lucky Hills.

For these two studies, data sets of ET, T (in the 2003 study), volumetric soil moisture at 5cm, surface temperature (from IRT), and soil temperatures at multiple depths (in the 2004/2005 study) were compiled to study the partitioning of E and T. Results were compared for the Kendall and Lucky Hills sites dominated by herbaceous and woody vegetation, respectively.

RESULTS

As discussed in the previous section, I_A is theoretically related to both surface and atmospheric conditions. This sensitivity is illustrated by the response of I_A to a variety of surface and atmospheric conditions associated with a spring storm at Kendall (Figure 1). A precipitation event on DOY 147 and 148 resulted in a dramatic decrease in I_A associated with an increase in ET (due to increased soil moisture) and an associated decrease in available solar energy (due to cloudiness). For the clear-sky days that followed the storm event (DOY 149-152), I_A steadily increased as ET decreased, finally reaching a value similar to that before the storm. However, cloudy conditions on the following day (DOY 153) resulted in another dramatic decrease in I_A without any significant change in soil moisture. This demonstrates the difficulty in interpretation of I_A , and introduces the rationale behind the approach used here.

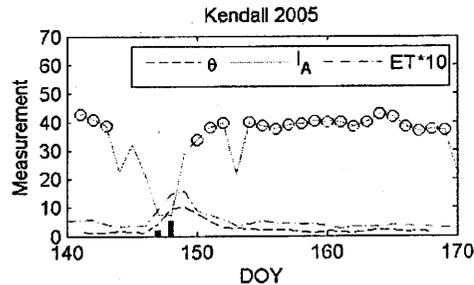


Figure 1--Comparison of volumetric soil moisture at 5 cm at 2:00 pm (θ , %), the apparent thermal inertia derived from IRT measurements at 2:00 pm and 5:00 am (I_A , °C, Eq. 2), and daily ET measured with the Bowen ratio method (mm/d, multiplied by 10 for presentation) for a storm event at Kendall in 2005. Circles indicate the cloudfree days and bars represent daily precipitation (mm).

Apparent Thermal Inertia Related to Soil Moisture and Evapotranspiration

Results showed that I_A was not well related to soil moisture for representative summer, winter and spring storms in 2004 and 2005 at WGEW (Figure 2). At Kendall (grassland) and Lucky Hills (shrubland), the I_A decreased immediately with precipitation, but returned to its pre-storm value within days of the storm, depending on atmospheric conditions. In contrast, surface soil moisture (at 5 cm) reached a peak a day or two after the storm, but continued to decrease for weeks thereafter.

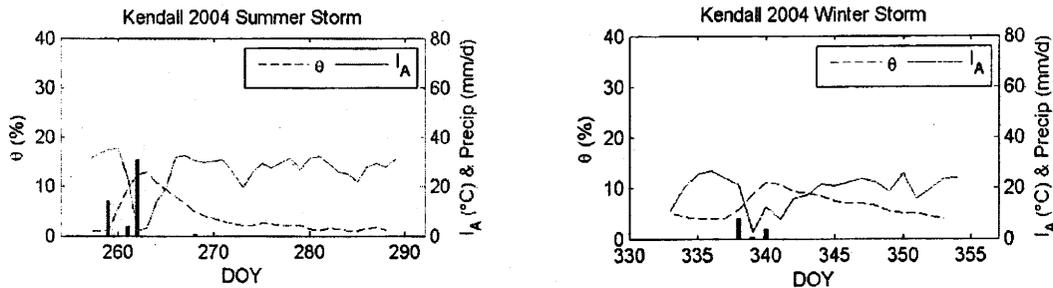


Figure 2--Comparison of I_A with volumetric soil moisture (θ) at 5cm at 2:00 pm for summer and winter storms at Kendall, followed by a long series of cloudfree days. Similar results were found (though not shown here) for Lucky Hills and for the spring storm in 2005. Bars represent daily precipitation (mm).

For the same winter and spring storms (when transpiration was known to be zero because vegetation was senescent), the I_A related well with ET (Figure 3b and 3c). Generally, the I_A was inversely correlated with ET and both measures returned to their pre-storm values within the same time period. For the 2004 summer storm (Figure 3a), the I_A post-storm recovery corresponded to a steep decline in ET (related to E). This was followed by a more gradual decline in ET, related to T. This trend was confirmed by the I_A and E measurements made at Lucky Hills in 2003 (Figure 4). For two small summer storms, variation (decrease and recovery) in I_A corresponded directly to the measured increase and subsequent cessation of E.

As one would expect from these results, the relation between soil moisture and ET is weak in both winter and summer (Figure 5). The ET is highly influenced by storm events and solar radiation, whereas the soil moisture has a less dramatic post-storm peak and steadily decreases until the next storm event.

Partitioning E and T from ET with Apparent Thermal Inertia

Based on the results in the previous sections, we postulated that the highest I_A values were associated with cloudfree days when soil evaporation was negligible. We also observed that I_A followed a seasonal trend in which higher values were obtained during the summer when solar radiation was at a maximum. To extract the days when $E \approx 0$ and $ET \approx T$, it was first necessary to detrend the annual I_A time series. For ease of computation, this was done in three 6-month sets, as follows.

For 6-month periods in 2004 and 2005, a polynomial was fit to the highest I_A values in the data stream (Figure 6a). Then, an adjustment was made to all the values to remove the seasonal trend relative to the first I_A value in the data stream (Figure 6b). Finally, a threshold was determined for the detrended I_A (I_{AD}) to select only the highest values of I_{AD} (Figure 6b). For dates which I_A exceeded that threshold, we presume that E was negligible and $ET \approx T$. Thus, daily T was estimated for selected dates for predominantly woody vegetation (Lucky Hills) and grassland (Kendall) over the time period 2004 and 2005 (Figures 6c and 6d).

DISCUSSION

This is a first step in an operational approach for partitioning E and T from *in situ* ET measurements. By using infrared thermometer measurements to determine dates when soil evaporation was negligible, it was possible to estimate plant transpiration rates for selected dates. The next step would be to derive either an empirical or theoretical model to determine the transpiration rates on dates when evaporation cannot be assumed to be zero. This could be based in part on transpiration rates identified through this process (Figure 6c and 6d) or empirical studies (Scott et al., in press) and/or on theoretical assumptions related to

soil surface energy balance and thermal regime (Moran et al., 1994). These investigations are ongoing, with some preliminary success.

Future work could also focus on the use of near-surface soil temperature measurements, rather than IRT measurements of surface soil temperature. This might be preferable since the instrumentation is less expensive. We found that the amplitude of I_A decreased with depth in the soil (Figure 7). Nonetheless, detrended I_A computed from soil temperatures at 1 cm produced results similar to I_{AD} based on IRT measurements (compare results in Figure 8 and Figure 6c).

These preliminary results show promise for determining the T/ET ratio for woody and grassland sites over prolonged periods to address the questions posed by Huxman et al. (2005). They presented hypotheses that, for semiarid sites, 1) woody plant encroachment should increase potential soil evaporation, and 2) the T/ET ratio is sensitive to changes in woody plant cover.

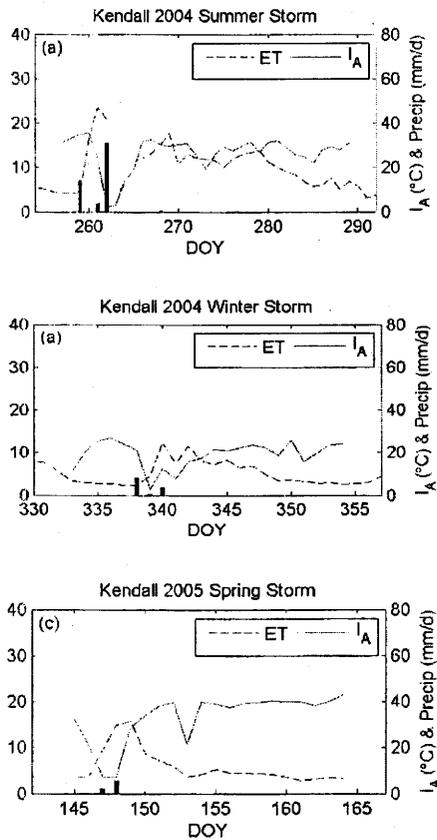


Figure 3--Comparison of I_A with daily ET (multiplied by 10 for presentation) for summer, winter and spring storms at Kendall, followed by a series of cloudfree days. Similar results were found (though not shown here) for Lucky Hills. Bars represent daily precipitation (mm).

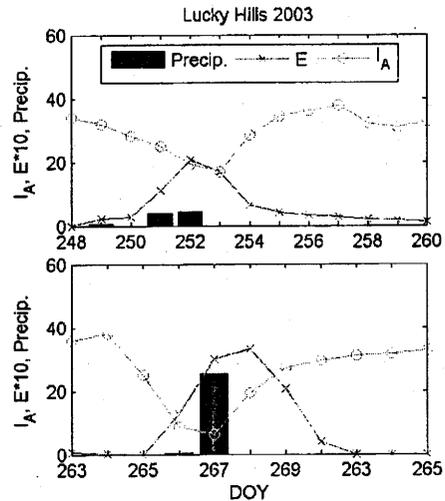


Figure 4--Comparison of I_A with daily E (multiplied by 10 for presentation) for two storms in 2003 at Kendall, when E was determined from the difference between ET (using Bowen ratio) and T (using sapflow technique). Bars represent daily precipitation (mm).

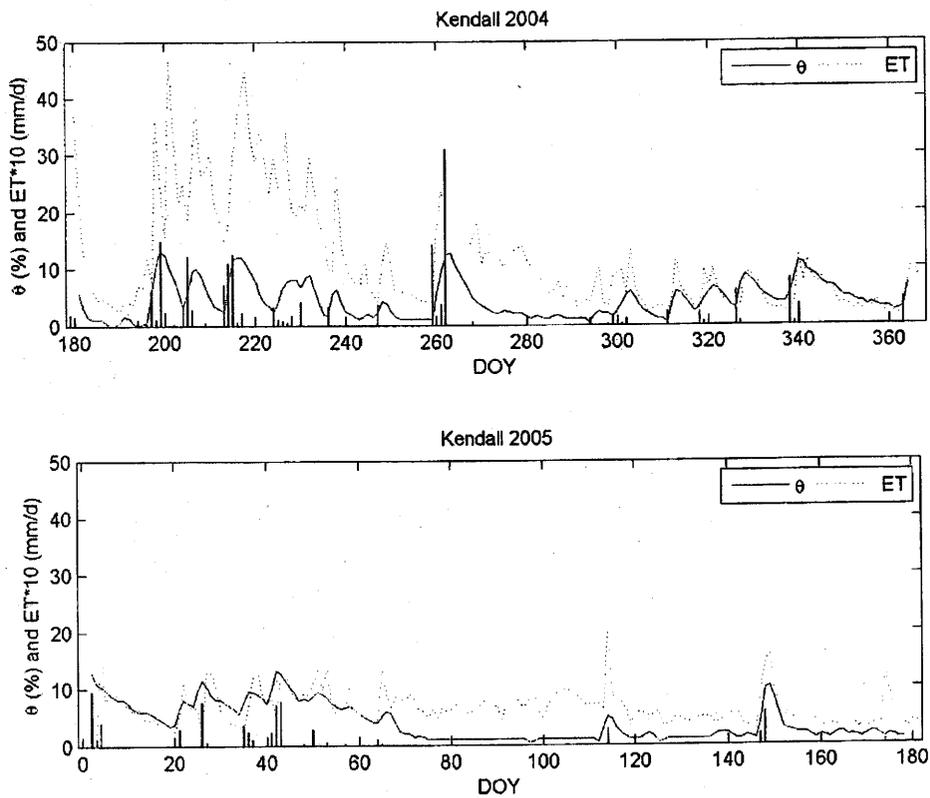


Figure 5--The weak relation between volumetric soil moisture (at 5 cm) and daily ET (multiplied by 10 for presentation) for a series of storms in the dry/hot season (DOY120-180), the North American monsoon (DOY 200-270) and the dry/cool season (near DOY 360). Bars represent daily precipitation (mm).

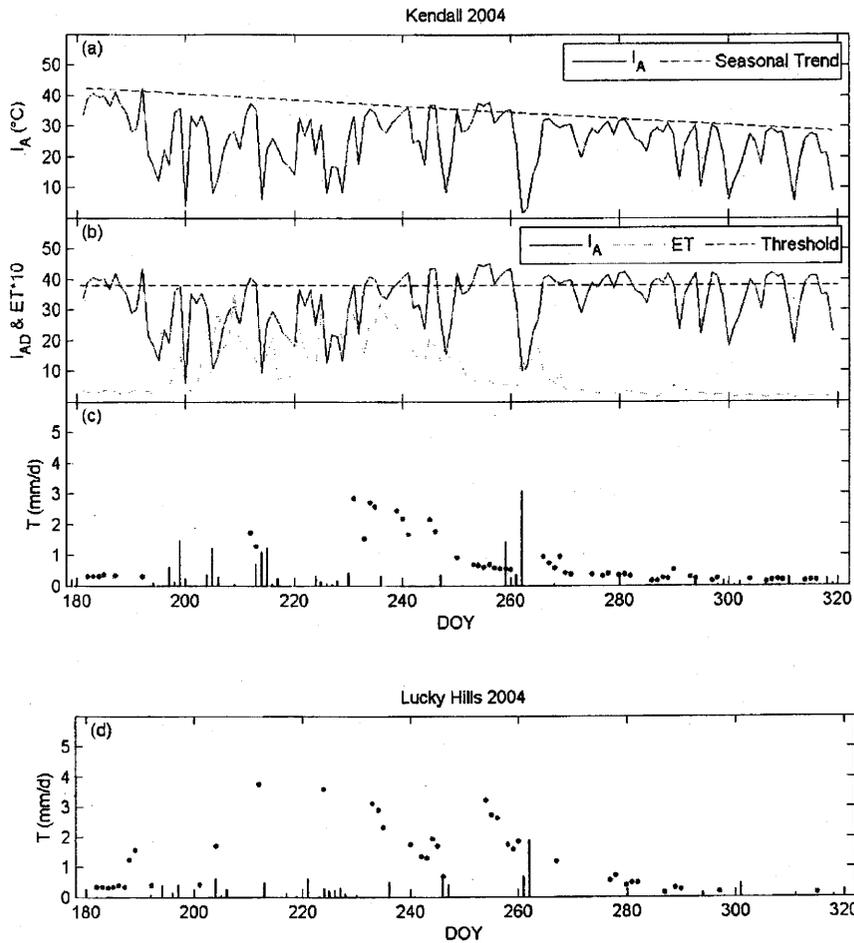


Figure 6--An illustration of the steps taken to partition ET using I_A at Kendall in 2004. (a) A polynomial was fit to the highest I_A values. (b) A threshold was set to discriminate the highest detrended I_A values (I_{AD}). (c) For dates when I_{AD} exceeded the threshold, then $ET=T$. (d) Values of T for Lucky Hills were derived using the same process illustrated at Kendall in Figures 6a-6c. Bars represent daily precipitation (mm).

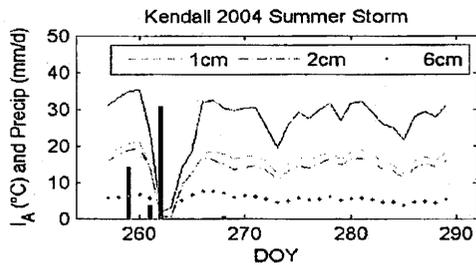


Figure 7-- I_A computed from soil temperature measurements at the surface (solid line), and at depths of 1 cm, 2 cm and 6 cm in the soil (with thermocouples). Similar results were found (though not shown here) for Lucky Hills and other storms. The bars represent daily precipitation (mm).

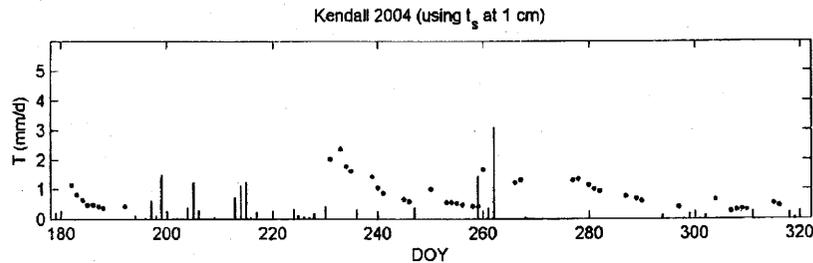


Figure 8--Daily transpiration at Kendall in 2004 derived from I_A using soil temperature at 1 cm instead of IRT measurements, following the steps illustrated in Figure 6a-6c. These results can be compared to results presented in Figure 6c using the IRT.

ACKNOWLEDGEMENTS

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EVALUATION AND APPLICATION OF THE MIKE SHE MODEL FOR A CYPRESS-PINE FLATWOODS WATERSHED IN NORTH CENTRAL FLORIDA

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Abstract--Pine flatwoods is an important ecosystem in the southeastern U.S. Long-term hydrologic impacts of forest management and climate change on this landscape are not well understood. A physically-based, distributed hydrologic modeling system, MIKE SHE, was evaluated and applied at a cypress-pine flatwoods watershed in north central Florida. This was part of the research that evaluated MIKE SHE/MIKE 11 for applications in examining disturbance impacts on water table and water yield across a physiographic gradient in the southeastern U.S. Our study showed that MIKE SHE could simulate the temporal and spatial dynamics of the shallow groundwater table. The model also identified and confirmed three horizontal groundwater flow patterns at this study site. The modeling results suggest that forest removal and climate change would have pronounced impacts on the groundwater table during the dry periods. However, there were no significant impacts under the wet conditions at this typical flatwoods landscape.

INTRODUCTION

Past studies suggest that current and future biotic and abiotic changes will have long-term impacts on watershed ecosystems through their direct effects on the water cycle in the southeast region (McNulty and others 1996, Sun and others 2004). Shallow groundwater tables dictate the water movement in forested watersheds on the flat coastal plains (Amayta and Skaggs 2001, Riekerk 1989). Over 70 percent of precipitation returns to the atmosphere as evapotranspiration (ET) in the coastal watersheds due to high temperature (Lu and others 2003, Lu and others 2005). The paired watershed experiments in the southeast provided much of our current knowledge about the hydrologic processes and how watershed responds to disturbances and alternative land management practices (Amayta and others 2005, Jackson and others 2004, Sun and others 2002, Sun and others 2005). These experiments used a 'black box' approach that focuses on the effects of land management on streamflow measured at the watershed outlets. Modern forest hydrologic studies focus on the processes and interactions between the hydrologic cycle and other biological processes under a changing environment at multiple temporal and spatial scales. Therefore, hydrologic modeling has become an essential and powerful tool in watershed hydrologic studies (Graham and Butts 2005), and perhaps the only way to extrapolate hydrologic experimental finding from small watersheds to large basins and the region. Process-based, spatially-distributed hydrologic models are best suited for understanding how watersheds respond to disturbance.

As an important ecological plant community in the southeastern Coastal Plain, the flatwoods covers approximately 50 percent of the Florida land area, or approximately three million hectares (Bliss and Comerford 2002). The Florida flatwoods landscape is a mixture of cypress wetlands and forest upland. The cypress swamps are depression wetlands providing many important ecological functions including groundwater recharge, water purification, wildlife habitat, and biomass production (Mansell and others 2000). Pine flatwoods are storage based hydrologic systems where the ground water table determines the timing and amount of surface runoff (Sun and others, 1998b). Models for the pine flatwoods have been developed (Mansell and others 2000, Sun and others 1998a, 1998b), and recently by Liu and others (2005). However, those models have not been thoroughly evaluated with spatial distributions of water tables.

This paper is part of the research that evaluated a physically-based, spatially-distributed watershed hydrologic modeling system, MIKE SHE/MIKE 11, for applications in examining the hydrologic processes and responses to forest management practices and climate change on the coastal plain and the mountainous upland in the southeastern U.S. (Lu 2006, Sun and others this volume). The objectives of this paper was to report results from model evaluation

and application at a typical flatwoods landscape. The model was used to study and the hydrologic responses (groundwater table depth) to potential land disturbance and potential climate change.

METHODS

Study Site

The study site was in Gator National Forest, which was located about 33 km northeast of Gainesville, Alachua County in north central Florida (Figure 1). A long-term intensive wetland hydrology study was conducted in the 1990s (Bliss and Comerford 2002, Crownover and others 1995, Sun and others 2000). Geology at this site was dominated by Plio-Pleistocene terrace deposits and the Hawthorne Formation with ground slopes ranging from 0 percent to 1.6 percent. The shallow ground water was separated from the underlying artesian secondary aquifer by impermeable blue-green clays (> 4 m thick), which was underneath the top organic and sandy soil layers (2-3 m thick).

The cypress swamps accounted for about 35 percent of the study area with wetland sizes ranging from a few square meters to more than 5-ha (Figure 1). Pond cypress (*Taxodium ascendens Brongn.*) dominated the vegetation in the wetland, along with slash pine (*Pinus elliottii Engelm*) and swamp tupelo (*Nyssa sylvatica var. biflora Sarg.*). The remaining upland was dominated by a 29-year old mature slash pine plantation in 1992 with saw palmetto (*Serenoa repens Small*) and gallberry (*Ilex glabra Gray*) shrubs as the understory (Sun and others 2000).

The average annual air temperature was 21 °C, with a mean monthly low of 14 °C in January and high of 27 °C in July. Average annual rainfall was about 1330 mm, with two distinct dry periods within a year. The first dry period was from April to June, and the second one was from October to December. The soil type was predominantly Pomona fine sand (a Spodosol; Utic Haplaquods; sandy siliceous, thematic) (Mansell and others 2000, Sun 1995).

Data Collection

Beginning in January 1990, the study area was surveyed to establish a 50 m × 50 m grid system. An arbitrary datum with the elevation of 30.48 m above mean sea level was set at the reference coordinate (0, 0). The actual elevation of the study site was about 47 m above mean sea level (Sun and others 2000). Each grid point was marked and labeled with a steel post, and its elevation relative to the datum was measured in the field.

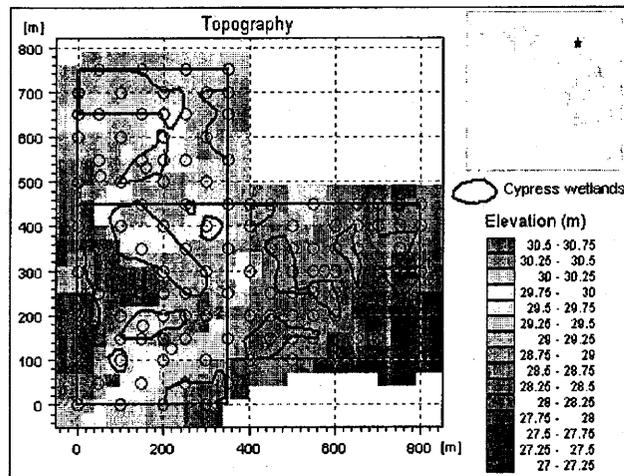


Figure 1--Topography of the Florida wetland site.

Shallow water table wells were installed at approximately every second grid point. Thus, 122 manual wells and six automatic wells were available at this site (Figure 1). The 5-cm diameter, 1.5-m long polyvinyl chloride (PVC) water table wells were installed in the holes drilled with a 5-cm hand auger. The bottom 1 m of the PVC pipes had well screening attached to a well point, and the remaining 0.5 m was a PVC riser with a well cap to cover the aboveground opening. Well depths varied from 1 m to 1.4 m, depending on the depth of the argillic horizon and water table conditions at the time of well installation (Crownover and others 1995). A more detailed description of the site establishment, well installations and earlier data reports can be found in Crownover and others (1995), Sun and others (2000), and Bliss and Comerford (2002).

Water table depths of the 122 manual wells were measured on a bi-weekly schedule from 1992 to 1995 (Crownover and others 1995, Sun and others 1998a). Water table depths of the six automatic wells were recorded daily from 1992 to 1996 (Sun and others 2000). Detailed description of well water table measurements and recordings were given in Crownover and others (1995) and Sun (1995).

The MIKE SHE Model

As a first generation of spatially distributed hydrologic modeling system, MIKE SHE simulates the full hydrologic cycle of a watershed across space and time. MIKE SHE can be applied to a wide range of water resources and environmental problems for the simulations of surface and ground water movement, the interactions between the surface water and ground water systems, and the associated point and non-point water quality problems. Detailed descriptions of the modeling procedures and mathematical formulation can be found in the MIKE SHE user's manual (DHI, 2004) and associated publications (Abbot and others 1986a, 1986b, Graham and Butts 2005).

Data requirements for the MIKE SHE model included: 1) Watershed topography and landuse data - retention storage, Manning roughness number, and vegetation distribution (leaf area index (LAI) and rooting depth dynamics); 2) Soil data - soil depth and hydraulic properties (conductivity, porosity, and soil moisture release characteristics); 3) Meteorological data - precipitation and temperature; and 4) Boundary conditions. Major outputs of the MIKE SHE model included ET, overland flow, unsaturated soil moisture content, and groundwater levels.

Model Calibration and Validation

Graphical inspections as a qualitative method and the statistical criteria as a quantitative method were used to evaluate the MIKE SHE model performances. The statistical parameters included mean error (ME), Pearson's correlation coefficient (R) and the Nash-Sutcliffe (1970) coefficient of efficiency (E). Water table data were used for the MIKE SHE model calibration and validation. After each model run, these statistical values were calculated to evaluate model performances. ME is commonly used to determine the average systematic error among the simulated and the observed values. Positive values of ME indicate model underpredictions, while negative values correspond to overpredictions. E varies from minus infinity to 1.0, with higher values indicating better agreement. R is a measure of the strength of the association between observed and predicted values. It may take any values between -1 and 1.

The study site was divided into three blocks - NW block, SW block and SE block (Figure 1). SW block was not disturbed during 1992-1996, but cypress wetlands in NW block were harvested and both wetlands and uplands in SE block were harvested by clear-cutting methods. Within each block, a representative cypress wetland-upland system was selected, and automatic wells were installed to record daily water table depth (Sun and others 2000). Thus, three wetland-upland well pairs with automatic recording water table data were used for MIKE SHE model calibration and validation. Among the six automatic recording wells, the three wells that were located in wetlands had data starting on January 23, 1992. However, the other three upland wells did not have data until May 01, 1993. All six automatic wells had data till the end of December 1996. Thus, additional three upland manual wells were chosen for model calibration and validation. The three manual wells had bi-weekly data during the period of February 02, 1992 to June 22, 1995.

The model was calibrated against water table data for a wet year (1992) and a dry year (1993) to cover a wide range of water table fluctuation conditions. Compared to the long-term annual average precipitation at the site (precipitation = 1330 mm in a normal year), the wet year had a surplus of 170 mm of precipitation while the dry year had the deficit of 230 mm (Sun and others, 1998b). The rest of the water table data (1994-1996) that represented a year with dry spring (1994) and two normal years (1995-1996) in terms of total annual precipitation were used for model validation.

Model Application Scenarios

After model calibration and validation were conducted, the MIKE SHE model was applied to simulate four scenarios. These scenarios included: 1) Base line (BL); 2) Clear Cutting (CC); 3) An average annual temperature increase of 2 °C (TI); and 4) An average annual precipitation decrease of 10 percent (PD). The purpose of these scenarios was to examine water table responses to land management and climate change.

BL scenario was based on the historically climatic data with the assumption that the watershed remained all forested throughout the study period. CC scenario represented a simple forest management practice that was also based on the historically climatic data but with the assumption that the entire watershed was clear-cut (LAI = 0.1 for uplands and LAI = 0.4 for wetlands (Clark and others 2004, Gholz and Clark 2002)). The last two scenarios represented two simple climate change cases. A temperature increase of 2 °C scenario represented the situation that every daily temperature was increased 2 °C with land cover remained the same as base line. The 10 percent precipitation

decrease scenario represented a case that daily precipitation decreased 10 percent when there was a rainfall event, while the land cover remained the same as the base line.

RESULTS AND DISCUSSIONS

Model Calibration and Validation

As mentioned earlier, nine wells were used for model calibration and validation. An example of the model calibration and validation results was presented at an automatic well pair in NW block (Figure 2, Figure 3, Figure 5, and Figure 6). The automatic wells had daily measurements except during some dry periods when the water table was lower than measurement range.

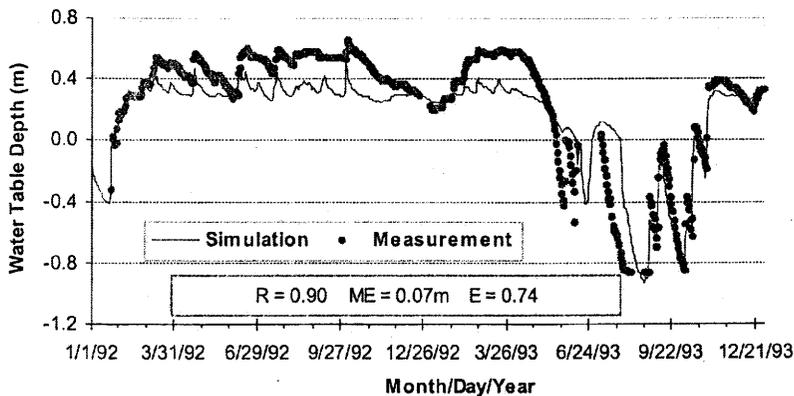


Figure 2--MIKE SHE model calibration at the automatic well in a NW block wetland during 1992-1993.

Generally, the correlations between measurements and simulations ranged from 0.81 to 0.94 with ME values in a range of -0.34 m to 0.07 m and E values within -0.87 to 0.77. The calibration results showed that MIKE SHE could capture the temporal dynamics of water table variations as demonstrated by the high R values at this typical flatwoods site (Figure 2 and Figure 3). During the wet year of 1992, water tables were very close to the ground surface and water was ponded on the wetlands during most time of that year (Figure 2). On the uplands, water table periodically reached the ground surface but did not have ponded surface water as wetlands due to the higher elevations (Figure 3). During the dry periods (June - October) in 1993, water table was much lower to the ground surface and most of the time there was no surface water on the wetlands. Overall, water table was closer to the ground surface in wetlands than in uplands under the dry conditions.

In general, the model underestimated water table depth in wetlands during wet periods but overestimated it in uplands during dry periods (Figure 2 and Figure 3). Discrepancies were more apparent in storm events when water table rose after a long drought period (i.e. 19993).

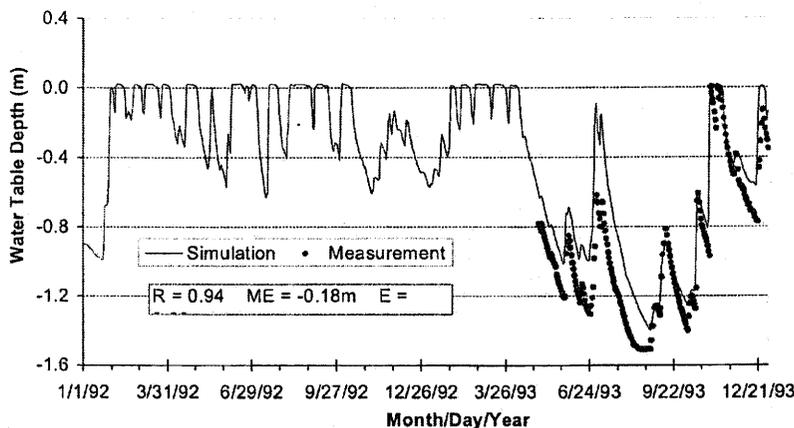


Figure 3--MIKE SHE model calibration at the automatic well in a NW block upland during 1992-1993.

In addition to comparing temporal water table dynamics at nine individual locations as described above, spatial discrepancies between simulated and measured water table depths were also examined. One normal day with water table neither too high nor too low was selected. The date October 29, 1992 was chosen because it had moderate water table depth with complete measurements available over the landscape. A total of 123 wells had water table data on October 29, 1992. With the ESRI ArcView 3.3 software, the point features of these water table data were interpolated to create a grid surface using the method "IDW, nearest of neighbors, Numbers of neighbors: 3, power: 1, No barriers".

Simulated spatial water table data on October 29, 1992 were converted from the MIKE SHE text format to an Arcview grid format. Then this grid was imported into ESRI ArcView 3.3. Water table differences were defined as the interpolated measured water table depths subtracted the simulated water table depths. The grid subtraction calculations were performed in Arcview 3.3 (Figure 4).

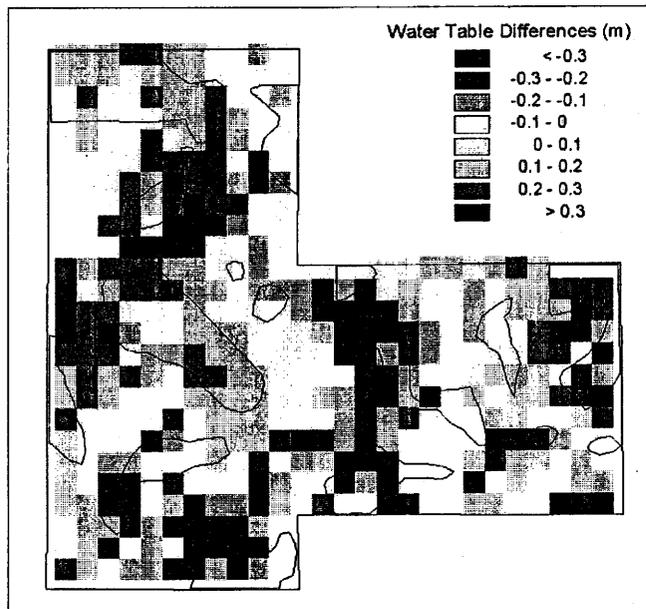


Figure 4--Spatial water table depth differences of measurements and simulations of water table depth on 10-29-1992 (30 m cell size; negative values indicate overprediction while positive values indicate

Across the entire site, MIKE SHE underestimated water table depth at some locations (i.e. wetlands) and overestimated water table depth at the other locations (uplands). Generally, on October 29 1992, MIKE SHE underestimated water table depth at most of the wetlands, but overestimated water table depth at most of the uplands (Figure 4). The spatial inspection confirmed the general pattern observed during the comparisons of nine individual wells. The discrepancies might be largely caused by inadequate representation of land topography at this flat landscape. The modeling cell size was set as 30 m. Thus, micro-landscape variations were likely not represented adequately in the modeling system. Well data were measured at particular points in the field, while the model simulated water table on a scale of 30 m x 30 m unit area. Furthermore, other spatial information was not likely represented adequately either, such as soil depth and soil hydraulic parameters.

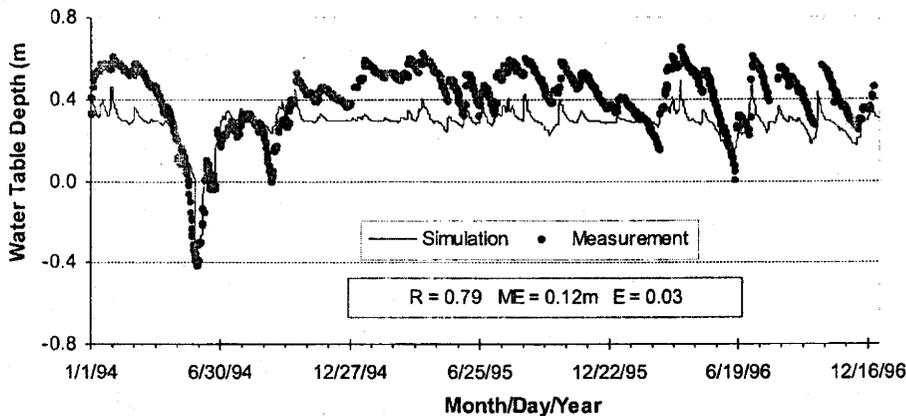


Figure 5--MIKE SHE model validation at the automatic well in a NW block wetland during 1994-1996.

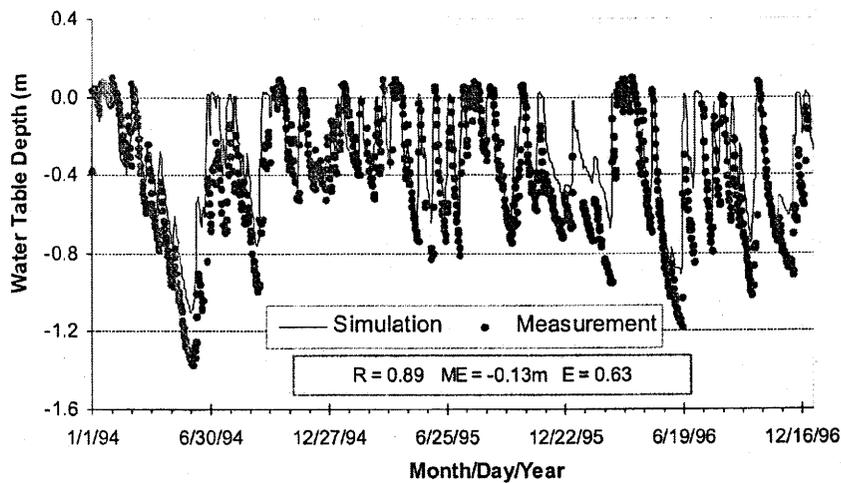


Figure 6--MIKE SHE model validation at the automatic well in a NW block upland during 1994-1996.

Frequency analysis of the spatial discrepancies indicated that water table differences between measurements and simulations were in a normal distribution. There were about 38 percent of differences in the range of (-0.1 m, 0.1 m), 70 percent in the range of (-0.2 m, 0.2 m), and 90 percent in the range of (-0.3 m, 0.3 m). Overall, the discrepancies were in a reasonable error range.

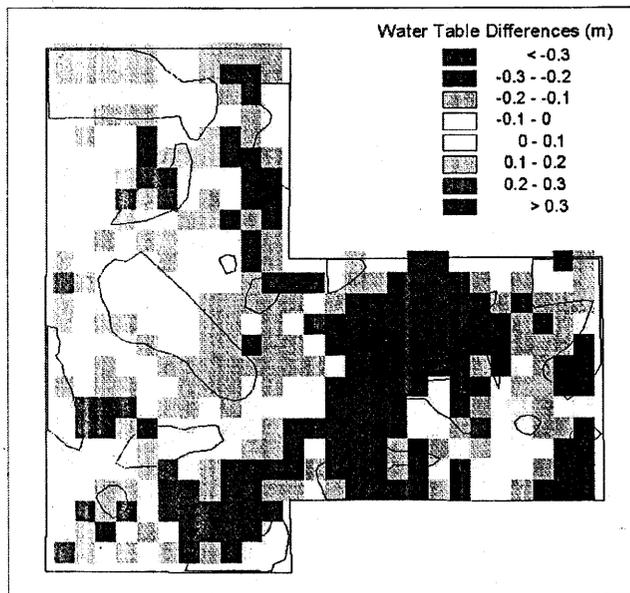


Figure 7--Spatial water table depth differences of measurements and simulations on 03-11-1994 (30 m cell size; negative values indicate overprediction while positive values indicate underprediction).

Similarly, both temporal and spatial water table data collected from the study site were used to validate the MIKE SHE model. The model validation indicated the promising applicability of MIKE SHE at this flatwoods site (Figure 5 and Figure 6). Generally, there were high correlations between measured and simulated water table depths at these nine various locations. R values varied from 0.73 to 0.91. ME values were in the range of -0.29 m to 0.15 m. E values ranged from -1.23 to 0.69. The validation showed that MIKE SHE performed reasonably well for the three years (1994-1996), especially for uplands. Similar model performances were achieved as those in the calibration. In general, the model underestimated water table depth in wetlands during wet periods.

Similar to the calibration, a normal day - March 11, 1994 - with moderate water table depth conditions was selected to examine the spatial water table depth discrepancies between measurements and simulations for the validation purpose (Figure 7). Again, with similar model performances in the calibration, MIKE SHE underestimated water table depth at some locations and overestimated it at the other locations. Generally, on 03-11-1994, MIKE SHE tended to overestimate water table depths across the flatwoods site.

Frequency analysis of the spatial differences confirmed that MIKE SHE mostly overestimated water table depth on 03-11-1994, because 73 percent of the discrepancies were less than zero. There were about 39 percent of the differences in the range of (-0.1 m, 0.1 m), 68 percent in the range of (-0.2 m, 0.2 m), and 85 percent in the range of (-0.3 m, 0.3 mm). Overall, the discrepancies were in a reasonable error range.

As a physically-based, distributed hydrologic model, MIKE SHE was able to simulate the interactive horizontal groundwater flow patterns (Figure 8, Figure 9 and Figure 10) between the wetlands and their surrounding uplands at this typical flatwoods landscape in Florida. The model identified three types of horizontal groundwater flow patterns: (1) Groundwater flows through the wetlands (Figure 8); (2) Groundwater flows into the wetlands (Figure 9); and (3) Groundwater flows out of the wetlands (Figure 10). Thus, the model simulations confirmed field observed groundwater flow patterns at this study site (Crowover and others 1995).

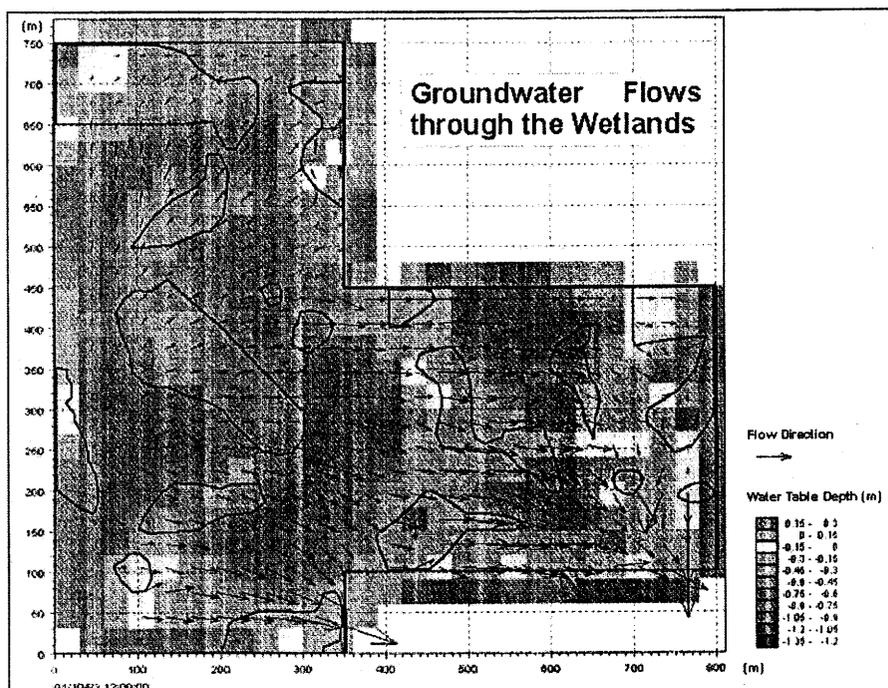


Figure 8--MIKE SHE simulated the flow pattern of groundwater flowing through wetlands on 01-10-1993 (30 m cell size; water table depth was defined as the distance of the water table to the ground surface; it was negative or positive when the water table was below or above the ground surface).

On 01-10-1993, groundwater generally flowed through the wetlands in response to the general landscape topographic gradients (Figure 8). Groundwater was transmitted through the wetlands from the uphill side toward the downhill side. This flow pattern usually occurs during the wet periods and dry periods.

On 02-27-1993, a wet day, groundwater flowed from the surrounding uplands to the wetlands (Figure 9). Wetlands were recharged and served as the sinks of water from uplands. This flow pattern usually occurs under very wet conditions when the water table is high.

On 04-25-1993, a day of transition from wet to dry, groundwater flowed away from wetlands toward the surrounding uplands as indicated by the flow direction (Figure 10). This is most obvious for K wetland. Wetlands served as the sources of water to the uplands. This flow pattern usually occurs during the transition periods between wet and dry conditions.

The horizontal groundwater flow patterns were essentially governed by hydraulic gradients across the study site. During very wet periods when water table were both high in uplands and wetlands, hydraulic heads in uplands were higher than the heads in wetlands. Driven by hydraulic gradient, water moved from uplands into wetlands (Figure 9). This was a low-relief landscape. Water that moved into wetlands was mainly limited to the uplands adjacent to the wetlands rather than from the entire surrounding upland areas. This flow pattern was not common at this study site. Water mainly either flowed through the wetlands or flowed from wetlands toward the uplands (Crowner and others 1995). When water table dropped lower, the general landscape topographic gradient played a dominant role in the hydraulic gradient. Thus, groundwater flowed through the wetlands from the uphill side toward the downhill side (Figure 8). When water table dropped further during the transitions from wet to dry conditions, water table dropped faster in uplands than in wetlands and resulted in higher water table in wetlands than in uplands. This was most likely resulted from the differential storage of water as surface water in wetlands and soil water in uplands as well as differential transpiration between vegetations in the wetlands and the uplands. Thus, hydraulic heads were higher in wetlands than in uplands. Driven by hydraulic gradient, groundwater flowed away from the wetlands toward the surrounding uplands (Figure 10). However, during dry periods when differential water storage in wetlands and in uplands was small, the general landscape topographic gradient dominated groundwater movement and resulted in the flow pattern of groundwater flowing through the wetlands (Figure 8).

These flow patterns were observed in the field (Crowner and others 1995). However, the advantages of model simulations enabled us to examine the flow in more intensive time steps and spatial scales. Furthermore, model simulations could help quantify the flow between the wetlands and the uplands.

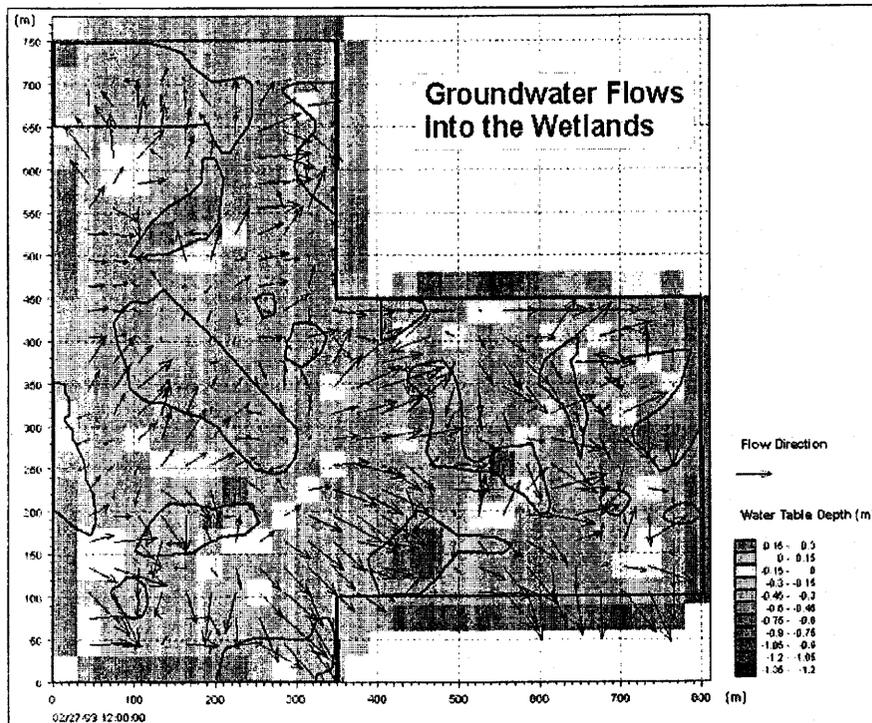


Figure 9--MIKE SHE simulated the flow pattern of groundwater flowing into wetlands on 02-27-1993 (30 m cell size; water table depth was defined as the distance of the water table to the ground surface; it was negative or positive when the water table was below or above the ground surface).

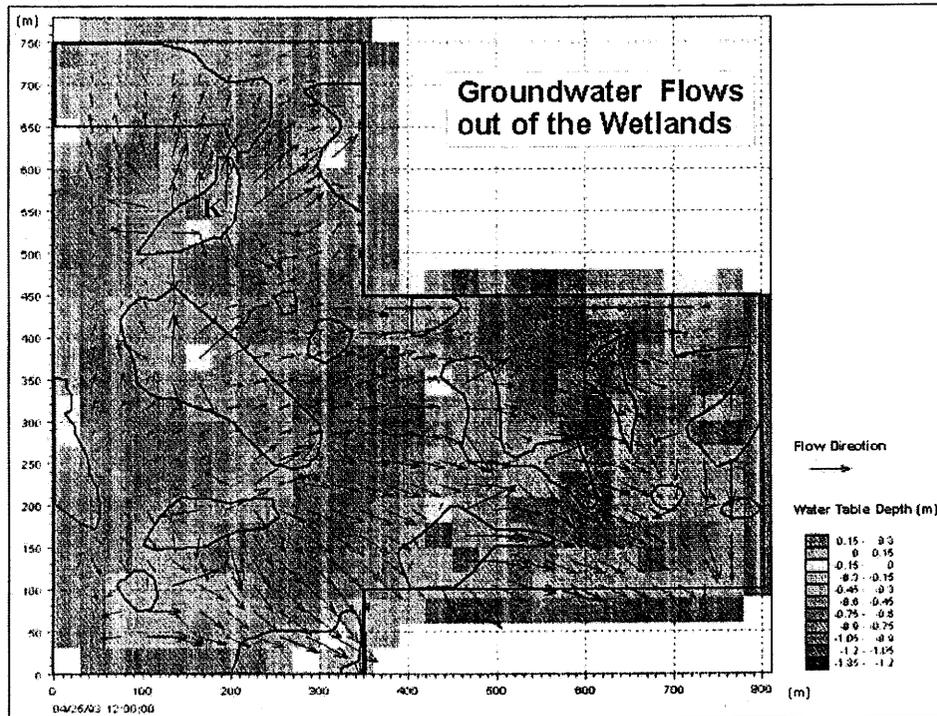


Figure 10--MIKE SHE simulated the flow pattern of groundwater flowing out of wetlands on 04-25-1993 (30 m cell size; water table depth was defined as the distance of the water table to the ground surface; it was negative or positive when the water table was below or above the ground surface).

Model Applications

Using the validated model, we evaluated the potential effects of three hypothetical scenarios on the groundwater table during 1992-1996 (Figure 11). In terms of water table variations, modeling results suggested that forest clearcut and climate change did not have much impact during the wet periods. However, clearcut and climate change would affect water table during the dry periods (Figure 11). In the dry year 1993, the water table elevation was higher under the clearcut scenario than the base line. When temperature increased 2 °C or precipitation decreased 10 percent, water table dropped deeper than the base line scenario. This pattern could also be seen in the dry spring season of 1994.

Under the clear-cutting situation, ET was reduced due to the lack of leaves. Thus, it caused the rise of the water table. This was especially true during dry periods. When temperature was increased 2 °C, ET was increased due to the increase in driven energy and higher PET, thus water table dropped further. Under the 10 percent precipitation decrease scenario, the water input was less than the base line condition and recharge was decreased, thus the water table was lower. During the wet periods, the water table was close to the ground surface or even reached the ground surface. Water was sufficient for plant water use and satisfied the PET demand for most of the time. Thus, the impacts on water table level from forest management and climate change were minimal. However, during the dry periods when water table was well below the ground surface, the impacts were stronger and the effects were showed on the differences of the water table levels.

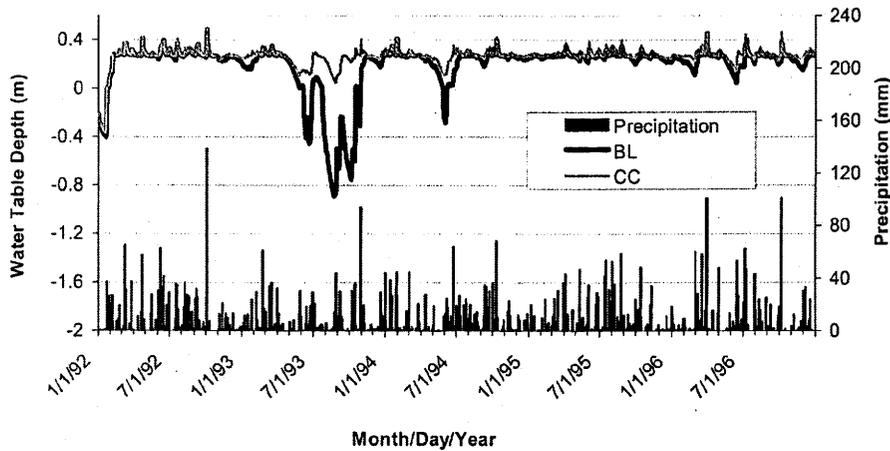


Figure 11 a

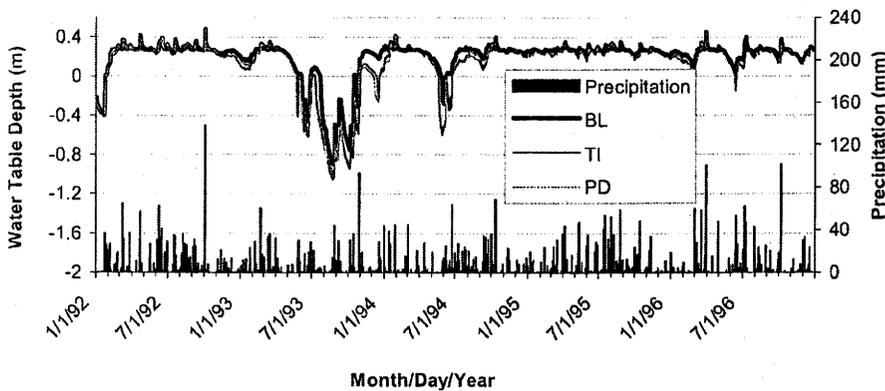


Figure 11 b

Figures 11 a and 11 b--Impacts of clearcut and climate change on water table at the automatic well in a SW block wetland during 1992-1996 (BL - base line; CC - clearcut; TI - 2 °C temperature increase; PD - 10 percent precipitation decrease. For the CC case, a 30 percent reduction of potential ET was assumed (Grace and Skaggs 2006, G. Sun Unpublished data)).

CONCLUSIONS

This study showed that MIKE SHE could simulate flatwoods hydrology with reasonable accuracy. MIKE SHE identified three horizontal groundwater flow patterns at this study site. The model simulations confirmed field observations of interactions of wetland surface water and groundwater in their surrounding uplands on a typical flatwoods landscape. MIKE SHE had the advantages in presenting temporal flow dynamics spatially across the landscape. Simulations of the four application scenarios indicated that forest removal and climate change would have impacts on the groundwater table dynamics during the dry periods. However, the impacts on groundwater table depth might be minimal under the wet conditions.

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Fire Effects on Watersheds



FIRE EFFECTS ON WATER QUALITY: A SYNTHESIS OF RESPONSE REGULATING FACTORS AMONG CONTRASTING ECOSYSTEMS

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Abstract—The key components of watershed processes are inputs in precipitation, interactions of vegetation, soil and water including evapotranspiration (water yield), overland flow (erosion), and storage and filtering (nutrients), and outputs in streamflow. Fire effects occur at the vegetation-soil interface and can result in altering overland flow and infiltration rate of water. Fire can affect infiltration rates by collapsing soil structure and reducing soil porosity, contributing ash and charcoal residues which can clog soil pores, and raindrop splash can compact soil and further contribute to loss of soil porosity. An extreme example is the development of hydrophobic soils as observed in the western U.S. following severe wildfire. Watershed responses to fire depend on intensity and severity. Many factors influence fire severity including the quality and quantity of fuels, soil properties, topography, climate, and weather. The most important factors influencing the response to fire are vegetation mortality and the loss of the forest floor which are directly proportional to fire severity. Vegetation mortality reduces nutrient and water uptake, soil stability with root death, and the litter source for forest floor replenishment. The forest floor litter and humus (duff) layers provide soil cover, act as a sponge, and enhance infiltration. Large storm events immediately after a fire can accelerate surface runoff and compact soil.

INTRODUCTION

Wildland fire has the potential to significantly impact hydrologic processes such as surface runoff, sediment yield, and sediment and nutrient transport to streams. The magnitude and duration of watershed responses to fire depends on the interactions among burn severity, post-fire precipitation regime, topography, soil characteristics, and vegetative recovery rate. The typical impact of fire is an immediate change in vegetative cover, forest floor surface, physical properties of the soil, followed by mid- and long-term changes in biological pools and nutrient cycling processes. Vegetation and litter protect the soil against the forces of erosion by maintaining high infiltration rates and low levels of overland flow (Covert and others 2005). Vegetative cover and forest floor are the primary drivers of sediment responses to fire. Large reductions of vegetative cover, particularly the ground vegetation and forest floor, leave the soil prone to raindrop impact and reduce rainfall infiltration and storage so that erosive overland flow tends to occur more readily (Shakesby and Doerr 2006). Nutrient responses are also impacted by changes in vegetation and forest floor, as well as changes in biological processes that regulate cycling processes.

The magnitude and duration of hydrologic and water quality responses vary greatly across ecosystem types in the continental U.S. As such, it has been difficult to generalize response or apply knowledge derived from one region of the U.S. to any other. Over the past several years, a growing body of research has provided hydrology/water quality response data across a range of ecosystems, fire types, soils, and climate regimes. In this paper, we synthesize current knowledge on factors regulating water quality in contrasting ecosystems.

Defining Fire Severity

Fire severity depends on the interaction between fire intensity (rate at which thermal energy is produced) and duration (length of time burning occurs at a particular point) and describes the magnitude of the disturbance and reflects the degree of change in ecosystem processes (Neary and others 2005). Fire severity is a qualitative measure of the effects of fire on site and soil resources; it can occur along a spectrum from high to low or can be described as a patchwork, mosaic, matrix or mixed-severity event. Debano and others (1998) describe a light severity burn as one that burns only surface fuels, leaves the soil covered with partially charred organic material, and little to no duff consumption (fermentation (Oe) + humus (Oa) layers). A moderate-severity burn results from a large proportion of the organic material burned away from the surface of the soil and the remaining fuel is deeply charred. A high-severity burn results from all of the organic material burned away from the soil surface, organic material below the surface is consumed or charred. Fire severity has been assessed by numerous methods such as degree of destruction of aboveground live and dead biomass (Neary and others 2005), amount of forest floor consumed, particularly the duff layer, or heat penetration into the mineral soil (Swift and others 1993).

Precipitation Regime

After fire, rainfall intensity and duration can influence the amount of sediment delivered to a stream channel. The detachment of soil particles by rainsplash or overland flow and their transfer downslope are sensitive to modifications in land surface properties caused by fire (Johansen and others 2001, Sakesby and Doerr 2006). In low rainfall ecosystems, surface runoff and erosion may not be observed if there is a long period of post-fire recovery before the first rainfall event. Even in ecosystems with low mean annual rainfall, a high-intensity rainstorm immediately after wildland fire can create runoff that alters the topography of the hillslope, which subsequently impacts stream channels. Rainstorm events need to have enough energy to transport sediment. Swift and others (1993) determined that rainfall events of $>50 \text{ mm hr}^{-1}$ were required to transport material after a fell-and-burn prescribed fire in the southern Appalachians. Sediment yields are typically higher in the first year after burning, especially when the burned watershed has been exposed to high-intensity rainfall events immediately after the fire has exposed the soil surface. Some of the largest increases in surface runoff have been observed where short-duration, high intensity convective rainstorms occur. For example, after the 1996 Buffalo Creek Fire in Colorado, two short-duration, high-intensity rainstorms ($\sim 90 \text{ mm hr}^{-1}$) removed ash from the hillslopes, rilled the hillslope surfaces, channeled subtle drainages, which led to a headward extension of the channel network, and deposited sediment in stream channels (Moody and Kinner 2006). Kunze and Stednick (2006) found that rainfall intensity explained more than 80% of the variability in sediment yields. After the 2000 Bobcat Fire in Colorado, a single storm with 30 min rainfall intensity of 42 mm hr^{-1} resulted in 370 kg ha^{-1} and 950 kg ha^{-1} sediment yields, on treated (erosion-control with contour log felling, grass seeding, and mulching) and untreated watersheds, respectively (Kunze and Stednick 2006).

Vegetation Recovery

Post-fire soil erosion amounts vary not only with rainfall but also with burn severity, topography, soil characteristics and amount of vegetative recovery. Under moderate to severe fire severity that removes vegetation and forest floor cover, transpiration, interception and surface storage capacity for rain are temporarily reduced. Conversely, any fire-induced alterations to storage capacity and water repellency will decline as vegetation and ground litter recover. Ground cover protects the soil from raindrop impact and offers resistance to overland flow. Vegetation recovery rates are strongly affected by fire size and severity, post-fire erosion events and vary by climate and geographic area. Rapid vegetation establishment has been regarded as the most cost-effective method to promote water infiltration and reduce hillslope erosion (Robichaud 2005). In the western U.S., land management agencies have spent tens of millions of dollars on post-fire emergency watershed stabilization measures to minimize flood runoff, onsite erosion, offsite sedimentation, mud and debris flows, and other hydrologic damage to natural habitats (Robichaud 2005). Post-fire hillslope rehabilitation treatments include seeding for vegetative re-growth, ground covers or mulches, and barrier and trenches that physically hold runoff and sediment. In the eastern U.S., such costly and dramatic post-fire rehabilitation efforts are typically not required. Even after severe fire, recovery rates of southern Appalachian watersheds are much faster than western forests due to rapid vegetative re-growth (Clinton and Vose 2000, Elliott and others 1999).

Low severity burning, such as prescribed fires, can promote a herbaceous flora (Elliott and others 1999, Gilliam 1988, Hutchinson and others 2005) increase plant available nutrients (Elliott and others 2004), and thin-from-below over-crowded forests. While large, severe fires can cause changes in successional rates, alter species composition, generate volatilization of nutrients and ash entrainment in smoke columns, produce rapid or decreased soil mineralization rates, and result in subsequent nutrient losses through accelerated erosion (Neary and others 1999).

Surface Runoff and Erosion

Wildland fires are often landscape-scale disturbances that can alter the hydrologic and erosion responses of catchments. Erosion can occur when ground cover is reduced or consumed, and subsequently infiltration rates are reduced, i.e., water repellency is high. Fire-induced or enhanced soil water repellency (hydrophobicity) is commonly viewed as a key contributor to the substantial increases in hillslope runoff and erosion observed following severe wildfire (Debano and others 2005, Doerr and others 2006, Huffman and others 2001), particularly in the western U.S. Soils do not all exhibit the same degree of water repellency; a water-repellent soil is classified as one on which a drop of water will not spontaneously penetrate. Water drop penetration time (WDPT) has been used extensively to characterize soil water repellency (Letey 2001). Several factors associated with fire, such as removal of surface litter and higher raindrop impact, would produce higher runoff and erosion from burned compared with unburned catchments, independent of water repellency. High runoff and erosion occurs from the combined effects of canopy

destruction and water repellency induced by fire (Letey 2001), typically higher water repellency results from high severity fires (Lewis and others 2006).

Sediment yields, in the first year after fire, range from very low in flat terrain without major rainfall events, to extreme in steep terrain affected by high-intensity thunderstorms. In the first post-fire year, sediment yield can vary from 0.01 to more than 110 Mg ha⁻¹ year⁻¹ (Robichaud and others 2000). High-intensity rainstorms after wildfire can create runoff that alters the topography of the hillslope, which subsequently impacts stream channels. In the coastal plain region of the Southeastern U.S., surface runoff and erosion from forested land would be minimal because the terrain is flat. On steep mountain slopes, Hendricks and Johnson (1944) found that sediment yield ranged from 71 Mg ha⁻¹ year⁻¹ on 43% slopes, 202 Mg ha⁻¹ year⁻¹ on 66% slopes, and 370 Mg ha⁻¹ year⁻¹ on 78% slopes after a wildfire in mixed conifer forests of Arizona. After the 1998 North 25 Fire in north-central Washington, Robichaud and others (2006) reported a first year mean erosion rate of 16 mg ha⁻¹ yr⁻¹ (Table 1), and this decreased significantly in the second year to 0.66 Mg ha⁻¹ yr⁻¹. Mean canopy cover (percent cover provided by live plants) was 18% the first year and 53% the second year after the wildfire. Total precipitation was below average during the four-year period of their study (Robichaud and others 2006), and most erosion occurred during short duration, moderate intensity summer rainfall events. In the southern Appalachian mountain region, terrain is steep and rainstorms events with enough energy to transport sediment (≥ 50 mm hr⁻¹) have been recorded (Swift and others 1993), but vegetative recovery is rapid minimizing hillslope erosion.

Table 1. Sediment losses the first year after prescribed (Rx) fire and wildfires.

Location	Community	Severity/activity	1 st year sediment loss (Mg ha ⁻¹)	Reference
North Carolina, Mountains	Pine/hardwoods	Low severity, fell-and-burn	0.087	Swift and others 1993
South Carolina, Piedmont	Pine/hardwoods	Low severity, Rx site preparation burn	0.137	Robichaud and Waldrop 1994
		High severity, Rx site preparation burn	5.748	
South Carolina, Piedmont	Loblolly pine	Control	0.027	Van Lear and others 1985
		Low severity, Rx understory burn	0.042	
		Moderate severity, Cut + Rx burn	0.151	
Arkansas, Foothills	Shortleaf pine	Control	0.036	Miller and others 1988
		High severity, slash cut + Rx site preparation burn	0.237	
East Texas, Foothills	Loblolly pine	Clearcut + Herbicide + Rx site preparation burn	0.885	Field and others 2005
		Clearcut + Mechanical tillage + Rx site preparation burn	1.273	
Colorado Front Range	Mixed conifer	Low to moderate severity, Rx fire	0.20 to 0.05	Benavides-Solorio and MacDonald 2005
		High severity, wildfire	2.0 to 10.0	
Colorado Front Range	Mixed conifer	Unburned hillslopes	0.30	Moody and Martin 2001 Wagenbrenner and others 2006
		High severity, wildfire	6.20	
North-central Washington	Subalpine fir	High severity, wildfire	16.0	Robichaud and others 2006

In the western U.S., erosion rates increase by several orders of magnitude from areas burned at high severity because of the loss of protective ground cover and increase in surface runoff (Benavides-Solorio and MacDonald 2001, 2002, Robichaud and others 2000). In the Colorado Front Range, highest mean sediment rates were 80 – 100 Mg ha⁻¹ from plots burned at high severity in recent wildfires (Benavides-Solorio and MacDonald 2005). The percentage of bare soil explained most of the variability in sediment yields (Benavides-Solorio and MacDonald 2001). Johansen and others (2001) found that post-fire sediment yield increased non-linearly as percent bare soil increased. Specifically, sediment yields increased little when percent bare soil varied from 0 up to 60%, then yield increased exponentially above 60% bare soil. Ground cover effects appeared to be more important in explaining hydrologic response than either surface roughness or slope (Johansen and others 2001). Hence, maintaining vegetation cover or a cover of forest floor organic layers on the soil surface is the best means of preventing excessive soil erosion rates (Debano and others 1998, Neary and Ffolliott 2005, Wagenbrenner and others 2006).

In the southeastern U.S., several authors have reported little to no soil erosion after light- to moderate-intensity fires (Neary and Currier 1982, Shahlee and others 1991, Van Lear and Danielovich 1988, Van Lear and Waldrop 1986, Swift and others 1993). For example, Douglas and Van Lear (1983) found no significant differences in runoff or soil export between burned and unburned watersheds in the Piedmont of South Carolina. Swift and others (1993) reported only minor and localized movements of burned plant fragments and soil after a fell-and-burn treatment in xeric pine-hardwood stands in the southern Appalachian Mountains of North Carolina. In their study, soil erosion was minimal primarily because the forest floor remained largely intact; i.e., duff consumption ranged from 30 to 67 percent (Swift and others 1993). Overall, these fires were classified as high intensity and low to moderate severity. Severity was moderate on portions of the burn where topography increased the fire intensity, causing greater proportions of forest floor consumption in small patches (Swift and others 1993). Effects were severe in a few spots where ribbons of soil were exposed after partially decomposed logs in contact with forest floor ignited and smoldered until consumed. After the burns, the bare soil exposure ranged from 7 to 14%. Where soil was exposed, the material was trapped within a short distance by residual forest floor and wood debris; thus, only two of eight sediment traps collected transported material resulting in < 0.10 Mg ha⁻¹ sediment lost (assuming a bulk density of 1.2 Mg ha⁻¹ and 40% charcoal by volume) the first year after the fires (Table 1). Sediment deposited at the lower margins of the study areas was transported by only three rainfall events that had enough force (> 50 mm hr⁻¹) to move sediment. Thereafter, no further sediment was lost because subsequent rainfall events were not of sufficient magnitude to transport material. In their study, the residual forest floor was resistant to erosion over the range of burn intensities and sediment was prevented from leaving the site by unburned brush and undisturbed forest floor at the lower margins of the burned areas (Swift and others 1993).

In the Piedmont region of South Carolina, Robichaud and Waldrop (1994) calculated sediment yields for low- and high severity site preparation burns in pine/hardwoods. For low severity fire (7% bare soil), sediments yields were 13.6 kg ha⁻¹ mm⁻¹ during simulated intense, rainfall (100 mm hr⁻¹ rain event lasting 30 min) with a total annual sediment loss of 0.137 Mg ha⁻¹ year⁻¹ under natural rainfall events; and for high severity fire (63% bare soil), sediment yields were up to 27.7 kg ha⁻¹ mm⁻¹ during simulated intense, rainfall with a total loss of 5.75 Mg ha⁻¹ year⁻¹ under natural rainfall events (Table 1). In loblolly pine forests in South Carolina, Van Lear and others (1985) reported 0.042 Mg ha⁻¹ yr⁻¹ and 0.151 Mg ha⁻¹ yr⁻¹ sediment loss from understory burn and burn + cut sites, respectively (Table 1). Field and others (2005) estimated annual soil losses of 1.273 and 0.885 Mg ha⁻¹ year⁻¹ from mechanical tillage and prescribed fire, respectively.

Stream Suspended Sediment

Severe wildfires can cause damage to plant cover and, thus, increase streamflow velocity, sediment delivery to streams, and stream water temperatures, as contrasted to low severity, cool-burning prescribed fires, which have less severe consequences (Reardon and others 2005). If surface erosion via overland flow reaches stream channels, then stream sediment concentrations increase proportional to the sediment delivered. Excess sediment is the principal pollutant of stream water associated with forest management (Phillips and others 2000) and is considered the primary threat to the integrity of aquatic resources (Henley and others 2000). After fire, excess sediment delivery to streams typically occurs after a measurable storm event. Watersheds severely denuded by fire are vulnerable to accelerated rates of soil erosion. While many fires increase sediment transport, wildfire often produces more sediment than prescribed fire (Debano and others 1998). Generally, prescribed fires, by their design, are not intended to consume extensive layers of forest floor litter. Without sediment transport via overland flow or surface runoff, input of sediment to streams would be minimal following prescribed fire or wildfire. If the forest floor

remains intact and little to no bare soil is exposed, there is no mechanism for long-distance transport of sediment to streams (Vose and others 1999), regardless of rainfall event.

In the western U.S., suspended sediment concentrations in streamflow can increase to very high levels following severe fire. For example, Hauer and Spencer (1998) found that stream sediment concentrations increased from 3.0 mg L⁻¹ before fire to 32.0 mg L⁻¹ after and Fredriksen (1971) recorded an increase from 2.0 mg L⁻¹ before disturbance to 150 mg L⁻¹ following clearcut + slash burn. In contrast, fire in the southeast and southern Appalachians typically does not create conditions that result in sediment delivery to streams (Table 2). Forested streams in the southern Appalachians with high TSS during storm events are usually influenced by roads or land-use conversion (Table 2).

Table 2. Total suspended solid (TSS) concentration in headwater streams with varying disturbance types and severity.

Location	Community	Severity/activity	TSS (mg L ⁻¹)	References
North Carolina, Mountains	Mesic hardwoods	Low severity, prescribed fire	1-11	Vose, unpublished
East Tennessee and North Georgia, Mountains	Pine/hardwoods	Low severity, prescribed fire	1-6	Elliott and Vose 2005
South Georgia, Coastal Plain	Mixed oak-pine	Military training using tracted vehicles, <7% catchment area disturbed (low severity)	4 (baseflow) 57-300 (stormflow)	Houser and others 2006
		>7% catchment area disturbed (high severity)	10 (baseflow) 847-1881 (stormflow)	
North Georgia, Mountains	Mixed hardwoods	Roads, land-use conversion	1-10 (baseflow) >100 (stormflow)	Riedel and others 2003
W. Oregon Montana	Douglas-fir	Clearcut, slash burn	150	Fredriksen 1971 Hauer and Spencer 1998
	Mixed conifer	Wildfire	32	

For prescribed fires in southern Appalachian pine-hardwoods, Elliott and Vose (2005) found no significant differences in total suspended solids (TSS) concentrations between burn and control streams over a 10-month post-burn sampling period (Table 2). Several factors were attributed to the explanation of their results; a small rain event did occur the first day after the burn treatments, but this event brought less than 15 mm of rainfall, the low intensity-low severity, prescribed fire consumed less than 20% of the forest floor mass, and the burns were in the spring when vegetative re-growth occurs. With other disturbance types and intensities, other researchers have found more distinct and larger increases in sediment concentrations in highly disturbed streams (Houser and others 2006, Webster and others 1990) than undisturbed streams during storm events (Table 2). For example, Houser and others (2006) investigated a range of disturbance intensities for typical low-gradient, Southeastern Coastal Plain streams to illustrate the impact of upland soil and vegetation disturbance on stream sediments. In catchments with a disturbance intensity of <7%, the mean maximum change in TSS ranged from 57 to 300 mg L⁻¹ during storm events. In catchments with a disturbance intensity of >7%, mean maximum change in TSS ranged from 847 to 1881 mg L⁻¹ during storm events (Table 2).

Stream Nitrogen

The potential for increased NO₃⁻-N in streamflow after burning is attributed mainly to accelerated mineralization and nitrification (DeBano and others 1998, Knoepp and Swank 1993, Vitousek and Melillo 1979) and reduced plant uptake (Vitousek and Melillo 1979). Several studies on effects of prescribed fire on streamwater quality (Bêche and others 2005, Clinton and others 2003, Douglas and Van Lear 1983, Elliott and Vose 2005, Field and others 2005, Richter and others 1982, Vose and others 1999), have found little to no detectable changes in streamwater chemistry after burning. For the few cases where a measurable increase in NO₃⁻-N was detected, timing of wildland fire influenced NO₃⁻-N delivery to streams. In the spring, less NO₃⁻-N will be transported to streams when vegetation uptake and microbial immobilization are typically high, compared to burns in the fall when vegetation is dormant.

For example, Clinton and others (2003) compared stream NO_3^- -N responses from watersheds burned in the fall and those burned in the spring. The two sites that showed a stream NO_3^- -N response were burned in the fall, whereas the sites that were burned in the spring showed no response (Table 3).

Table 3. Stream nitrate-nitrogen (NO_3^- -N) responses following prescribed fire (Rx) and wildfire in the southeastern U.S.

Site location	Treatment	Community	Fire severity	Season	NO_3^- -N response (mg L^{-1})	Duration	References
Jacobs Branch, NC	Fell and burn, Rx	Mid-elevation; Pine/hardwood	High intensity, moderate severity	Fall	0.065	30 weeks	Knoepp & Swank 1993
Wine Spring, NC	Restoration, Rx	High elevation; Pine/hardwood	Moderate intensity, low severity	Spring	0	None	Vose and others 1999
Joyce Kilmer, NC	Wildfire	High elevation; old-growth hardwoods	Low intensity, low severity	Fall	0.100	6 weeks	Clinton and others 2003
Hickory Branch, NC	Restoration, Rx	Mid elevation; Pine/hardwood	Moderate intensity, low severity	Spring	0.004	2 weeks	Clinton and others 2003
Conasauga, TN & GA	Understory, Rx	Low elevation; Pine/hardwoods	Low-to-moderate intensity, low severity	Spring	0	None	Elliott & Vose 2005
Robin Branch, NC	Understory, Rx	High elevation; Mesic, mixed oak	Low intensity, low severity	Spring	0	None	Vose and others 2005
Roach Mill, GA	Understory, Rx	Piedmont; pine/hardwoods	Moderate intensity, moderate intensity	Spring	0	None	Vose and others 2005
Uwarrie, NC	Understory, Rx	Piedmont; pine/hardwoods	Moderate intensity, moderate intensity	Spring	0	None	Vose and others 2005
Croatan, NC	Understory, Rx	Coastal Plain; longleaf pine	Low to moderate intensity, low severity	Winter	0	None	Vose and others 2005

Vose and others (2005) compared the effects of low severity prescribed fire in Piedmont and southern Appalachian mountain streams (Table 3). In streamwater, measured NO_3^- -N was extremely low ($<0.1 \text{ mg NO}_3^-$ -N L^{-1}) before and after burning. Both sites were burned in early spring and fires were confined to the understory and forest floor. There was generally no overstory mortality to prevent the rapid vegetation N uptake and immobilization of soil nutrients typical of the spring growth flush. Fires were of low enough intensity to prevent significant overland flow and movement of nutrients off-site via physical changes in hydrologic processes. Vose and others (2005) also used a nutrient cycling model to simulated stream NO_3^- -N response under three fire scenarios: moderate-severity prescribed fire, high-severity prescribed fire, and high-severity wildfire. Only under the wildfire scenario was there a significant increase in stream NO_3^- -N concentrations. Vose and others (2005) attributed this simulated increase to reduced nitrogen uptake since the wildfire simulation included 100% overstory mortality. Under their wildfire scenario, streamwater NO_3^- -N concentrations only reached 0.20 mg L^{-1} even with these extreme fire effects. Unlike low to moderate-severity prescribed fires, large-severe wildfires often result in dramatic increases in stream solutes, which may last for years after the fire (Earl and Blinn 2003, Minshall and others 2001, Spencer and others 2003). For example, Hauer and Spencer (1998) observed stream NO_3^- -N concentrations from 0.12 to 0.30 mg L^{-1} in impacted streams after a wildfire in the Rocky Mountains, which were concentrations >5 fold over those observed in control streams. However, not all prescribed fires are low severity burns. Prescribed fire in the Tharp's Creek 16-ha catchment, Sierra Nevada of California killed most of the younger trees and understory vegetation, and the larger trees were scared, but left alive. Most forest litter was combusted in the fire leaving an ash layer throughout the catchment (Williams and Melack 1997). This prescribed burn in the Sierra Nevada of California resulted in the stream NO_3^- -N concentration briefly exceeding 0.84 mg L^{-1} the first month of streamwater runoff after the fire, then exceeding 1.96 mg L^{-1} three months after the fire. The following spring NO_3^- -N concentrations increased above 1.68 mg L^{-1} , persisted above 0.84 mg L^{-1} for several weeks, then returned to pre-fire conditions for the remaining years after the fire (Williams and Melack 1997). Whereas, pre-fire stream NO_3^- -N concentrations seldom exceeded 0.01 mg L^{-1} .

In a recent national evaluation of forested streams, NCASI (2001) found that NO_3^- -N concentrations for small forested watersheds averaged 0.31 mg L^{-1} (median 0.15 mg L^{-1}), and some streams averaged 10 times that level. In streams draining both mountain and Piedmont regions of the southeast, from a range of fire intensities (from low to high; prescribed fire and wildfire), impacts on inorganic stream nitrogen levels are much lower (Clinton and others 2003, Elliott and Vose 2005, Vose and others 2005) than the average reported from NCASI (2001).

CONCLUSIONS

Hydrologic and water quality responses to fire in the continental U.S. vary considerably. When a wildland fire occurs, the principal concerns for changes in water quality are delivery of sediment and nutrients, particularly nitrate, into the stream channel. Fire managers can influence the effects of prescribed fire on water quality by limiting fire severity, limiting fire size, and avoiding burning on steep slopes. Wildfires are typically larger and more severe consuming more fuel and releasing more nutrients than prescribed fire, which increases susceptibility to erosion of soil and nutrients into the stream. Our synthesis of a wide array of studies from across the U.S. support the following conclusions:

1. Maintaining an intact forest floor and promoting rapid vegetation recovery is critical to minimizing the magnitude and duration of sediment transport (surface erosion), sediment delivery (suspended solids) and subsequent water quality responses,
2. Burned areas are most vulnerable to surface erosion immediately post-fire and during extreme rainfall events,
3. Generally, water quality responses are much lower in the eastern U.S. than the western U.S. due to more moderate topography, lower fire severity, and rapid vegetation recovery

These regional differences emphasize the need for localized assessment and analyses of fire prescriptions, post-wildfire rehabilitation, and associated monitoring efforts.

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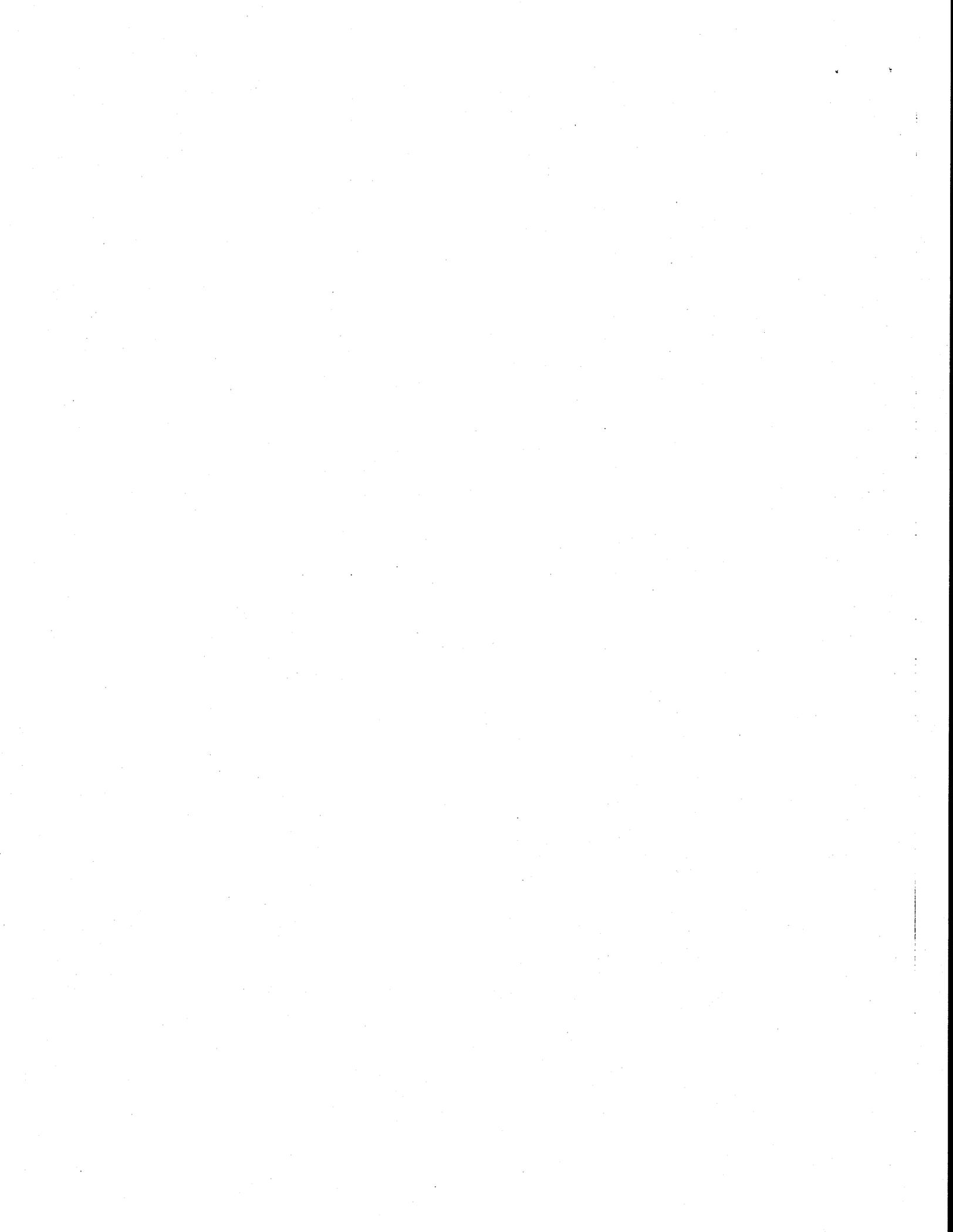
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INITIAL STREAMWATER RESPONSE TO THE HAYMAN FIRE, COLORADO FRONT RANGE

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Abstract--Fire behavior regulates the effects of forest fire on watershed processes. During the 2002 Hayman Fire, high-severity crown fire killed the overstory forest and consumed forest floor on forty percent of the area burned. The extreme fire behavior and its location in watersheds that deliver water to the Denver metropolitan area drew attention to the adverse effects of wildfire on Front Range water quality and supply. A streamwater monitoring network begun prior to the fire allowed pre- and post-fire comparisons of streamwater properties in three burned and three unburned catchments, and post-fire comparisons of ten basins burned to varying extents (0-87 percent). Here we present the response of stream chemistry, temperature and turbidity during the first year following the Hayman Fire. Flow-weighted concentrations of dominant streamwater anions and cations were about 60 percent higher in burned basins during the four months after the fire; within eight to twelve months most chemical attributes were similar between burned and unburned streams. Temperature and nitrate increased immediately after the fire and remained elevated at the end of study period. The proportion of the total area burned and the proportion burned at high severity explained a significant amount of the variation in streamwater nitrate and sediment.

INTRODUCTION

The extent and severity of combustion determine how wildfire alters watershed processes. High-severity burning kills most canopy and understory plants and consumes roots, rhizomes and surface organic matter (Romme and others 2003). This type of combustion can result in widespread change in forest structure and soil conditions and dramatically alter the watershed processes that control streamflow, peak discharge, soil erosion, channel stability, and streamwater nutrient export (Morris and Moses 1987, Spencer and Hauer 1991, Riggan and others 1994, Minshall and others 1997, DeBano and others 1998, Robichaud and others 2003, Neary and others 2005). In contrast, low-severity fire kills few overstory trees and has minimal effect on belowground plant structures, litter layers and watershed conditions. The influence of fire behavior is overlaid upon soil, vegetation and geomorphic features to determine basin-scale consequences of wildfire (Ice and others 2004).

The duration of wildfire effects on aquatic conditions can span hours to centuries (Minshall and others 1989). Immediate, direct effects on stream chemistry relate to the smoke and ash deposited directly into the stream channel during the fire (Tiedeman and others 1978, Earl and Blinn 2003). Exceeding lethal temperatures during burning can also kill aquatic vertebrate and invertebrates (Minshall 2003, Dunham and others 2003). Sustained, second-order watershed effects result from the loss of aboveground structure and subsequent alterations in soil and hydrologic processes. Return of watershed conditions (i.e. stream discharge, temperature, chemical composition, sediment content) to within their pre-fire range follows forest regeneration, typically occurring within a few years or decades (Moody and Martin 2001, Benavides-Solario and MacDonald 2001, Minshall and others 1997). The residual vegetation, litter or forest floor (Pannuk and Robichaud 2003) found after moderate and low intensity fire facilitates watershed recovery (Wagenbrenner and others 2006).

On 8 June 2002, the Hayman Fire was ignited in the Colorado Front Range following a prolonged drought. Precipitation (Western Regional Climate Center 2006) and the water contained in the 2002 snowpack were each about half of long-term annual averages. Low fuel moisture and relative humidity (5-8 percent) and strong, gusty winds (30-80 km hr⁻¹) triggered rapid rates of spread (>3.2 km hr⁻¹) and long-range spot fires (Finney and others 2003). The dense, continuous horizontal and vertical fuel structure permitted burning to advance for 28 days prior to containment. High severity crown fire killed the overstory forest and consumed forest floor across forty percent of the area burned.

The 558 km² Hayman Fire was the largest fire in Colorado history. Its location, 75 km from downtown Denver, aroused public anxiety about protection of human safety and private property in the expanding residential areas of the Front Range foothills. Concern regarding the status of the Denver water supply and recreational fishing locations focused attention on the response of watersheds to high severity wildfire. Public awareness prompted by the Hayman Fire and other large wildfires during 2002 and recent years has prompted widespread implementation of hazardous fuels reduction projects on national forest lands (USDA/USDOI 2005). These efforts include timber

harvesting, prescribed burning and fuels reduction treatments conducted in the wildland-urban interface. However, in spite of the broad mandate for this work, active management of national forestlands remains controversial (Beschta and others 2004, Winter and others 2004).

During the past two decades, the incidence of large forest fires in the western U.S. has increased in response to warmer spring temperatures (Westerling and others 2006). Wildfires such as the Hayman Fire periodically disturb watersheds in Colorado's montane forest zone (Romme and others 2003, Veblen and Donnegan 2005), yet the influence of wildfire and fire behavior on aquatic processes remains poorly understood. As part of the Upper South Platte Watershed Protection and Restoration Project, a multiple-basin monitoring network originated one year prior to the Hayman Fire allowed us to compare streamwater properties in burned and unburned catchments over a broad range of variability in both fire behavior and watershed characteristics. This assessment evaluated immediate changes in streamwater chemistry, temperature and sediment immediately following the fire, and over the course of the first post-fire year.

METHODS

Study Site and Methods

The Hayman Fire burned in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests of the lower montane elevation zone (1980 to 2750 m). Mean annual precipitation recorded at a site within the area burned by the Hayman Fire is 41 cm (Western Regional Climate Center 2006); summer rain and snow each account for about half the annual precipitation.

The majority of the Hayman Fire area and the study watersheds are underlain by the Pikes Peak batholith, a coarse-grained biotite and hornblende-biotite granite (Bryant and others 1981). Soils weathered from the Pikes Peak granite are weakly developed (Typic Ustorthents), excessively-drained, coarse sandy loams (Cipra and others 2003). Depth to bedrock ranges from 25 to 50 cm and coarse fragments represent 25 to 50 percent of the soil volume. A mixture of sandstone and limestone strata that form deeper soils with higher organic matter and cation content underlie a small portion of the area (Bryant and others 1981, Irvine 1995).

Soils formed from Pikes Peak granite can be highly susceptible to erosion, sheetwash, rilling and gullyng (John 2002, Robichaud and others 2003). Streams draining the Pikes Peak batholith also contain naturally-high fluoride levels due to pegmatite inclusions that are enriched in F, Li and rare earth elements (Simmons and others 1987, Foord and others 1995). Tributaries of the Upper South Platte supply water for the city of Denver and support popular sport fishing sites. Pre-fire water quality concerns regarding sediment and temperature of several of the study streams resulted in their occurrence on Colorado's 303(d) Monitoring and Evaluation list (Colorado Water Quality Division 2002).

Sampling and Analysis

Sampling began in August 2001 on six streams (Table 1). The Hayman Fire affected half the original basins, so our assessment compared pre- and post-fire flow weighted streamwater concentrations in three burned and three unburned watersheds. Four additional sample locations were established following the fire to allow comparisons of

Table 1 Study watersheds within or adjacent to the Hayman Burn, Upper South Platte drainage, Colorado Front Range.

	Watershed Area ha	Burned Area		Burn Severity		
		ha	%	Low	Moderate	High
				-----	%	-----
Burned Watersheds						
Brush	609	532	87	29	3	56
Fourmile	2,145	1,513	71	12	10	49
Goose	4,963	2,574	52	18	4	30
Wigwam	5,753	2,742	48	15	2	31
West	17,888	8,482	47	22	15	11
Trout	30,088	2,555	8	4	4	1
Horse	48,670	11,688	24	11	8	5
Unburned Watersheds						
No Name	345	0	0	0	0	0
Sugar	3,129	0	0	0	0	0
Pine	3,364	0	0	0	0	0

the unburned drainages with drainages affected by varying fire extent (Table 1).

Monthly streamwater samples were collected for dissolved chemical constituents and sediment. Streamwater grab samples were kept cool and then filtered (0.45 μ) prior to analysis by ion chromatography (Waters Ion Chromatographs with a Dionex AS12 A Separator Anion Column and Waters IC Pak Cation M/D Column; APHA 1998a). Detection limits were 0.01 mg/L for Na^+ , NH_4^+ , Cl^- , and F^- , 0.02 mg/L for Ca^{2+} , Mg^{2+} , K^+ , and NO_3^- , 0.03 mg/L for PO_4 , and 0.05 mg/L for SO_4^{2-} . Acid neutralizing capacity (ANC) was measured by Gran titration (Gran 1952) and pH and electrical conductivity (EC) were analyzed automatically with PC Titrate sensors (Man-Tech Co.). Streamwater suspended sediment were collected in 1 L bottles, filtered onto 0.45 μ filters and dried at 105°C (APHA 1998b). Turbidity was measured in the 1 L samples using the nephelometric method (APHA 1998c; HF Scientific, Inc. Micro 100 Turbidimeter). On site stream temperature and EC measurements complimented laboratory analyses and monthly discharge measurements were used to flow-weight streamwater concentrations.

Pre-fire flow-weighted mean concentrations of chemical constituents, temperature and sediment were compared among the six original sample streams using one-way analysis of variance (SPSS Ver. 10.1.3, SPSS Inc., Chicago, IL). Pre-fire / post-fire differences in water chemistry, temperature and sediment were compared graphically and statistically between the three burned and three unburned basins that were part of the original sampling network. To account for pre-fire differences among basins, individual differences for burned and unburned streams were compared using AOV. Comparisons were divided into summer (June-Sept), winter (Oct-Jan) and spring snowmelt (Feb-May) periods. Levene's statistic was used as a test of homogeneity of variance and data were log-transformed when needed. Log transformation succeeded in reconciling violations of the assumption of homogeneity of variance. Significance is reported at the $\alpha = 0.05$ critical value unless noted otherwise. Relations between water quality parameters and basin characteristics were evaluated using least-squares linear regression (SPSS Ver. 10.1.3, SPSS Inc., Chicago, IL).

RESULTS

Prior to the Hayman Fire average streamwater pH was 8.0, EC was 203 $\mu\text{S}/\text{cm}$ and ANC was 1410 $\mu\text{eq}/\text{L}$ in the six original basins. Calcium was the dominant cation in pre-fire streamwater (25.5 mg/L), followed by Na^+ (10.4 mg/L) and Mg^{2+} (5.5 mg/L) and K^+ (3.0 mg/L). ANC was the dominant anion (86 mg/L), followed by SO_4^{2-} (17.0 mg/L) and Cl^- (5.4 mg/L). Nitrate averaged 0.5 mg/L (0.1 mg N/L); the maximum nitrate value measured (1.0 mg N/L) was an order of magnitude below EPA's safe drinking water level (10 mg N/L; U.S. Environmental Protection Agency 2003). NH_4^+ never and PO_4 rarely exceeded detection limits. Pre-fire mean fluoride levels (3 mg/L) exceed the National Secondary Drinking Water Standard (2 mg/L; U.S. Environmental Protection Agency 2003). Turbidity averaged 5 NTUs and did not exceed 10 NTUs in most streams.

The four streams draining basins entirely underlain by Pikes Peak granite had lower concentrations of many streamwater constituents than the two streams (Horse and Trout) that were influenced by mixed sedimentary geology (Fig. 1). Pre-fire conductivity and ANC for Horse and Trout Creeks were both double levels

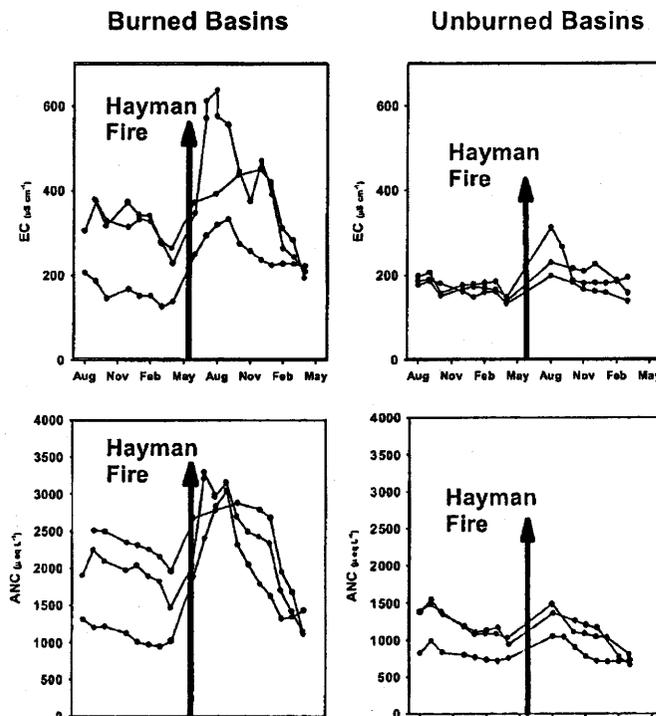


Figure 1. Streamwater conductivity and ANC of three burned and three unburned watersheds in upper South Platte watershed. The arrow indicates the date of fire ignition (6/02).

Pre-fire conductivity and ANC for Horse and Trout Creeks were both double levels

measured in the remaining streams (Fig 1) and pH for those streams was 0.3 units higher. Nitrate, SO_4^{2-} , suspended sediment, turbidity and temperature were similar between the six streams.

During the first four months after the Hayman Fire, pre-fire / post-fire differences in EC (Fig. 1), sediment and flow-weighted concentrations of various ions were greater in the burned compared to the unburned basins. Summer EC increased $120 \mu\text{S}/\text{cm}$ in the three burned catchments (Fig. 1). The average change in ANC was $800 \mu\text{eq}/\text{L}$ for burned basins compared to a slight decline in ANC ($40 \mu\text{eq}/\text{L}$) in unburned streams (Fig 1).

Seasonal mean turbidity was 36 NTUs higher in the burned streams during the first four-month period after the fire; in contrast, turbidity was unchanged between the pre- and post-fire summer season for the unburned streams. During the winter months, the changes in EC, ANC and dominant cations (Ca^{2+} and Mg^{2+}) remained significantly higher in the burned basins. EC was the only chemical constituent that was significantly elevated in burned watersheds during spring snowmelt. After the fire, the three streams draining burned watersheds were about 5°C warmer during summer and snowmelt compared to their pre-fire temperatures in those seasons (Fig. 2); winter temperatures did not differ.

Comparison of the three unburned basins and five basins underlain by Pikes Peak granite during the first year following the Hayman Fire describe similar responses as the pre- and post-fire differences among the original sample streams. Differences in mean streamwater chemistry between burned and unburned basins were greatest during the summer season (Table 2). ANC was twice as high, and Ca^{2+} and EC were 65 and 44 percent higher respectively, in burned streams during the summer period. Also, dissolved PO_4 and NH_4^+ were detectable in burned streams during the post-fire summer season; both compounds were below detection during subsequent seasons in burned streams and at all times in unburned streams (Table 2). Nitrate concentrations were higher in burned streams during summer and winter, but not during snowmelt due to high variability during snowmelt (Table 2). Only streamwater temperature, turbidity and suspended sediments were statistically higher in burned streams throughout the first post-fire year.

The percent of a watershed burned or burned at high severity were both linearly related to nitrate and suspended sediment concentrations of the sample streams (Fig. 3a and 3b); no other streamwater attribute responded to the extent of fire in the basins. Streamwater nitrate concentrations increased five-fold across the range of watershed area burned by the Hayman Fire at high severity (Fig 3a). The area burned explained 28 to 41 percent of flow-weighted streamwater nitrate, depending on the season. In general, streams draining basins that experienced stand-replacing fire on > 30 percent of their area had 3.3-fold higher nitrate concentrations than basins where high-severity fire impacted ≤ 10 percent of the area (Fig 3a). Suspended sediment increased with the proportion of a basin burned ($r^2 = 0.79$; $p = 0.001$) and that burned at high severity ($r^2 = 0.73$; $p = 0.003$) during the snowmelt season only (Fig. 3b).

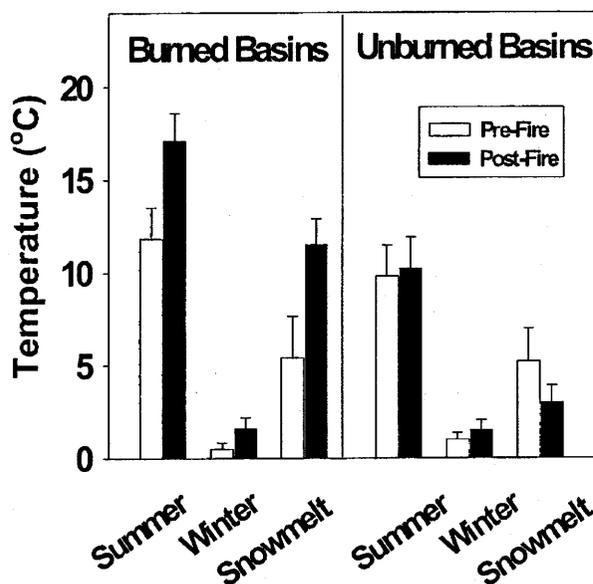


Figure 2. Streamwater temperature of three burned and three unburned watersheds in the upper South Platte watershed. Bars show means and standard errors for four-month periods during the year preceding and the year following the fire.

Table 2 Streamwater ionic composition of burned (n = 5) and unburned (n = 3) tributaries to the Upper South Platte River. Concentrations are seasonal flow-weighted means and (SE) during the first year following the Hayman fire.

	Ca ²⁺	Mg ²⁺	Na ⁺	K ⁺	NH ₄ ⁺	ANC	SO ₄ ²⁻	Cl ⁻	F ⁻	NO ₃ ⁻	PO ₄ ²⁻
	mg/L										
Summer											
Burned	45.1** (2.8)	6.3 (0.9)	7.8 (0.9)	4.7* (0.3)	0.1 (0.0)	149.3** (14.1)	13.9 (1.7)	3.4 (0.6)	2.6 (0.2)	0.4** (0.1)	0.3 (0.2)
Unburned	27.2 (3.3)	4.7 (0.1)	10.0 (1.1)	3.5 (0.6)	0.0 (0.0)	72.2 (5.4)	25.9* (8.3)	7.0** (1.6)	3.5 (0.8)	0.0 (0.0)	0.0 (0.0)
Winter											
Burned	26.2 (4.7)	4.2 (1.0)	7.0 (0.8)	2.3 (0.4)	0.0 (0.0)	86.6 (15.7)	12.7 (2.3)	3.4 (1.1)	2.8 (0.2)	0.8* (0.3)	0.0 (0.0)
Unburned	21.9 (1.9)	4.2 (0.2)	7.5 (1.5)	2.7 (0.5)	0.0 (0.0)	60.7 (8.0)	23.1* (4.2)	3.8 (1.8)	3.2 (0.7)	0.3 (0.2)	0.0 (0.0)
Snowmelt											
Burned	19.7 (4.5)	3.0 (0.9)	5.4 (0.9)	2.5 (0.4)	0.0 (0.0)	53.9 (14.4)	16.6 (3.8)	3.2 (1.1)	2.4 (0.3)	3.0 (1.0)	0.0 (0.0)
Unburned	19.4 (2.0)	3.7 (0.3)	7.0 (0.9)	2.5 (0.7)	0.0 (0.0)	45.2 (2.1)	22.6 (5.8)	4.5 (1.5)	2.9 (0.8)	1.4 (0.9)	0.0 (0.0)

* Seasonal means of burned streams differ from unburned streams at $p \leq 0.1$ significance level.

** Seasonal means of burned streams differ from unburned streams at $p \leq 0.05$ significance level.

DISCUSSION

Immediate Effects

The chemical composition of streams draining watersheds burned by the Hayman Fire exhibited the short-term fire response commonly associated with particulate and gaseous chemical inputs from ash and smoke (Minshall and others 1989). Following experimental ash introduction to a first-order tributary of the Gila River, stream conductivity, ion concentrations and turbidity all peaked one hour after the inputs were added, and then returned to pre-fire levels within 24 hours of ceasing additions (Earl and Blinn 2003). Smoke and ash elevated streamwater nitrogen and phosphorus 5- and 60-fold during wildfire in northwestern Montana; the values returned to background levels several weeks after the burn (Spencer and others 2003). Similarly, concentrations of various streamwater constituents (NH₄⁺, NO₃⁻, K⁺, P, alkalinity) that increased following wildfire in mixed conifer, ponderosa pine and scrub oak (*Quercus turbinella*) communities returned to pre-fire levels one to four months following burning (Earl and Blinn 2003). Even moderate-severity prescribed fire burning in mixed conifer forests generated immediate increases in Ca²⁺ and SO₄²⁻ (1.3- and 13-fold, respectively) in streams of the Lake Tahoe Basin; streamwater chemistry returned to pre-fire levels within three months (Stephens and others 2004).

Chemical concentration of South Platte River tributaries influenced by Hayman Fire combustion was significantly higher during the four-month period following the fire (Fig. 1 & 2; Table 2). Several constituents (ANC, EC) and the sums of cations and anions remained higher than unburned streams or pre-fire levels during the 2002/2003 winter season as ash deposits continued to enter streams in burned watersheds. Differences diminished further during the 2003 spring snowmelt period.

Sustained Effects

Second-order fire effects associated with loss of forest vegetation and altered soil processes are expected to reach their peak a few years after wildfire (Minshall and others 1989). Streamwater NO₃⁻ was higher in the burned streams during summer, winter (Table 2) and the first post-fire year as a whole. Streamwater temperature was significantly

higher in burned streams immediately after the Hayman Fire and remained elevated above that of unburned streams at the end of the post-fire study period (Fig. 2). Increased light caused by loss of riparian vegetation may maintain higher stream temperatures for two to six years before shade from regenerating shrub and tree canopies attains pre-fire levels (Minshall 1978, Minshall and others 1989).

Wildfire initiates changes in terrestrial nutrient cycling that continue for decades before forest composition and soil processes return to pre-fire conditions. Mineral and organic soil layers are the largest reservoirs of nitrogen (N) in most forest ecosystems (Schlesinger 1991) so combustion of these N pools or acceleration of N cycling processes can increase nutrient leaching following fire (Chorover and others 1994, Murphy and others 2006). Oxidation of

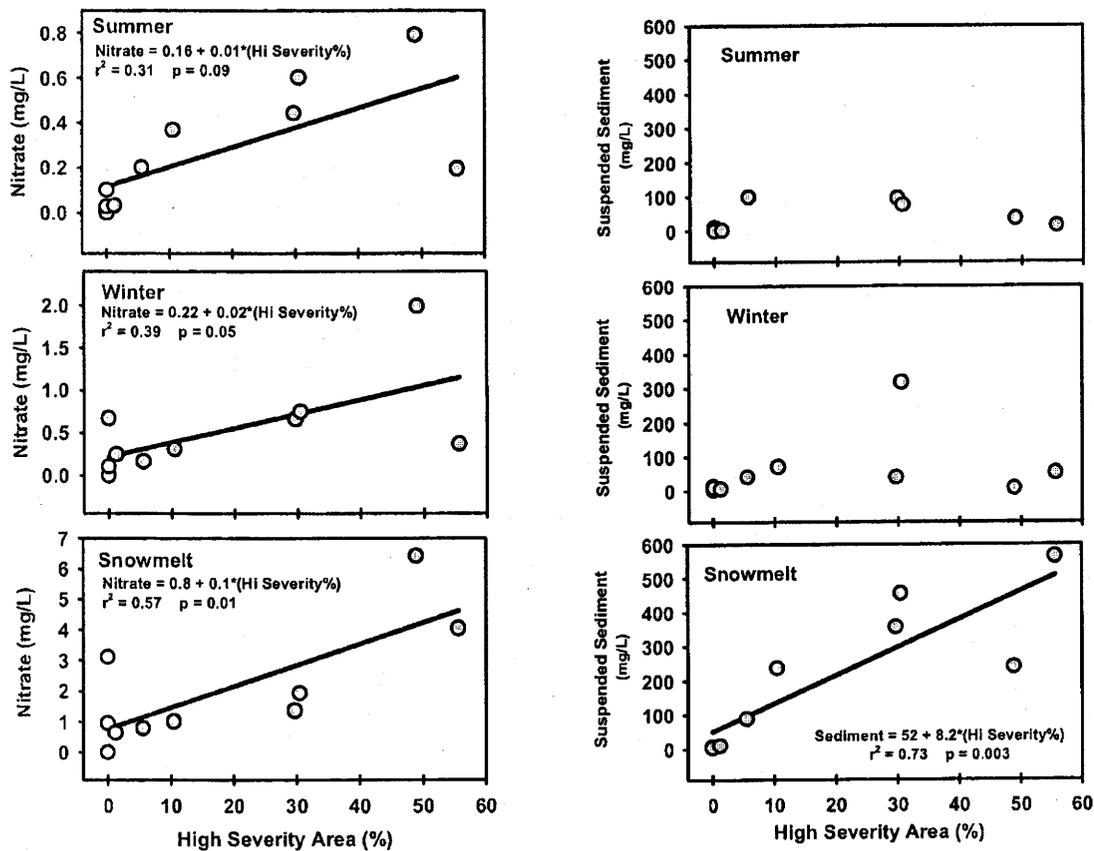


Figure 3a and 3b. Seasonal relations between flow-weighted streamwater nitrate (a) and suspended sediment (b) and the percent of study basins affected by high-severity combustion during the Hayman Fire.

forest biomass, litter and soil organic matter commonly increases the availability of NH_4^+ in soil for months to several years following wildfires or slash burning (Covington and others 1991, Giardina and Rhoades 2001). Prolonged changes in streamwater N result from the combined influence of reduced plant N demand, stimulated N mineralization and nitrification due to increased soil temperature and pH, and subsequent release to groundwater and surface water. Nutrient export to aquatic ecosystems declines as soil microbes and regenerating vegetation increase N demand. Following the Yellowstone fires, streamwater nitrate remained higher than background levels for five years (Brass and others 1996, Robinson and Minshall 1996). It is assumed that due to the extent and severity of the Hayman Fire that forest structure will not recover for many decades (Personal communication, Laurie Huckaby, 2006, Ecologist, Rocky Mountain Research Station, Fort Collins, CO 80526). Likewise, we expect the legacy of the Hayman Fire on streamwater nitrate to persist for years.

Basin Characteristics

The Hayman Fire burned from 8 to 87 percent of the study basins included in our assessment (Table 1). The area of the seven basins spanned nearly three orders of magnitude (5 to 500 km²), and the percent burned in a basin decreased linearly as the log of basin area increased ($r^2 = 0.83$, $p = 0.004$). The proportion of the study basins burned at high severity was also well-related to the log of basin size ($r^2 = 0.62$, $p = 0.036$).

We found that the extent of a basin burned at high severity was related to post-fire change in chemical and physical water quality (Fig. 3a and 3b). High-severity fire generated seven-times more nitrate release from two southern California chaparral watersheds compared to two basins burned at lower severity (Riggan and others 1994). Nitrate losses were attributed to increased soil nitrification combined with sediment movement from surface erosion and debris flows. In the New Jersey Pine Barrens, water and cation fluxes to groundwater were highest in wildfire areas, intermediate in prescribed burned areas and lowest in unburned sites (Boerner and Forman 1982).

Basins inside the Hayman Fire with ≥ 30 percent burned at high severity had roughly twice the streamwater nitrate and four times the suspended sediment concentrations as basins with ≤ 10 percent burned under such conditions (Fig. 3b). Following the 1988 Yellowstone fires, changes in streamwater chemistry and stream habitat increased with extent burned (0 to 90 percent) across 21 basins (Robinson and Minshall 1996). In both Hayman and Yellowstone fires post-fire streamwater condition were related to basin size (Minshall and others 1989); greater discharge from the larger basins dilutes the increased solute, temperature and sediment triggered by the fire. Conversely, smaller basins undergo greater changes following fire.

The streams and watersheds burned by the Hayman Fire will continue to respond for decades. It is uncertain how the recovery process will vary among basins of different size and those affected by high-severity fire to differing degree. Future work will evaluate the links between recovery of riparian vegetation and streamwater quality across a range of stream order and burn severity.

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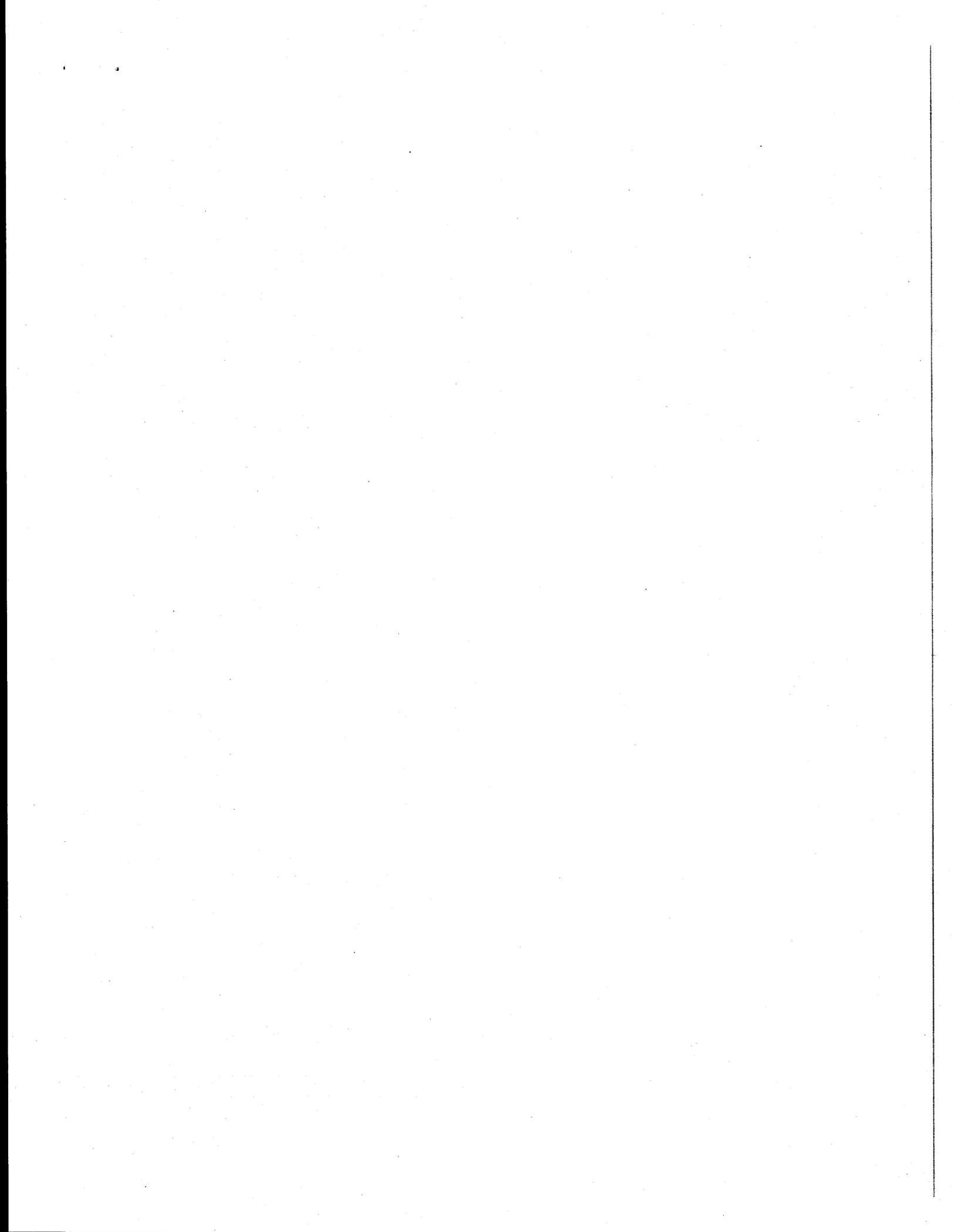
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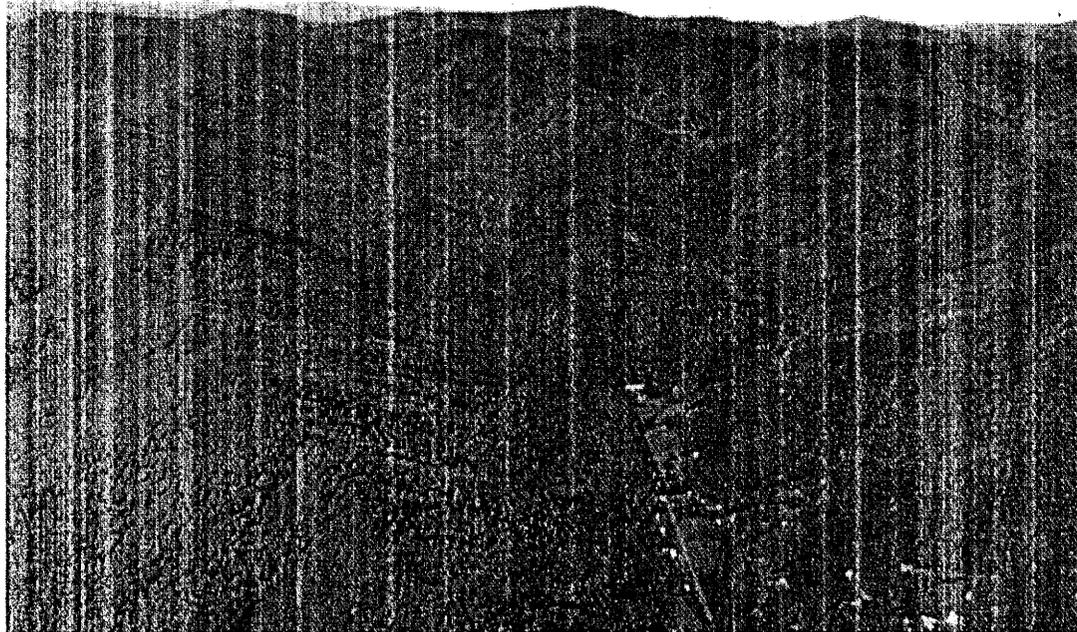
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THE COMPARISON OF TWO WATERSHEDS USING A WATERSHED NUTRIENT LOADING MODEL

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Abstract - Nutrient monitoring data from the Yaquina River, Oregon were used to develop a relationship between nutrient flux and water discharge as part of an effort to quantify estuarine-watershed nutrient processes. The resulting empirical nitrate loading (linear regression) model showed that in-stream nitrate concentrations were generally a direct function of water discharge normalized to drainage area. The r^2 value was found to be 0.65, although the statistic increases to 0.86 upon excluding early fall major storm events. To test its generality, the model was applied to data obtained from the Alsea River in the watershed immediately to the south. It was found that the unadjusted model overestimated observed nitrate concentrations by about a factor of two. An investigation of the greater relative nitrate loading in the Yaquina River, including consideration of atmospheric input and other sources, suggested that the nitrogen-fixing tree species, red alder or *Alnus rubra*, was responsible for most of the difference. This would also explain the change in the r^2 statistic early in the water year, if nitrate in Oregon coastal rivers contained nitrogen leaching from litter fall in alder stands in the surrounding watershed. The analysis supported the alder hypothesis, although the conclusions would be strengthened by data from other watersheds or longer time series. Specifically, although a strong statistical relationship was shown to exist using multiple linear regression (MLR), the new discriminating variable, alder density, is a categorical variable. Therefore, it can only provide the offset between the two data clusters. The results indicated that the model provides a rapid and reasonable first estimate of nitrate concentrations from water discharge data and alder distribution.

INTRODUCTION

In the Northeastern Pacific, hydrographic input to coastal rivers and estuaries is controlled by large seasonal variations in rainfall. During winter months of peak rainfall and river discharge (Table 1), dissolved nutrients and terrestrial sediments are transported down the river into the estuaries. During summer months of low river discharge, upwelling events along the Oregon coast provide pulses of nutrient rich water that supplement post-bloom, nutrient depleted waters in coastal bays and estuaries (Colbert and McManus, 2003; Sigleo et al., 2005). Despite the importance of nutrients on ecosystem processes, quantitative data to evaluate the potential impacts of future development are not available for most Northeastern Pacific estuaries (Colbert and McManus, 2003).

The relationship between nutrient flux and water discharge was derived as part of an effort to quantify watershed-river nutrient processes. The resulting empirical nitrate loading model (linear regression) showed that in-stream nitrate concentrations were generally a direct function of water discharge normalized to drainage area. The r^2 value was found to be 0.65, although when early fall major storm events were excluded, the statistic increased to 0.86 (Sigleo and Frick, 2003). During the time interval studied, over 94% of the dissolved nitrate was transported from the watershed during the winter months of greater rainfall, indicating that seasonality and river flow are primary factors when considering annual nutrient loadings from this watershed system.

The specific objective of the following work was to compare the predicted and measured amounts of watershed derived dissolved nitrate transported into the Yaquina and adjacent Alsea Rivers from their respective watersheds using the previously derived regression.

YAQUINA AND ALSEA WATERSHEDS, OREGON

The Yaquina and Alsea rivers flow into macro-tidal estuaries on the west coast of the United States, and then into the Pacific Ocean at Newport and Waldport, respectively. The watersheds rise from sea level to an elevation of 1037 m at Mary's Peak. The Yaquina drains a surface area of 655 km², whereas the Alsea watershed drains a surface area of 1220 km². The region is characterized by a maritime climate with mild wet winters and cool drier summers. Winter rainfall is controlled by regional cyclonic storms, suggesting that storms will affect both watersheds in a similar manner.

Table 1. Watershed parameters.

	Yaquina	Alsea
Watershed area	655 km ²	1220 km ²
Gaged watershed area	177 km ²	865 km ²
Maximum discharge	78 m ³ s ⁻¹ (1996)	1184 m ³ s ⁻¹ (1964)
Summer discharge	0.28 m ³ s ⁻¹	1.27 m ³ s ⁻¹
Land use—silviculture*	40% conifer	59% conifer
Hardwoods (red alder)*	13% (22%) [†]	8% (12%) [†]
Mixed, 20-65% Hardwoods*	21% (15%) [†]	21% (16%) [†]

*CLAMS 1996; [†]vegetation: version 2 (version 1)

PROCEDURES

Water samples were collected simultaneously from the Yaquina River at USGS gaging station 14306030 and from the Alsea river at Boundary Road bridge above USGS gaging station 14306500. The filtered samples were analyzed for dissolved nitrate, ammonium, phosphate and silica using a Lachat Quikchem 8000 Flow Injection Analyzer. Dissolved organic nitrogen was the difference between total Kjeldahl nitrogen and inorganic nitrogen (nitrate and ammonium). Temperature and conductivity were measured with a Yellow Springs Instruments (YSI) 30 meter.

GIS METHODOLOGY AND WATERSHED DELINEATIONS

The Yaquina and Alsea watersheds were delineated by a combination of BASINS 3.1 (Better Assessment Science Integrating point and Nonpoint Sources) and ArcGIS 9.1 software. Elevation grids from the National Elevation Dataset (NED), and stream network shapefiles from the National Hydrography Dataset (NHD) for cataloging units 17100204 and 17100205, were downloaded using BASINS. Flow direction grids were created for each HUC (Hydrologic Unit Code) using the above datasets and BASINS' automatic watershed delineation tool. ArcGIS basin tool was used to delineate the drainage basins inside each HUC using the flow direction grids created earlier in BASINS. The Yaquina and Alsea drainage basins were extracted and converted to shapefiles using ArcGIS.

The vegetation classification for the Yaquina and Alsea watersheds was obtained from the Coastal Landscape Analysis and Modeling Study (CLAMS). Data sets 1996 Veg (34 class) - ver 2.0 and 1996 Vegetation - ver 1.0 were from www.fsl.orst.edu/clams/data_index.html on 04/19/2006 and 05/03/2006, respectively. ArcGIS tools were used to clip the datasets by the watersheds' boundaries, calculate the number of unclassified cells and extract the cells corresponding to water. Excel 2003 was used to calculate the percentage of each land cover class inside each watershed for both versions of the data.

RESULTS

The dissolved nitrate concentrations in the Alsea River varied from 0.43 to 0.739 mg l⁻¹, whereas the nitrate in the Yaquina River varied from 1.26 to 1.66 mg l⁻¹, more than double that of the Alsea (Fig.1). Temperature, conductivity and the concentrations of dissolved silica, ammonium, phosphate, and organic nitrogen, however, were similar in both rivers (Table 2).

The Yaquina and Alsea drainage basins were delineated with GIS software and percentages of different vegetation classes were calculated from 1996 CLAMS data sets (Fig. 2, Table 1).

Dissolved Nitrate

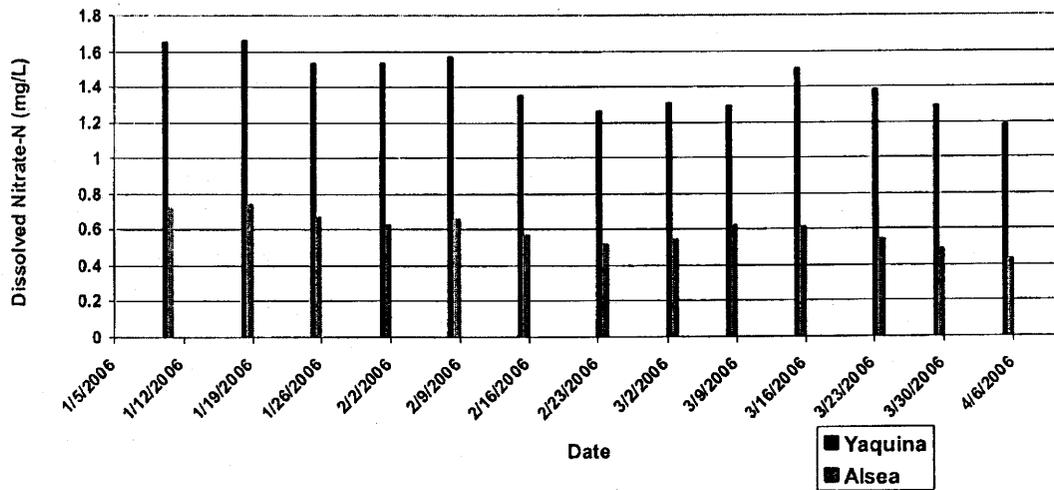
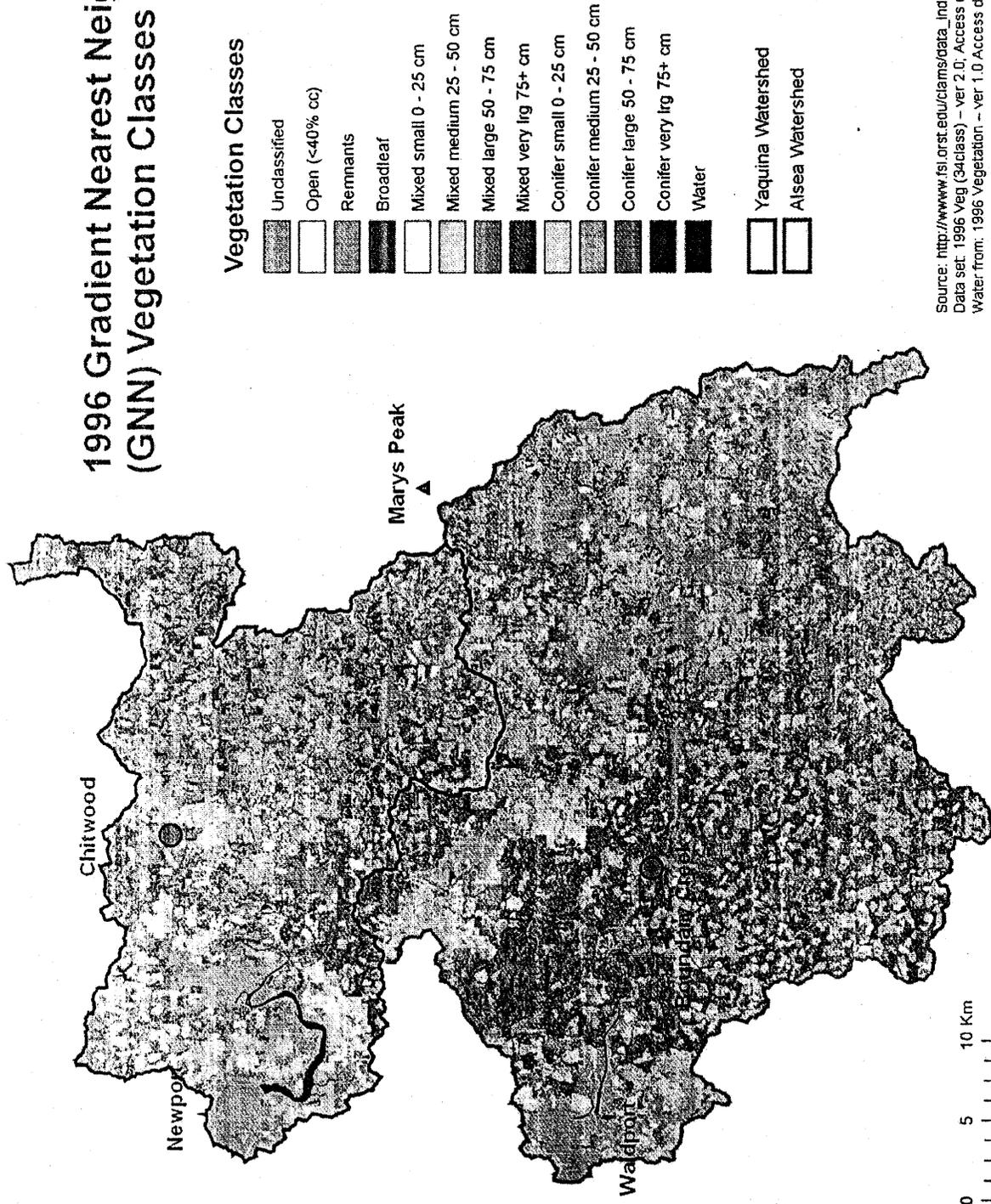


Figure 1. Dissolved nitrate concentrations in the Aelsea and Yaquina Rivers.

Table 2. Nutrient concentrations and flow in the Yaquina and Aelsea Rivers at the gaging stations.

Yaquina River at Chitwood								
Date	Temperature	Conductivity	NH ₃ -N	Nitrate	PO ₄ -P	SiO ₂ -Si	TON	Flow
	C	μS	mg l ⁻¹	mg l ⁻¹	μg l ⁻¹	mg l ⁻¹	mg l ⁻¹	m ³ s ⁻¹
CT060110	10.3	69.5	0.017	1.65	11	5.45	0.183	63.12
CT060118	9.2	69.2	0.008	1.66	8	6.09	0.102	52.81
CT060125	7.7	69.8	0.008	1.53	10	6.35	0.092	14.30
CT060201	9.1	64.1	0.006	1.53	11	5.88	0.104	40.12
CT060208	8.5	71.9	0.009	1.57	16	6.55	0.082	12.74
CT060215	6.3	70.1	0.009	1.35	9	6.45	0.096	5.27
CT060223	7	77.3	0.0105	1.26	8	6.42	0.101	3.28
CT060302	7.3	67.4	0.016	1.3	8.5	6.17	0.178	4.39
CT060308	8.04	66	0.008	1.29	8	6.27	0.113	8.69
CT060315	7.55	64	0.0095	1.5	8.5	6.25	0.094	10.08
CT060323	8.9	64	0.008	1.38	8	6.2	0.042	7.67
CT060329	9.99	64	0.0085	1.29	7	6.17	0.052	5.41
CT060405	9.98	65	0.0095	1.18	8	6.13	0.048	6.51
Aelsea River at Boundary Road Bridge								
AL060110	10.1	49.4	0.017	0.716	13	5.09	0.151	410.6
AL060118	9	58	0.001	0.739	9	5.85	0.134	305.8
AL060125	8.3	64.6	0.006	0.66	11	6.55	0.128	87.22
AL060201	8.9	59.5	0.005	0.627	11	5.63	0.108	276.7
AL060208	8.3	62.1	0.007	0.655	12	6.63	0.050	88.07
AL060215	6.3	67.4	0.008	0.565	10.7	6.77	0.105	40.78
AL060223	6.9	76.5	0.01	0.513	9.5	6.74	0.123	25.63
AL060302	8	55.4	0.008	0.546	9.5	6.44	0.109	42.19
AL060308	8.75	56	0.01	0.62	8	6.41	0.188	69.09
AL060315	7.4	55	0.005	0.611	8	6.23	0.052	58.33
AL060323	9.2	54	0.008	0.54	8	6.25	0.019	50.69
AL060329	9.6	53	0.007	0.479	6	6.11	0.074	46.16
AL060405	9.7	56	0.008	0.43	5	6.12	0.048	36.81

1996 Gradient Nearest Neighbor (GNN) Vegetation Classes



Source: http://www.fs.orst.edu/claims/data_index.html
 Data set: 1996 Veg (34class) - ver 2.0; Access date: 04/19/2006;
 Water from: 1996 Vegetation - ver 1.0 Access date: 05/03/2006

Figure 2. The Alsea and Yaquina river watersheds.

DISCUSSION

The preliminary normalized Watershed Nutrient Loading model applied to the Alsea River, a contiguous watershed to the south of the Yaquina watershed, indicated that Alsea River nitrate concentrations were overestimated by a factor of about two, suggesting that the Watershed Nutrient Loading model required an additional parameter to predict nitrate concentrations adequately. Various factors, including the low nitrate concentrations in local rainfall (Sigleo, unpublished data) and the lack of local primary anthropogenic sources, suggested that red alder (*Alnus rubra*), a nitrogen fixing species, may be the primary source of higher concentrations in the Yaquina, relative to the Alsea River.

A major source of nitrate in Oregon coastal rivers comes from the surrounding watersheds (Compton et al, 2003, 2006). Within the watersheds, the regional vegetation includes considerable broad-leaf hardwood consisting of alder and maple (Table 1). Alder (*Alnus rubra*), a pioneer species in logged areas, forms a symbiotic relationship with the actinomycete *Frankia spp.* that fixes nitrogen from the atmosphere (Vogel and Gower, 1998). In comparative studies of conifer forests with and without alder, the C and N contents of the overstory biomass, total vegetation, and forest soil were greater in conifer forests with alder than in those without alder (Binkley et al., 1992; Vogel and Gower, 1998). In other words, the added nitrate from the alder increased the overall productivity of the system. Compton et al. (2003) found that nitrate and dissolved organic nitrogen (DON) concentrations were positively related to broadleaf cover (94% red alder). The strong relationship between nitrate and DON to broadleaf cover within entire watersheds indicated that leaching from upland alder litter was important in the movement of watershed nitrogen.

To evaluate the alder component, a two-parameter multiple linear regression (MLR) model including flow and alder density (estimated relative hardwood species areal coverage obtained using GIS techniques) yielded an r^2 of 0.974 and a standard error of 0.074 mg l⁻¹. Although the results appear robust, it can not be shown statistically that alder is the most significant source of the nitrate. The relative coverage in the two watersheds provided only information that helps the model separate the two clusters of data, a function that might be served by any other categorical parameter. To make a rigorous statistical inference, data from a third watershed or a longer time-series are required.

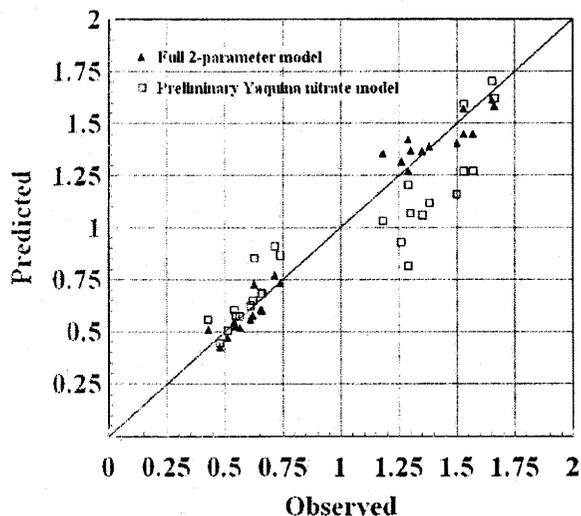


Figure 3 shows predicted versus observed nitrate concentrations (mg l⁻¹) in the Alsea River (left cluster) and Yaquina River (right cluster) measured weekly January through April 2006. The results of a two-parameter multiple linear regression (MLR) model analysis are shown as triangles. Compared to the original analysis, the additional (categorical) variable, alder density, is expressed as a fraction of coverage. Both independent variables are significant (p-value < 0.000036). R-squared is 0.969 with a standard error of 0.081 mg l⁻¹.

The results of the initial single-variable regression model adjusted by a factor of the ratio of watershed alder density to upper Yaquina watershed density are shown as squares. The ratio is approximately 0.5 for the Alsea watershed predictions. R-squared is 0.836 with a standard error of 0.067 mg l⁻¹.

CONCLUSION

The results provided striking evidence that increased alder density in the surrounding watershed supplied increased nitrate to adjacent streams and rivers. The hypothesis that alder provides a major source of nitrate is supported by the absence of other strong nitrate sources. Measurements from additional watersheds will hopefully strengthen the significance of alder as a source of nitrate. While water samples obtained at gaging sites largely represent integrated watershed properties, other factors such as alder distribution and nearby stream plumes can affect measurements and should be considered.

ACKNOWLEDGMENTS

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REMOVING STORM WATER POLLUTANTS AND DETERMINING RELATIONS BETWEEN HYDRAULIC FLOW-THROUGH RATES, POLLUTANT REMOVAL EFFICIENCY, AND PHYSICAL CHARACTERISTICS OF COMPOST FILTER MEDIA

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Abstract--Compost filter socks are generally used to control sediment on construction sites or land disturbing activities. Higher sediment removal efficiencies of compost filter socks, relative to silt fence, have been attributed to its larger surface area and sediment storage capacity, due to its tubular construction. Compost has been used widely to bioremediate polluted soils. By adding new materials to the compost filter media within the sock, these innovative sediment control devices may be used for storm water pollutant removal applications beyond sediment. Under laboratory test conditions, similar to test methods designed to evaluate silt fence (ATSM D-5141), results from 12 in and 18 in compost filter socks show TSS removal efficiency of runoff of 70% and a turbidity reduction of 74 and 84%, respectively. When compost filter media is specified and designed for high runoff flow conditions, generally, flow through rates are increased at the expense of suspended solids and turbidity reduction. By adding anionic polymers to the compost filter media, such as PAM or a polysaccharide biopolymer, turbidity reduction of runoff can increase from 21% to 90 and 77%, respectively; and TSS removal efficiency can improve from 58% to 90 and 88%, respectively. Additionally, polymers can be added to the filter media to remove hard to capture soluble pollutants, such as dissolved reactive phosphorus. By adding a polymer to the filter media that can adsorb soluble P, test results show that removal efficiencies from storm water runoff can increase from 6% to 93%. Based on 45 samples of compost filter media tested for physical characteristics and runoff pollution control performance, the mean hydraulic flow through rate was 24 gpm/linear ft, mean total solids removal was 92%, mean suspended solids removal was 30%, mean turbidity reduction was 24%, and mean motor oil removal rate was 89%. Based on preliminary correlations, particle size distribution of filter media is the best indicator of hydraulic flow through rate and pollutant removal efficiency, although bulk density of the filter media may be used if particle sizes are unknown, and void space of the filter media may be used to predict flow through rate but not pollutant removal efficiency. The greater the hydraulic flow through rate of a filter media, generally the lower the pollutant removal efficiency. Results from this study indicate that compost filter socks are an effective sediment control device and by adding new materials to the filter sock its applications expand beyond only sediment control to storm runoff filtration capable of capturing target soluble pollutants, such as petroleum hydrocarbons and phosphorus. These practices should be considered to improve receiving water quality and in watersheds where there is a potential for pollution from sediment or soluble pollutants.

INTRODUCTION

Sedimentation rates from construction sites are typically 10 to 20 times greater than from agricultural operations, and 1000 to 2000 times greater than forestlands (US EPA 2005). In a short period of time, sedimentation from a construction activity can exceed decades of natural sedimentation that causes physical, chemical, and biological harm to our nation's water system (US EPA 2005). In 1990, as an extension of the 1972 Clean Water Act, the National Pollution Discharge Elimination System (NPDES) permit program included storm water discharges from construction sites for the first time (US EPA 2005). Enacted in 1999, but not required until 2003, Phase II of this program reduced the minimum construction site area requiring a storm water discharge permit from 5 acres to 1 acre (US EPA 2005).

Organic filter media used to control sediment from construction sites has been widely used but little researched. In unreplicated field trials in Oregon, Ettl and Stewart (1993) reported that compost filter berms reduced sedimentation of total solids by 83% relative to bare soil, and by 72% relative to silt fence over 5 natural rainfall events on a 34% slope. Additionally, the compost filter berm reduced total suspended solids by 93% relative to bare soil and by 91% relative to silt fence.

In a similar study in Connecticut, using unreplicated field plots, Demars and others (2000) reported on a 2:1 slope that a mulch filter berm reduced sedimentation by 97% relative to straw bales and 80% relative to silt fence during a ¼ in rain event. During a 4.35 in rain event, the mulch filter berm reduced sedimentation by 91% relative to a straw

bale and by 92% relative to silt fence. In a preliminary study Demars and Long (1998) reported that a mulch filter berm reduced suspended solids by 93% relative to a bare soil under 1.7 in of natural rainfall using the same experimental site and set-up. In a follow-up study, Demars and Long (2001) reported that hydraulic flow through a mulch filter berm using one liter of runoff containing 500,000 mg L⁻¹ silty sand on a 2:1 slope, in a flume 6 in wide by 8 in high by 36 in, deep was approximately 850 ml/sec.

In replicated field plots, a study at the University of Georgia by Faucette and others (2005) reported that on hydroseeded sandy clay loam subsoils, on a 10% slope over three simulated rain events, with storm intensities in excess of 3 in/hr for 1 hr duration, mulch filter berms reduced total solids loads on a meter squared basis by 35% relative to silt fence, and 98% relative to a bare soil. They also reported that average peak runoff flow rates for a single storm were 0.015 mm/sec and total solids loads from a bare soil with no sediment control treatment (control) were 308,544 g. Faucette and others (2005) found that during the second storm event the filter berm reduced total N loading by 52%, nitrate N loading by 63%, total P loading by 32%, and soluble P loading by 26% compared to silt fence.

In a replicated flume study used to evaluate hydraulic flow through rates of silt fence and compost filter socks at Ohio State University, Keener and others (2006) reported using a sediment-laden runoff concentration of 10,000 mg L⁻¹, containing only clay and silt (no sand), on a 20 degree slope for 30 minutes. Results showed that runoff flow through rates of compost filter socks on average were 50% greater than silt fence and the ponding height behind a 24 in silt fence was 75% greater than a 12 in compost filter sock. At flow rates less than 5 gpm/linear ft an 8 in compost filter sock overtopped at the same time as a 24 in silt fence, while a 12 in compost filter sock took longer to overtop relative to a 36 in silt fence. At 6 gpm/linear a 12 in compost filter sock overtopped at the same time as a 36 in silt fence, while a 18 in compost filter sock did not overtop as quick as a 36 in silt fence (Keener and others 2006). The researchers concluded from these results that the design height of a compost filter sock does not need to be as high as a silt fence, due to higher flow through rates of the filter socks, and that they could be specified on design plans to control sediment on larger watershed areas or longer slope lengths, relative to silt fence. This may be a substantial financial advantage to builders or contractors where fewer linear ft of a sediment control practice need to be purchased, installed, maintained, removed and ultimately disposed.

Using the same apparatus and test methods described later in this study, Faucette and Tyler (2006), found that filter media used for compost filter socks removed an average of 98% total solids, 71% suspended solids, and reduced turbidity by 55%. No physical characterization of the compost filter media was reported.

While specifications for silt fence used for sediment control are not new, specifications for compost filter media (compost specified for use in a berm or a sock) are relatively new. Although many state agencies have approved and adopted specifications for compost filter socks and/or filter berms, the specifications published by the US EPA National Menu of BMPs for Storm Water Phase II Construction Sites (2006), the Association of State Highway Transportation Officials (2003), Filtrexx International (2006), the Texas Department of Transportation (2005), and the New England Transportation Consortium are generally regarded as the best and most comprehensive.

It is important to note that specifications for compost used for sediment control are quite different than those commonly used for vegetation and planting applications, such as in potting mixes, landscaping, nursery operations, and gardening and agriculture. Specifically, the particle size distribution of the compost filter media is much larger. Table 1 lists particle size distributions in a variety of standard specifications for compost used in sediment control applications. Table 2 lists the hydraulic flow through rate, sediment removal efficiencies, and their associated particle size distribution for filter berms and filter socks reported in the research literature. Although the specifications vary, the research indicates that smaller particle size distributions have higher sediment removal efficiencies; particularly, the greater the amount of fine particles (<1/4 in) the better the resultant sediment removal. In light of this, particle size specifications listed here may be too coarse to remove fine particulates characteristic to sediments that are commonly in suspension (clay and silt), e.g. sediments less than 2 mm. However, as shown in Table 2, hydraulic flow through rate is dependant on sediment concentration of runoff, and greater sediment removal efficiency may be associated with slower hydraulic flow through of runoff (or greater retention time of runoff) - which is also characteristic to smaller particle size distributions of the filter media.

Although standard test methods have been established for evaluating silt fence performance (ASTM D 5141- Standard Test Method for Determining Filtering Efficiency and Flow Rate of a Geotextile for Silt Fence Application

Using Site Specific Soil), none have been established for any other sediment control device (for example, tubular sediment control devices). ASTM standard test method D-5141 uses a 12 in silt fence, 6:1 slope, runoff sediment concentration of 2890 mg L⁻¹ using site specific soil, 50 liters of runoff, in plots 48 in long by 34 in wide, and the silt fence is pre-wet using 50 L of clean water. Barrett and others (1995) found that many of the sediment removal efficiencies reported for silt fence, employing this test method, use predominately sand (instead of silt or clay), which is relatively easy to remove from runoff water due to its larger and heavier characteristics that prevent it from becoming suspended in water – which consequently has little influence on reported turbidity and suspended solids values. Barrett and others (1995) went on to report that 92% of the total suspended solids in runoff are clay and silt, which are an order of magnitude smaller than the openings in the silt fence fabric, and due to very low settling velocities are normally not removed by this method of sedimentation (Barrett and others 1998). They concluded that effective sediment trapping efficiency of silt fence is a result of increased ponding behind the silt fence, while a similar study by Kouwen (1990) concluded that excessive ponding is largely due to eroded sediment clogging the fabric of the silt fence. By these reports, the same function that causes silt fence to trap sediment, is also the function that severely reduces flow through rates, which often causes it to fall over and fail (assuming it has been installed correctly).

The test methods used to evaluate compost filter media in this study are similar to ASTM D-5141 for silt fence. The Standard Test Method for Sediment and Chemical Removal of Filter Media Used in Filtrex Filter Soxx has been reviewed and published in the 2006 International Erosion Control Association (IECA) Annual Proceedings, Long Beach, CA (Faucette and Tyler 2006). This test method uses a 3:1 slope, an 8 in compost Silt Soxx, runoff sediment concentration of 3000 mg L⁻¹ of 33% sand and 67% silt, using 50 liters of runoff, in a flume 4 ft long by 12 in wide, and the compost filter media is pre-wet using 50 L of clean water.

Because the particle size distribution of compost filter media can be easily manipulated, compost filter socks can be customized for high concentrated flow applications, such as ditch checks or inlet protection practices. When the compost filter media is specified for high flow conditions, generally, flow through rates are increased at the expense of suspended solids and turbidity reduction. Polymers such as polyacrylamides (PAM) have been used to reduce turbidity in erosion control applications (Hayes and others 2005). By adding anionic polymers, such as PAM or a polysaccharide biopolymer, to the compost filter media, turbidity reduction and TSS removal efficiency from storm runoff may be improved.

In a 1998 water quality assessment conducted for the US EPA, 35% of streams sampled were found to be severely impaired and nutrient loading was the main cause of 30% of those listed (US EPA 2000). Total maximum daily load (TMDL) listed streams for phosphorus have become increasingly common in recent years. While erosion and sediment control BMPs may reduce sediment bound P, they do little to reduce soluble P in runoff. Additionally, when soil becomes detached, sediment bound P can quickly become desorbed, therefore transforming into soluble P (Westermann and others 2001). Where sedimentation is minimal due to effective erosion control management practices, soluble P can be more than 80% of total P (Berg and Carter 1980). In order improve receiving water quality, and in particular to meet TMDL requirements for phosphorus, BMPs need to be developed to reduce soluble P loading to streams. Soluble P is more reactive, or bioavailable, than sediment-bound P to aquatic plants, therefore, it is more likely to cause algae blooms and eutrophic conditions which contributes to the degradation of our nation's surface waters.

Studies have shown that using polymers can reduce soluble P in sediment ponds (Leytem and Bjorneberg 2005) and total phosphorus in storm runoff by as much as 75 to 90% (Moore 1999, Harper and others 1999). These polymers may be added to the filter media to remove hard to capture soluble pollutants, such as dissolved phosphorus. If effective, filter sock applications may expand beyond being primarily a tool to control sediment to a storm water filtration device capable of capturing soluble pollutants in a wide variety of applications used to improve storm water runoff quality and ultimately our nation's surface waters as well.

The objectives of this study on compost filter socks were: 1) to determine pollutant removal efficiencies for suspended solids, turbidity, and soluble P, and how the addition of selected polymers affect removal efficiencies for targeted pollutants; 2) to determine if physical properties of the filter media affect the hydraulic flow through rate and pollutant removal efficiency of the filter media; 3) to determine if there is a relationship between hydraulic flow through rate and pollutant removal efficiency.

MATERIALS AND METHODS

Beginning in the spring of 2004, compost products were sampled and tested for efficacy as sediment control and storm water runoff filter media. Forty-five compost products were sampled from commercial and municipal composting operations from the United States, Canada, Japan, and New Zealand. The compost products used in this study were produced from a range of carbonaceous feedstocks (i.e.: yard waste and tree trimmings). It should be noted that the dimensions of the testing apparatus, the sediment-laden runoff concentrations, and runoff volume used for this study are similar to ASTM D-5141 used to determine flow through rate and sediment removal efficiency for silt fence technology.

Sampling Procedure and Design

To test for filtration efficacy, compost filter medium were subjected to a laboratory scale storm runoff event, meant to simulate the conditions of storm water passing through an 8 in diameter compost-filled Filtrexx Filter Sock. To achieve this, a tilt table was designed and produced (by Soil Control Lab of Watsonville, CA) to hold the compost filter media while water washed down a slope and through the material. The tilt table used was 4 ft in length where water flows from one end of the table, through the filter medium, and out the other end of the table, where runoff water samples can be taken. The part of the tilt table responsible for holding the filter material (the basket) is adjustable in its height (up to 10 in) and its length (6-24 in), while the width is set at 8 in across. For the purpose of these studies the basket was maintained at 8 in high, 8 in long, and 8 in across to mimic an 8 in diameter Filtrexx Filter Sock. The basket provided a secure fit around the filter media, therefore preventing water from bypassing the tested material. The basket was composed of two 10 in by 8 in rectangles of ½ cm steel mesh and wrapped in Filtrexx Filter Sock mesh material. This steel mesh spanned across the tilt table, snugly fitting across the bottom and the two sides (8 in across). Inside the basket the 8 square in of filter media to be tested is compacted. In order to mimic slopes encountered where sediment control devices are generally specified for construction activities, the tilt table had adjustable slope ratios from 4:1 to 1:1. In this study, however, the slope was maintained at a ratio of 3:1. The runoff distributors were situated at the top of the slope to generate an even runoff sheet flow. The runoff distributors were connected to a 57 L open-top water tank, equipped with a pump-enabled siphon tube. For the duration of this study, 2 gal/min/linear ft of runoff was pumped through the runoff distribution system.

Test Procedure

After the sample filter media was packed into the Filtrexx Filter Sock-lined basket, City of Watsonville, CA tap water was run down the tilt table and through the filter media for 10 min. After this 10 minute period the inflow tap water and outflow (post filtration water) was sampled and tested for soluble salts (EC). Additionally, at this time the maximum flow rate of the filter media was calculated by measuring the height of the water ponded behind the filter media. After these measurements were taken, the runoff distributors supplied a pollutant-laden storm water runoff containing a predetermined amount of nutrients, metals, organic matter, sand, silt, and clay. Sediment concentrations were approximately 3000 mg L⁻¹, unless the purpose of the test was to evaluate sediment removal efficiency under higher sediment-laden runoff conditions. After 10 min of running the pollutant-laden water through the filter media, the inflow and outflow runoff were sampled and tested for physical and chemical constituents. After this sampling, clean tap water was then run through the runoff distribution system and through the filter material. While tap water was running, motor oil was dripped into the inflow stream, at a constant rate, for a period of 10 min. After this time an outflow sample was taken and tested for oil and grease concentration. The total inflow application of motor oil was determined by calculating the weight of dripping oil for a known period of time.

Water Analysis

The first inflow/outflow samples collected after tap water was run through the system for ten minutes were analyzed for soluble salts (EC) (SM 2510 B). The inflow and the outflow of the pollutant-laden runoff water were analyzed for the following physical and chemical constituents and test methods: Total solids (ASTM D3977-97C), suspended solids (SM 2540 D), total suspended solids (ASTM D3977-97C), turbidity (SM 2130 B), Ammonia (SM 4500-NH₃ H), Nitrate (SM 4500-NO₃⁻ C), total N (calc.), organic N (Leco), reactive P (SM 4500-P), organic P (SM 4500-P), acid hydrolysable P (SM 4500-P), total P (SM 4500-P), total Potassium (EPA 3050/EPA 6010 ICP), total Calcium (EPA 3050/EPA 6010 ICP), total Magnesium (EPA 3050/EPA 6010 ICP), total Sulfate (EPA 3050/EPA 6010 ICP), total Copper (EPA 3050/EPA 6010 ICP), total Zinc (EPA 3050/EPA 6010 ICP), total Iron (EPA 3050/EPA 6010 ICP), total Manganese (EPA 3050/EPA 6010 ICP), total non-soluble Carbon (Leco), pH (SM 4500H+ B), electrical conductivity (SM 2510 B). The motor oil-contaminated outflow sample was tested for oil and grease content by partition gravimetric method (SM 5520 B).

Compost Analysis

A sub sample of the compost filter material taken prior to runoff performance analysis was analyzed for the following constituents using the assigned test methods: Particle size distribution (TMECC 02.02 B), bulk density (TMECC 03.03A), moisture (TMECC 03.09), and packed void space by sand displacement (Soil Control Lab). Packed void space was determined by taking a 500 cc sub-sample of filter media. The wet weight and dry weight of the sample were recorded, and then the dried sample was screened through 4mm mesh. Using an Imhoff cone, the volume and weight of material passing through the 4mm screen was measured. The volume of material that was greater than 4mm in size was measured. Using 500 cc of sand and an Imhoff cone, the sand was added to the cone containing wood chips greater than 4mm. The volume of sand and wood chips was measured. The amount of void space was calculated based on the amount of volume displaced by the sand.

The test procedure described above was conducted on 45 different compost filter media as part of a survey and larger study to evaluate the performance of filter media under these runoff conditions, only the following will be reported in this study:

A 12 in and 18 in filter sock were tested for TSS removal and turbidity reduction efficiency to determine if a larger filter sock performs at a higher removal efficiency.

Polymers were added to the compost filter media to evaluate their potential to remove soil colloids and soluble P from runoff. A PAM and biopolymer were each added separately to filter media to target fine soil colloids typically suspended in water and to reduce TSS and turbidity in runoff. An alum based polymer was added to the filter media of an 8 in wide filter sock, at 25, 50 and 150 g/linear ft, to target soluble P in runoff to potentially remove this pollutant from storm runoff. All polymers were weighed (dry) and manually mixed with the filter media prior to installation and testing on the tilt table.

Particle size distribution, void space, and bulk density of filter media was determined for 45 filter media samples as part of the survey mentioned above. These results were used to determine potential relationships on hydraulic flow through rate and pollutant removal efficiency from runoff of these filter media, respectively. TSS, turbidity, and motor oil were selected as pollutants to be analyzed in the survey, since these are commonly found in storm runoff at or near construction sites and land disturbing activities. The relationship between hydraulic flow through rate and pollutant removal efficiency of the filter media was also investigated.

RESULTS AND DISCUSSION

Compost filter socks are available in a variety of design diameters (8 in, 12 in, 18 in, 24 in), similar to design heights for silt fence (24 in, 30 in, 36 in). Total suspended solids (TSS) removal efficiency and percent turbidity reduction were used to evaluate potential differences in performance between a 12 in and an 18 in filter sock. Results in Figure 1 show no difference in TSS removal efficiency, both approximately 70%, however the 18 in filter sock did increase percent turbidity reduction, relative to the 12 in sock, from 74 to 84%. TSS concentrations may have remained the same due to the fact that the porosity of the filter is essentially the same for both filter socks, e.g. the particle sizes of the filter media used are exactly the same, and therefore solids in runoff moving through one should readily move through the other. The 10% increase in turbidity reduction from the 18 in sock may be due to the size of the filter. The larger the silt barrier, the longer it may take for storm runoff to pass through the filter, allowing for some additional sediment deposition to occur in the process. It should be noted that the main reason for using larger diameter filter socks is to increase the design height to handle potentially high runoff flow conditions or large watershed drainage areas flowing to the sock, or where increased sediment storage capacity and less maintenance are desired.

Although compost filter socks have high total solids removal efficiencies (Faucette and Tyler, 2006), removing finer sediment particles, such as clay and silt, which are characteristic to suspended solids and turbidity values, can be a greater challenge. Testing has shown that by reducing the particle size distribution of the compost filter media, TSS and turbidity can be greatly reduced, however a concomitant reduction in flow through rate may also occur, therefore reducing its ability to flow storm water (possibly requiring a larger diameter filter sock to prevent overflow and/or reduce maintenance of sediment removal behind the sock due to increased rate of sediment accumulation). By adding polymers such as polyacrylamides (PAM) and polysaccharides (biopolymer) to the compost filter sock, turbidity and suspended solids in storm runoff may be greatly reduced (Figure 2 and Figure 3) without manipulating

(reducing) the particle size of the filter media and therefore potentially reducing hydraulic flow through rates. These polymers act as an anionic flocculent, thereby flocculating suspended clay particles in storm water and increasing the likelihood of deposition from runoff. They are also coagulants, which act as a mobility buffer to the finer particles transported in storm runoff.

By adding PAM to a filter sock designed for high flow conditions (where removal of fine sediments is more difficult), adding a PAM or a biopolymer can increase turbidity reduction from 21% to 90% and 77%, respectively. The PAM and biopolymer can also increase TSS removal efficiency from 58% to 90 and 88%, respectively. Additionally, turbidity and TSS reduction from runoff were not substantially diminished during a second runoff event, indicating that the PAM and biopolymer were still effective after a first runoff event, and possibly effective for multiple runoff events, thereby increasing its value in the marketplace.

Polymers may also be used as colloidal flocculents to chemically adsorb soluble phosphorus (P) from storm runoff. By adding these polymers to the compost filter sock, through chemical adsorption of runoff soluble P as it flows through the filter media, the filter sock becomes a management practice for removal of soluble P from storm water runoff. Once the soluble P becomes chemically bound to the polymer within the filter sock it has been removed from the runoff and is also no longer bioavailable to aquatic plants, thereby reducing algae growth and subsequent eutrophication effects that can negatively affect surface water quality. By adding 150 g/linear ft of alum-based polymer to an 8 in filter sock, runoff with a soluble P concentration of 100 mg L⁻¹ was reduced from 6% (no polymer) to 93% (polymer added) (Figure 4). Additionally, with a runoff soluble P concentration at 100 mg L⁻¹, and polymer application rate of 50 g/linear ft, soluble P was reduced by 79%; and reducing the application rate of polymer within the filter sock to 25 g/linear ft, soluble P removal efficiency was 67%. Soluble P reduction (chemical adsorption) from the polymer additive was not seen after two consecutive runoff events without additional polymer applications. This may indicate at this runoff soluble P concentration all chemical reaction sites present on the polymer have been filled, thereby reaching its capacity to adsorb P, or the polymer has been transported from the filter sock by the runoff. A small percentage of soluble was removed by the filter sock without polymer addition; this may be due to chemical adsorption of soluble P by humus colloids in the compost filter media.

It is well understood that the more porous a filter is, the greater the rate at which materials move through the filter. It has been assumed that the more porous the compost filter media, the faster runoff will flow through and lower pollutant removal efficiencies will be observed. State and federal standard specifications for compost filter berms and filter socks list particle size distribution requirements to insure adequate hydraulic flow through rates and sediment removal efficiencies in the field. These specifications make the assumption that larger particle size distributions lead to greater hydraulic flow through rates and smaller particle size distributions lead to greater sediment removal efficiency. In addition to particle size distribution of the filter media (within the sock or berm), bulk density and void space of the filter media may give an indication, and be used to predict, hydraulic flow through rate and pollutant removal efficiency.

Based on 45 samples of compost filter media tested, mean void space was 41%, mean bulk density was 26 lbs/cubic yard, mean particle size distribution above 3/8 in was 41%, mean particle size distribution below 1/4 in was 44%, mean hydraulic flow through rate was 24 gpm/linear ft, mean total solids removal was 92%, mean suspended solids removal efficiency was 30%, mean turbidity reduction was 24%, and the mean motor oil removal rate was 89%.

Results from this survey showed there was a linear relationship in hydraulic flow through rate and percent of pollutants passing through the filter media. Generally, the lower the flow through rate of the filter media the higher the resultant pollutant removal efficiency. This may be due to sediment deposition from runoff flow restriction, or simply because fewer pores and smaller pore spaces lead to an increased ability to physically trap small sediments in runoff. This relationship was observed for suspended solids, turbidity, and motor oil removal efficiencies relative to hydraulic flow through rates for filter media (Figure 5, Figure 6, Figure 7).

Similarly, there was a correlation in the percent of particle sizes over 3/8 in and under 1/4 in and hydraulic flow through rate (Figure 8 and Figure 9). Generally, the greater the percent of particle sizes over 3/8 in, the higher the flow through rate; conversely the greater the percent of particle sizes below 1/4 in the lower the flow through rate. This is likely because the greater the amount of small particle sizes in the filter media matrix the lower the porosity (or number of pores) and the smaller the pore spaces. Additionally, more small particles generally means more surface area, which may increase friction on the runoff water passing through the filter media, thereby slowing its

movement through the media. Although the relationship does not appear to be as strong, greater void space and lower bulk density of the filter media also lead to higher hydraulic flow through rates (Figure 10 and 11).

Void space, bulk density, and particle size distribution of the filter media may also indicate (and predict) how well the filter media will remove target pollutants from runoff. Figures 12 through 23 graph the relationships between each of these physical parameters of the filter media and their associated performance in reducing pollutants from storm runoff. Based on the results presented in Figures 12 through 23, void space is not a good indicator of pollutant removal efficiency (for TSS, turbidity, motor oil), while bulk density and particle size distribution are good indicators. High bulk density, a low percent of particle sizes above 3/8 in, and a high percent of particle sizes below 1/4 in. in the filter media are good indicators that the filter media will have high pollutant removal efficiencies. Small particle sizes present in the filter media likely trap sediment for the same reasons they slow down hydraulic flow through rate described above. Greater surface area within the filter media may also provide for greater potential to trap and remove solids from runoff. Fewer pores and smaller pore spaces, characteristic to smaller particle size distribution of filter media, are more likely to trap smaller sediments transported in runoff, thereby reducing TSS and turbidity.

SUMMARY AND CONCLUSION

Under laboratory test conditions, similar to ASTM D5141 standard test methods used to evaluate silt fence, results from 12 in and 18 in compost filter media (used in filter socks) show a TSS removal efficiency of 70% and a turbidity reduction of 74 and 84%, respectively. When the compost filter media is specified for high flow conditions, generally, flow through rates are increased at the expense of suspended solids and turbidity reduction. By adding anionic polymers to the compost filter media, such as PAM or a polysaccharide biopolymer, turbidity reduction can increase from 21% to 90% and 77%, respectively; and TSS removal efficiency can improve from 58% to 90 and 88%, respectively. Additionally, polymers can be added to the filter media to remove hard to capture soluble pollutants, such as soluble phosphorus. By adding a polymer to the filter sock that adsorbs soluble P, test results show that removal efficiencies from storm runoff can increase from 6% to 93%. Results from this study indicate that compost filter socks are an effective sediment control device; however, by adding new materials to the filter sock its applications may expand beyond a tool used primarily to control sediment, to a storm water filtration device capable of capturing target pollutants, making it useful in a wide variety of applications where improvement of storm water runoff and receiving water quality is needed. Based on a survey of 45 samples of compost filter media, the mean hydraulic flow through rate was 24 gpm/linear ft, the mean total solids removal was 92%, the mean suspended solids removal was 30%, the mean turbidity reduction was 24%, and the mean motor oil removal rate was 89%. Generally, the greater the hydraulic flow through rate of a filter media, the lower the pollutant removal efficiency. Particle size distribution of filter media was the best indicator of hydraulic flow through rate and pollutant removal efficiency, although bulk density of the filter media may be used if particle sizes are unknown. If pollutant removal efficiency of filter media is inadequate smaller compost particles may be added to increase removal rates; however, this will likely reduce hydraulic flow through rate of the filter media. State and federal standard specifications for compost filter socks and filter berms should determine optimum hydraulic flow through rates and pollutant removal efficiencies based on particle size distribution of the filter media for various soil (sand, silt clay) and rainfall/runoff potential conditions.

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Table 1--Particle size specifications for compost filter berms and filter socks.

Specifying Agency	% Pass 2 in	% Pass 1 in	% Pass ¾ in	% Pass ¼ in
TX DOT 5049* - berm/sock	95	65	65 (5/8 in)	50 (3/8 in)
AASHTO MP 9-03 - berm	100 (3 in)	90-100	70-100	30-75
US EPA - berm	100 (3 in)	90-100	70-100	30-75
US EPA - sock	100		10-30 (3/8 in)	
NETC - berm	100 (3 in)		70-95	30-75 (1/20 in)
Filtrex International - sock	99		<30 (3/8 in)	

* 1:1 blend of compost and untreated wood chips (termed Erosion Control Compost).

Table 2--Particle size distributions of filter media and their reported sediment-laden hydraulic flow through rate and sediment removal efficiencies.

Treatment	Reference	Hydraulic Flow Through	Sediment Concentration (mg L ⁻¹)	Total solids (%)	Suspended solids (%)	Particle size % pass		
						1 in	¾ in	1/4 in
Filter Berm ¹	Demars and Long 1998	ND	15,000	ND	93	98	ND	ND
Filter Berm ¹	Demars and others 2000	ND	340,000	99	ND	94	64	55
Filter Berm ²	Demars and Long 2001	ND	500,000	99	ND	85	68	38
Filter Berm ³	Demars and Long 2001	1.1 g/min/lin ft	500,000	99	ND	45	34	18
Filter Berm ³	Demars and Long 2001	0.9 g/min/lin ft	500,000	20	ND	30	18	6
Filter Berm ²	Faucette and others 2005	1.0 g/min/lin ft	1,200,000+	98	ND	99	95	93
Filter Sock ⁴	Keener and others 2006	7.5 g/min/lin ft	100,000	38	ND	99	73 (1/2 in)	36
Filter Sock ⁴	Gharabaghi 2006	7.25-10.91 g/min/lin ft	0	ND	ND	92	86	25
Filter Sock ⁴	Gharabaghi 2006	7.25-13.09 g/min/lin ft	0	ND	ND	99	91	27
Filter Sock ⁴	Gharabaghi 2006	7.25-13.39 g/min/lin ft	0	ND	ND	99	89	12

¹Did not meet TX DOT specification for filter berm particle size distribution

²Did not meet TXDOT, AASHTO, USEPA, or NETC specification for filter berm particle size distribution

³Did not meet AASHTO, USEPA, or NETC specification for filter berm particle size distribution.

⁴Did not meet TX DOT, USEPA, or Filtrex International specification for filter sock particle size distribution.

ND = no data available

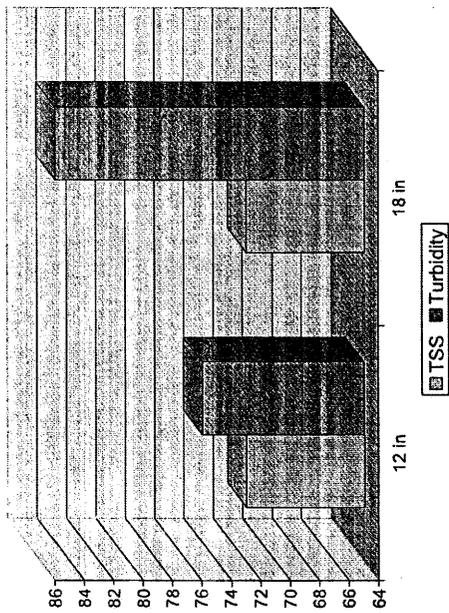


Figure 1--TSS removal efficiency and turbidity reduction for 12 in and 18 in filter socks.

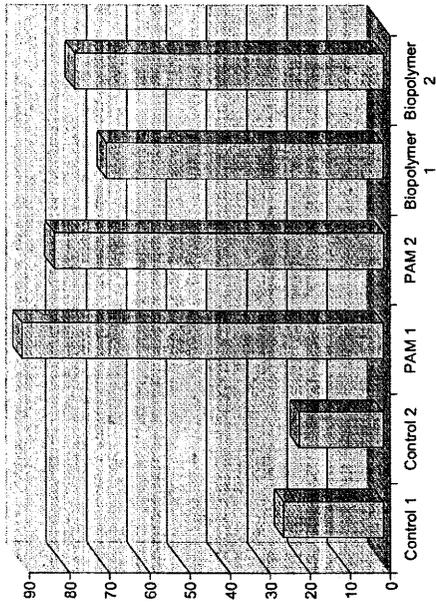


Figure 2--Percent turbidity (NTU) reduction from two consecutive runoff events with polymer added to filter socks.

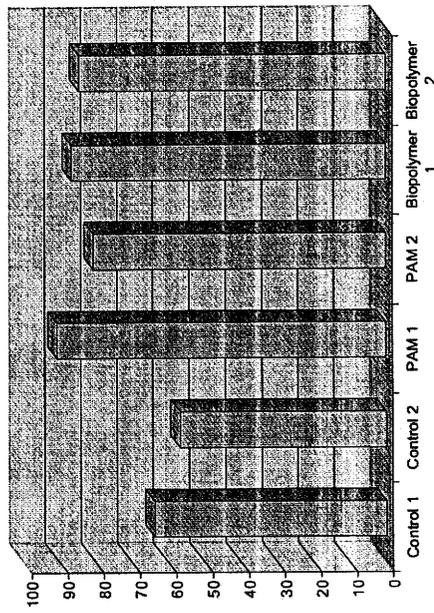


Figure 3--Percent TSS removal efficiency from two consecutive runoff events with polymer added to filter socks.

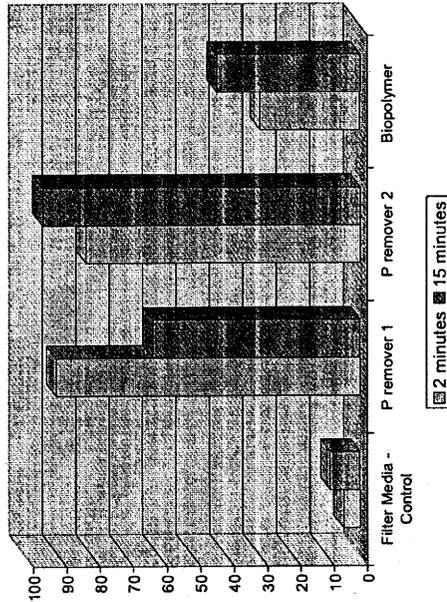


Figure 4--Soluble P removal efficiency from storm water runoff with polymer added to filter sock.

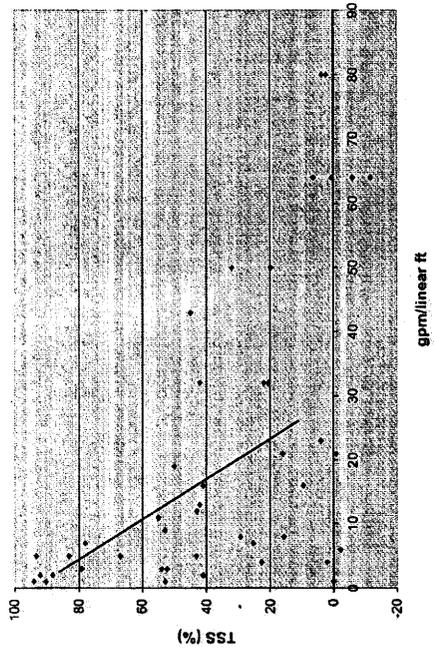


Figure 5--Hydraulic flow through rate of filter media relative to TSS removal efficiency of runoff.

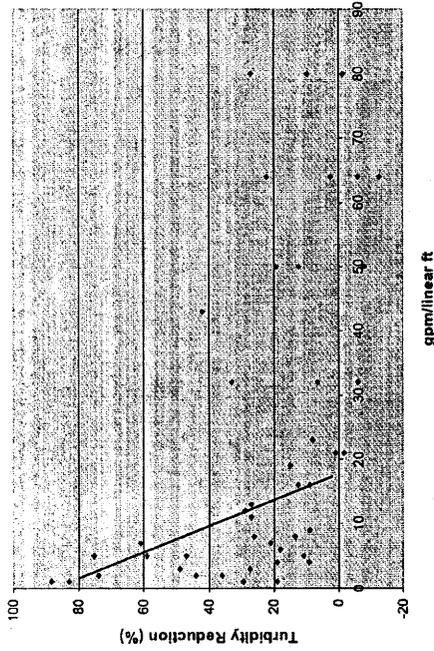


Figure 6--Hydraulic flow through rate of filter media relative to turbidity reduction in runoff.

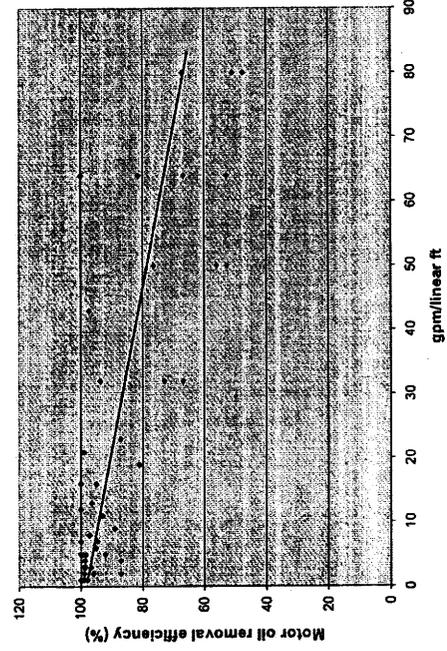


Figure 7--Hydraulic flow through rate of filter media relative to motor oil removal efficiency in runoff.

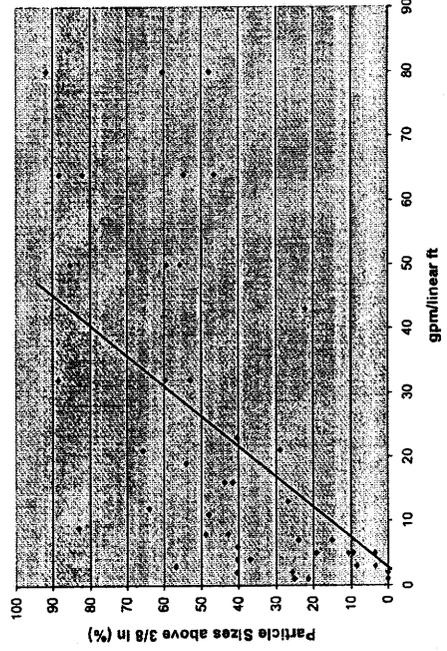


Figure 8--Hydraulic flow through rate relative to particle sizes above 3/8 in for filter media.

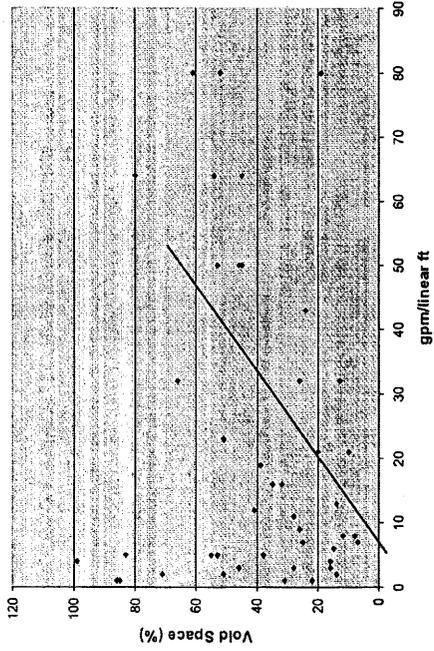


Figure 10--Hydraulic flow through rate relative to percent void space within filter media.

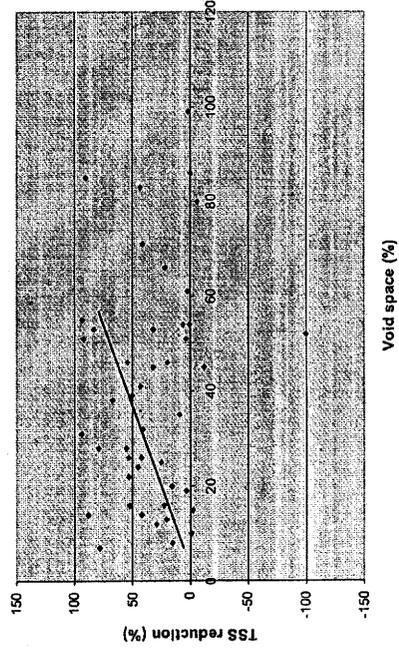


Figure 12--Void space of filter media relative to TSS removal efficiency in runoff.

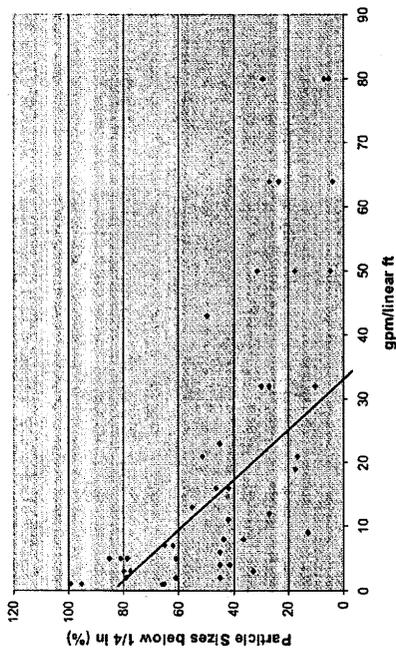


Figure 9--Hydraulic flow through rate relative to particle sizes below 1/4 in for filter media.

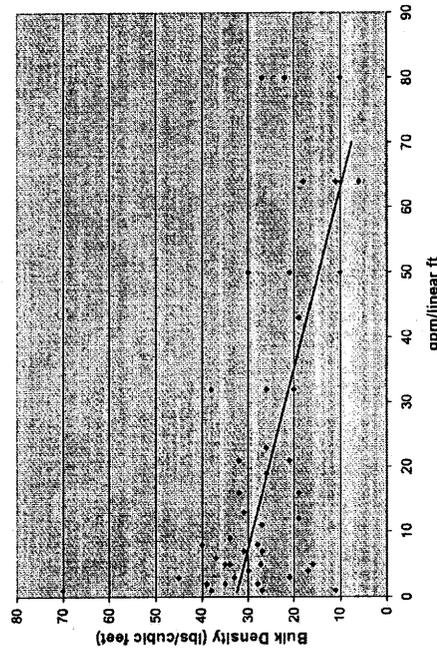


Figure 11--Hydraulic flow through rate relative to bulk density of filter media.

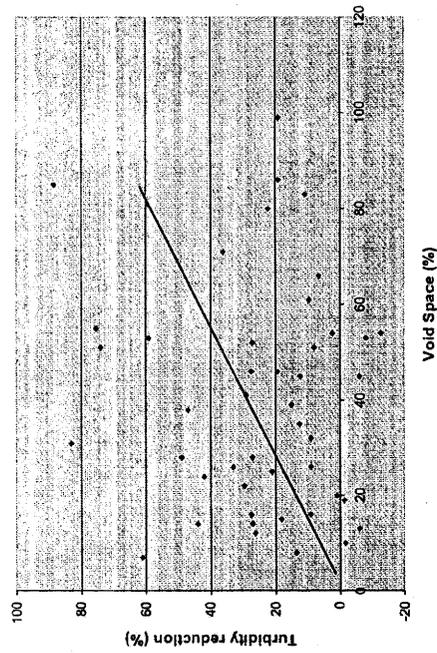


Figure 13--Void space of filter media relative to turbidity reduction in runoff.

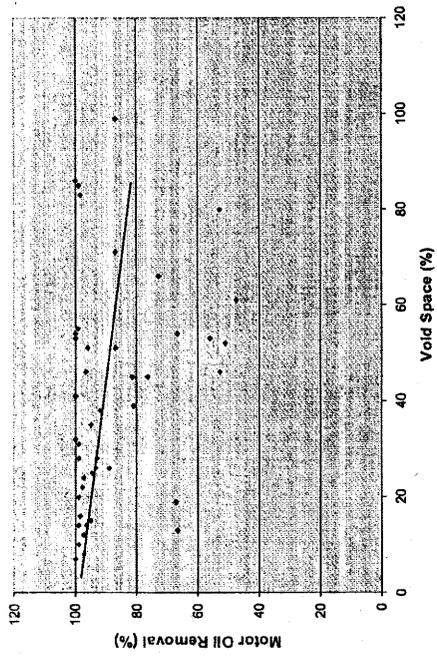


Figure 14--Void space of filter media relative to motor oil removal efficiency in runoff.

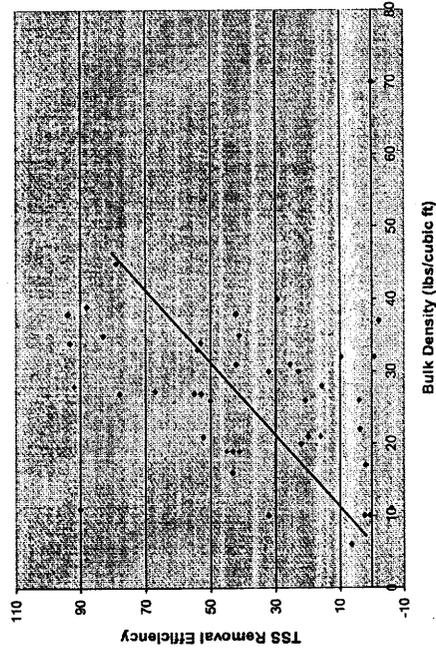


Figure 15--Bulk density of filter media relative to TSS removal efficiency in runoff.

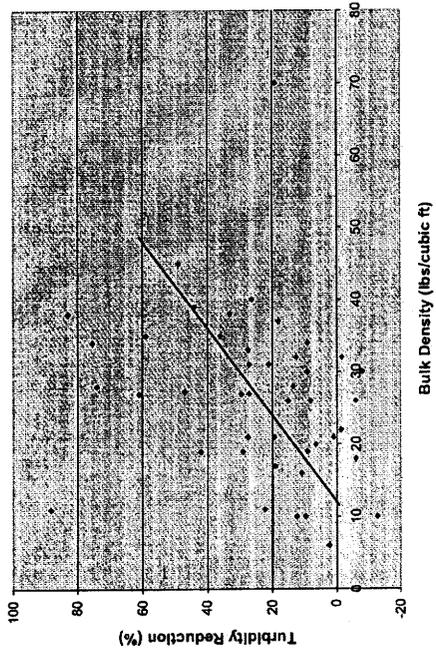


Figure 16--Bulk density of filter media relative to turbidity reduction in runoff.

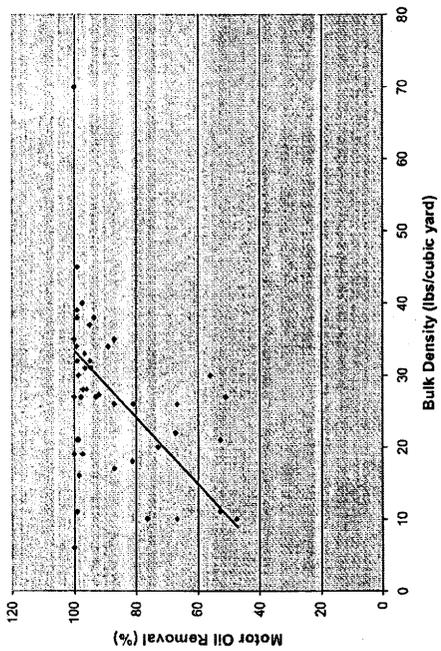


Figure 17--Bulk density of filter media relative to motor oil removal efficiency in runoff.

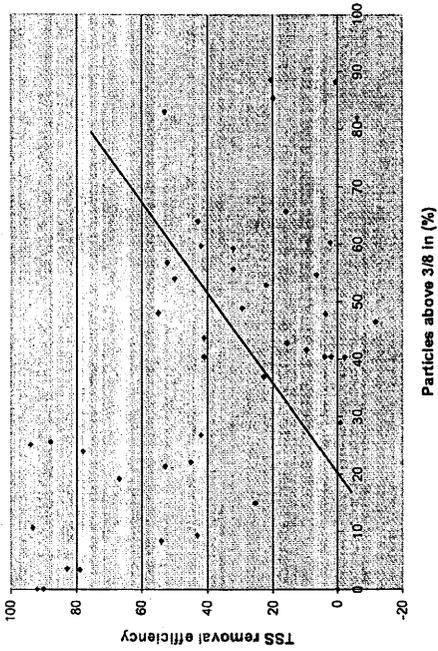


Figure 18--Particle sizes above 3/8 in of filter media relative to TSS removal efficiency in runoff.

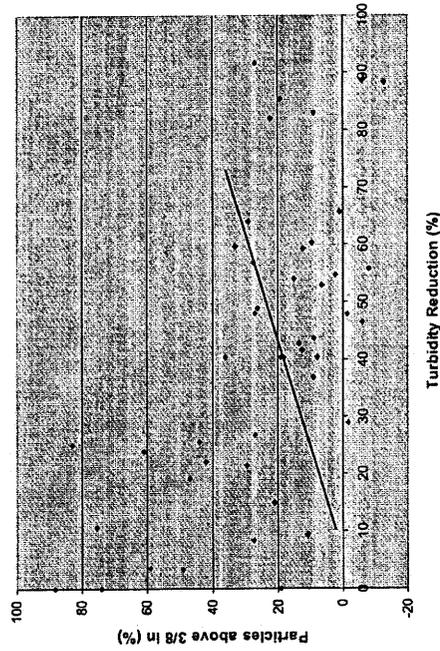


Figure 19--Particle sizes above 3/8 in of filter media relative to turbidity reduction in runoff.

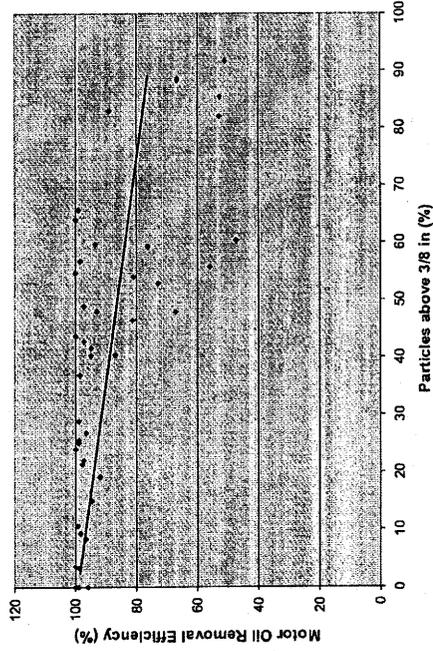
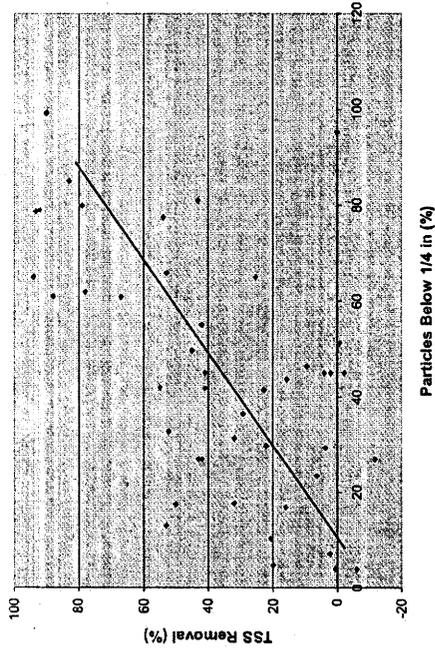
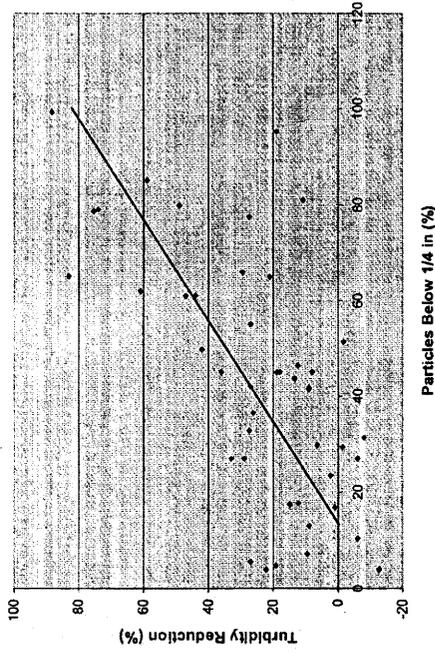


Figure 20--Particle sizes above 3/8 in of filter media relative to motor oil removal efficiency in runoff.



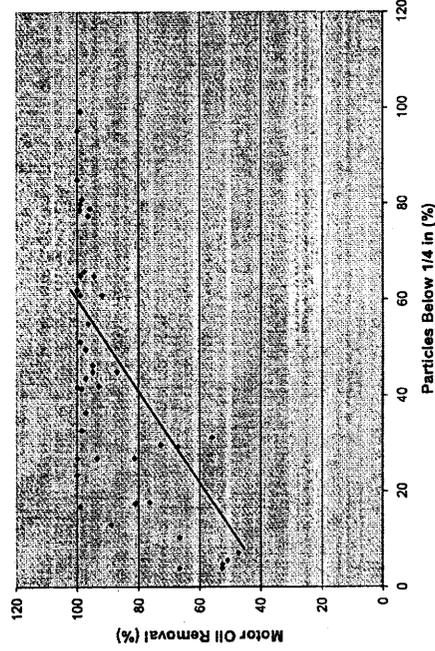
Particles Below 1/4 in (%)

Figure 21--Particle sizes below 1/4 in of filter media relative to TSS removal efficiency in runoff.



Particles Below 1/4 in (%)

Figure 22--Particle sizes below 1/4 in of filter media relative to turbidity reduction in runoff.



Particles Below 1/4 in (%)

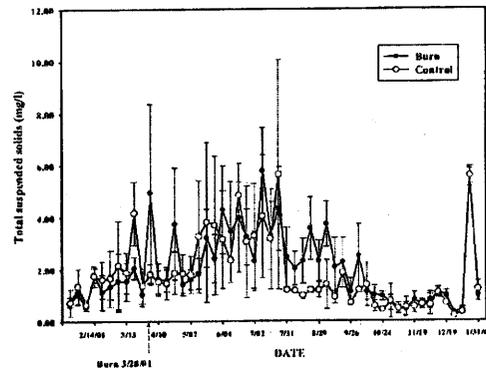
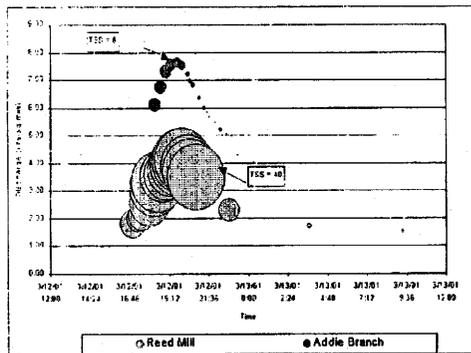
Figure 23--Particle sizes below 1/4 in of filter media relative to motor oil removal efficiency in runoff.



Second Interagency Conference on Research in the Watersheds

May 16 – 18, 2006

Long Term Data and Watershed Response to Disturbance



LONG TERM HYDROLOGIC TRENDS ON THE LITTLE RIVER EXPERIMENTAL WATERSHED

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Abstract-- The USDA-ARS, Southeast Watershed Research Laboratory (SEWRL) in Tifton, Georgia has continuously collected hydrologic and climatic data from the Little River Watershed since 1968. The data are representative of conditions throughout the low-gradient regions of the Southeastern U.S. Coastal Plain physiographic region. Hydrologic and climatic data are available from up to eight watersheds ranging in area from 2.6 to 334 km². Total streamflow from each of these watersheds were characterized for the period of record and compared to climatic patterns. Relationships and trends between observed precipitation and flow were examined. Basic statistical characterizations of annual precipitation and flow were determined. Long term hydrologic budgets indicate approximately 30% of the watershed precipitation becomes streamflow. Generalizations relating watershed yield to watershed drainage area illustrate differences between this region and other areas of the U.S.

INTRODUCTION

The Little River Watershed is in the headwaters of the Suwannee River Basin, a major interstate basin that begins in Georgia and empties into the Gulf of Mexico in the Big Bend region of Florida. The Suwannee River Basin is completely contained in the Coastal Plain Physiographic Region and is the largest free-flowing river in the Southeastern U.S. Coastal Plain. The Southeast Watershed Research Laboratory (USDA-ARS) has collected hydrologic and climatic measurements in the Little River Experimental Watershed (LREW) since 1968. The LREW is instrumented to measure rainfall and streamflow for the 334 km² drainage area and for seven subwatersheds that range from approximately 3 km² to 115 km² (Fig. 1). The experimental watersheds are located in a paired and nested arrangement that facilitates testing of analytical formulas and modeling concepts.

Instrumentation was installed in the late 1960's and early 1970's and has been in continuous operation since that time. Continued operation of this hydrologic network supports hydrologic research as well as environmental quality and riparian research programs of the SEWRL and cooperators.

Streamflow within the watershed varies considerably from other regions of the U.S. The LREW is located within the Tifton Upland of the Southern Coastal Plain physiographic region. The watershed is typical of low-gradient watersheds throughout the region. The Tifton Upland lies within the outcrop area of the Miocene series Hawthorn Formation, considered to be a continuous aquiclude for surficial aquifers in the region (Stringfield, 1966). Upland soils are primarily classified as fine-loamy (or loamy) siliceous, thermic Plinthic Paleudults (Calhoun, 1983), generally with infiltration rates in excess of 5 cm hr⁻¹ (Rawls and others 1976). Bottomland soils adjacent to drainage networks are primarily loamy, siliceous, thermic Arenic Plinthic Paleaquults with some Fluvaquents and Psammaquents (Calhoun, 1983). Drainage of bottomland soils is poor to very poor with standing water on the surface during significant portions of the year. River channel slopes

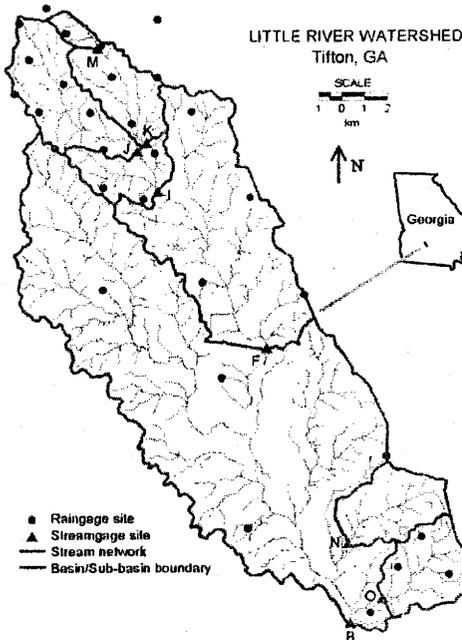


Figure 1--Little River Experimental Watershed of Georgia, U.S.

are generally less than 0.1%, whereas upland side slopes generally range up to 5% (Yates, 1976). These type watersheds have been classified as floodplain swamps (Kitchens and others 1975), floodplain wetlands (Kibby, 1978), seasonally flooded wetlands (Johnston and others 1984), forest wetlands (Leitman and others 1983), and blackwater swamp systems (Wharton, 1978).

Flow characteristics from these watersheds have been shown to be significantly different from other watersheds across the U.S. (Sheridan, 1997). Water yield is typically higher on a per unit area basis than that observed from many other areas in the U.S. The streamflow pattern is consistent with that of a drainage network that intercepts surface runoff and lateral subsurface flow from uplands and has low deep seepage losses (Sheridan, 1997).

The objectives of this research were to:

1. Examine trends in the streamflow characteristics from the LREW
2. Relate trends to physiographic traits of the region
3. Make comparisons between hydrologic patterns observed in the LREW to other regions throughout the U.S.

METHODS

Precipitation and streamflow data were used to examine the long term streamflow characteristics from a Coastal Plain Watershed. Observed climatic and hydrologic data from the LREW in Tifton, Georgia U.S. (Sheridan and others 1982; Sheridan, 1997) were used for the analysis. The watershed is instrumented to measure rainfall and streamflow within the main watershed, Little River B (LRB) as well as within seven subwatersheds (LRM, LRK, LRJ, LRI, LRF, LRN, and LRO) that range from 3 km² to 115 km² (Fig. 1, Table 1).

Table 1--Studied subwatersheds of the Little River Experimental Watershed.

Subwatershed	Years of record	Area (km ²)	Channel length (km)	Channel slope (%)	Length to width ratio (km km ⁻¹)	Drainage density (km km ²)
LRM	22	2.62	2.41	0.35	4.44	1.73
LRN	14	15.67	6.60	0.36	2.00	1.84
LRO	25	15.93	6.11	0.37	2.18	2.02
LRK	37	16.65	8.73	0.29	3.20	1.56
LRJ	37	22.12	10.30	0.25	2.71	1.60
LRI	37	49.91	12.71	0.22	2.24	1.67
LRF	36	114.84	24.02	0.14	2.67	1.64
LRB	34	334.33	39.10	0.10	4.41	1.60

Extensive land use information (Williams 1982, Perry and others 1999, Bosch and others 2006) and physical characterization data (Sheridan and Ferreira, 1992) exist for the LREW. The watershed land use is a mixture of row-crop agriculture, pasture and forage production, upland forest, and riparian forest. Sub-watersheds range from about 25% to about 60% tilled agricultural land. A detailed data management system exists to provide processing, editing, and summarization of LREW data (Sheridan, and others 1995). Rainfall in the region is poorly distributed and often occurs as short-duration, high-intensity convective thunderstorms (Bosch and others 1999). These thunderstorms promote runoff and erosion which may carry soluble and sorbed phases of applied nutrients and pesticides to lower landscape positions or into surface waters. Hydrologic and water quality measurements collected on the watershed include streamflow, precipitation, and nutrient, pathogenic bacteria, and pesticide content. The hydrologic measurement network consists of eight horizontal broad-crested weirs with v-notch center sections. Five-minute continuous upstream and downstream stage data are recorded. Within the watershed a network of 35

tipping bucket precipitation gages record five-minute cumulative rainfall (Fig. 1). The spacing between the precipitation gages varies from three to eight km. There is one NRCS SCAN site within the watershed (<http://www.wcc.nrcs.usda.gov/scan/index2.html>) and three University of Georgia meteorological stations. Rainfall, air temperature, relative humidity, incoming solar radiation, and wind speed and direction are measured at the SCAN and the University sites.

Streamflow and precipitation data collected from 1969 through 2004 were examined. Partial year data from 1968 were not used for this analysis. Precipitation data collected from the rain gages within and immediately surrounding each watershed were used to calculate an area-weighted daily precipitation (Sheridan and others 1994). Flow rates were calculated using stage-discharge ratings developed for each structure (Sheridan and others 1994). Flow volumes and rainfall depths are calculated on a per unit area basis using the watershed drainage areas. Additional details on the streamflow and precipitation data are available (Bosch and others 1999; Sheridan and others 2002).

RESULTS AND DISCUSSION

Over the 33-year observation period, the average annual weighted precipitation for LRB was 1215 mm with a coefficient of variation (CV) of 13.6% (Table 2). The average area-weighted flow for the same period was 334 mm with a CV of 44.5% and the average ratio of the annual flow to the annual precipitation was 0.27 with a CV of 36.4%. Streamflow exhibits a greater degree of variability than the precipitation. As noted by Sheridan (1997), seasonal variation in the precipitation has a large impact on the flow. In this region, evapotranspiration from May through October can be a major portion of the hydrologic budget. Precipitation received during the winter months when evapotranspiration is low generates a large percentage of streamflow whereas precipitation received during summer months generates little to no streamflow (Fig. 2).

Table 2--Summary statistics for the precipitation, area-weighted flow, and ratio of the flow to the precipitation for selected subwatersheds of the Little River.

Subwatershed	Precipitation		Area Weighted Flow		Ratio of Flow to Precipitation	
	Average (mm)	CV* (%)	Average (mm)	CV (%)	Average	CV (%)
LRM	1237	12.8	300	40.5	0.24	34.3
LRN	1158	28.0	350	38.0	0.30	24.7
LRO	1196	15.1	342	40.6	0.28	33.0
LRK	1241	15.6	390	44.5	0.30	34.5
LRJ	1242	15.5	408	48.9	0.32	38.1
LRI	1239	15.3	421	44.7	0.33	33.9
LRF	1230	14.4	365	41.5	0.29	31.5
LRB	1215	13.6	334	44.5	0.27	36.4

* CV - coefficient of variation

Average annual precipitation ranged from a high of 1242 mm observed in LRJ to a low of 1196 observed in LRO (Table 2). Above average precipitation years generally yield above average streamflow while years with below average precipitation generally yield less streamflow. Above average flow conditions were observed in the early 1990's while below average flow conditions were observed in the late 1990's and the early 2000's (Fig. 3). The high flow conditions of the early 1990's coincided with years of above average rainfall while the low flow conditions from 1999 to 2003 corresponded to a drought that occurred during that period.

There is some indication that both annual precipitation and annual streamflow for LRB have decreased over time (Fig. 3). The average annual precipitation has decreased from the 1247 mm reported by Sheridan

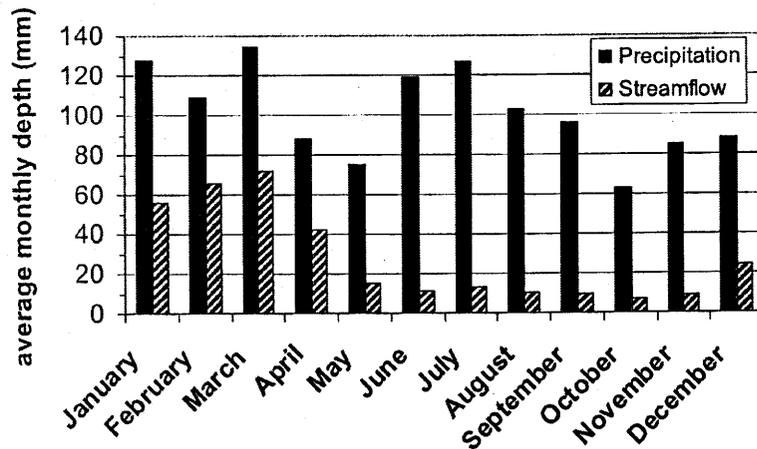


Figure 2--Average monthly observed precipitation and streamflow on LRB from 1972 to 2004.

(1997) for the period from 1972 to 1992 to 1215 mm for the same period extended to 2004. This may be heavily influenced by the drought that occurred from 1999 to 2003. A linear regression conducted using the annual data indicate that precipitation is decreasing more rapidly than flow. However, correlation coefficients for the linear regressions were low ($r^2 < 0.11$) and the linear decreases were not statistically significant ($\alpha = 0.05$) for any of the examined watersheds (Table 3).

The observed ratio of annual measured streamflow to watershed weighted annual precipitation for LRB varied from 0.06 (1981) to 0.41 (1998) (Fig. 4). The average ratio for LRB was 0.27 (Table 2). As was found with precipitation and streamflow, over the 34-year observation period there has been no statistically discernable temporal trend in the ratio of annual streamflow to annual precipitation for LRB ($\alpha = 0.05$). Likewise, no statistically significant temporal trends in the ratios for the subwatersheds were found.

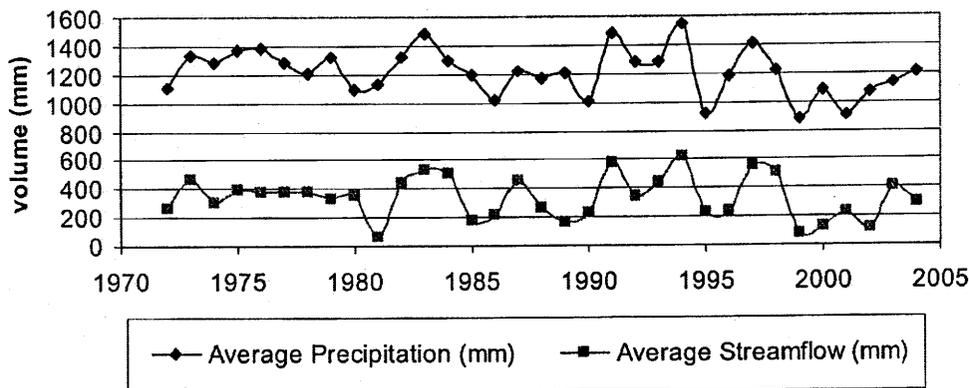


Figure 3--Area-weighted annual precipitation and flow observed on LRB from 1972 to 2004.

A significant correlation ($\alpha = 0.01$) was found between annual precipitation and streamflow for each examined watershed (Table 3). The slopes for the linear regressions were found to vary from 0.62 for LRO to 0.90 for LRJ (Table 3). Watersheds LRJ, LRK, and LRI were more responsive to increases in annual precipitation than was watershed LRO or the entire LRB. For LRB, linear regression using annual precipitation to predict annual flow accounts for 61% of the variability in annual flow. However, examination of annual streamflow data as a function of precipitation for LRB indicates that there are several years where the data did not follow the typical trends (Fig. 5). Within the LREW, the most significant precipitation occurs in the months from December through March and from June through

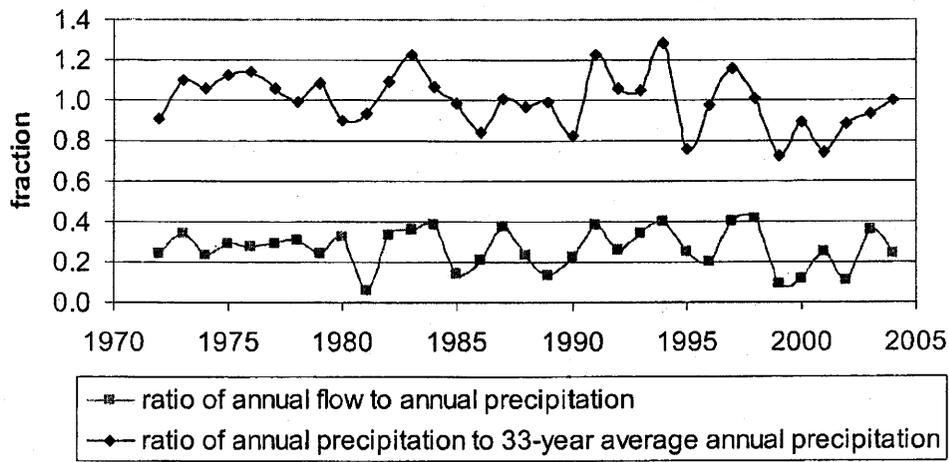


Figure 4--Ratio of observed annual flow to annual precipitation and annual precipitation to the 33-year average annual precipitation observed on LRB from 1972 to 2004.

August, while the most significant streamflow occurs during the months from December through April (Fig. 2). The data points which fall below the typical trend between annual precipitation and annual streamflow tend to be years with low winter rainfall while points which fall above the typical trend tend to be years with high winter rainfall. Seasonal timing of precipitation is an important factor in determining annual flow from the watershed. Summer rainfall does not generally have a large impact on annual streamflow. When greater precipitation is received in January through April, greater annual flow can be expected. Multiple linear regression including the precipitation from the previous year December through the current year March along with the annual precipitation to predict annual flow depth accounts for 76% of the variability in the annual flow versus the 61% accounted for with the annual precipitation alone. Watersheds that received greater annual precipitation (LRJ, LRK, LRF), also received greater winter rainfall. Thus, the ratio of streamflow to precipitation on these watersheds was greater (Table 2).

Table 3--Regression characteristics for annual precipitation as a function of time, annual flow as a function of time, and annual flow as a function of precipitation.

Watershed	Precipitation as a function of year			Flow as a function of year			Flow as a function of precipitation			Threshold Rainfall (mm)
	Slope*	r ² **	P***	Slope	r ²	P	Slope	r ²	P	
LRO	-5.58	0.15	0.06	-2.37	0.05	0.31	0.62 a	0.64	1.36E-06	642
LRK	-4.46	0.06	0.15	-2.22	0.02	0.43	0.77 ab	0.74	1.79E-11	735
LRJ	-4.96	0.07	0.11	-2.06	0.01	0.53	0.90 b	0.75	7.37E-12	789
LRI	-4.85	0.07	0.11	-2.32	0.02	0.45	0.85 ab	0.74	1.56E-11	746
LRF	-4.43	0.07	0.12	-1.78	0.02	0.47	0.74 ab	0.73	2.68E-11	733
LRB	-5.68	0.11	0.06	-2.25	0.02	0.42	0.70 ab	0.61	9.10E-08	739

* slope of the linear regression equation

** correlation coefficient of the linear regression

*** probability of the significance of the linear regression, values less than 0.05 indicate significant linear correlation ($\alpha=0.05$)

Values followed by the same letter within a column are not significantly different at $\alpha < 0.05$ using the analysis of covariance test.

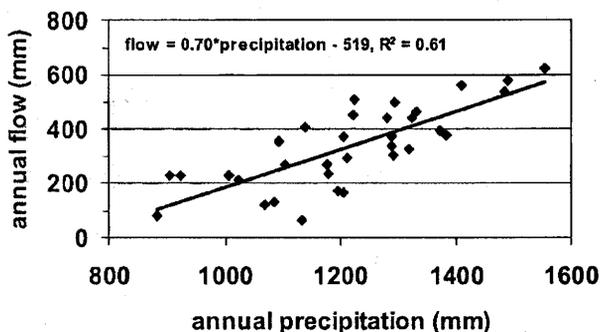


Figure 5--Relationship between annual precipitation and annual flow for LRB.

Sheridan (1997) evaluated a threshold rainfall, a value indicative of the annual watershed evapotranspiration requirements that must be satisfied before significant streamflow is produced. These values are equivalent to the intercept divided by the slope of the linear regression between annual precipitation and annual flow for any given watershed. The threshold rainfalls found for this study varied from 642 mm for LRO to 789 mm for LRJ (Table 3). These values are similar to those reported by Sheridan (1997) for the period from 1972 to 1992.

Comparison of subwatersheds within LREW indicates some hydrologic differences (Tables 2-3). These differences may be related to differences in land-use and physical characteristics. Flow as a response to rainfall relationships was found to be statistically different for subwatershed LRO than from similar watersheds in the northern portion of LREW. LRO lies in the southern portion of LREW and has approximately 53% forest coverage whereas the northern watersheds have approximately 61% forested coverage (Bosch and others 2006). LRO produces less streamflow as a response to rainfall than either LRJ or LRK in the northern portion of the watershed. The proportion of precipitation that flows from LRO is 28% while it is 30% for LRK and 32% for LRJ. It appears that these differences are related to differences in seasonal rainfall rates, land-use, and physical characteristics of the watersheds.

Data from the Little River Watersheds were combined with flow data from nine other watersheds throughout the Southeastern Coastal Plain to develop a generalized watershed area to annual flow response curve for the Coastal Plain Region (Fig. 6). The watersheds ranged from 0.34 ha to 3626 km² (Table 4). The relationship was compared to established curves for other regions using data presented by Renard (1977) and updated by Sheridan (1997) (Fig. 6). As shown by the illustration, the Coastal Plain watersheds generally yield more water on a per unit area basis than do watersheds from other regions. The water yield relationship for this region is indicative of a drainage network that intercepts surface runoff and shallow groundwater flow from the uplands and has low deep seepage losses. This is dramatically different from regions such as those in Texas and Arizona where significant portions of the streamflow are lost to deep seepage out of the stream channel.

CONCLUSIONS

Precipitation and flow patterns within the LREW over the 33-year hydrologic observation period appear to be fairly stable. There has not been a statistically significant temporal trend in the ratio of the annual flow to the annual precipitation for this watershed (Fig. 4). The long term average annual ratio has remained stable at approximately 0.27 for LRB. Year-to-year fluctuations due to variations in precipitation are large. There is no indication from the observed annual flow data that land-use changes implemented within LRB have significantly impacted the streamflow. Annual and seasonal precipitation amount and timing have the greatest overall impact on streamflow. Generalizations relating watershed yield to watershed drainage area illustrate large differences between watersheds in this region and those in other regions of the U.S. Watershed yield as a function of area tends to be independent of the area, indicating very little percolation loss out of the channel system.

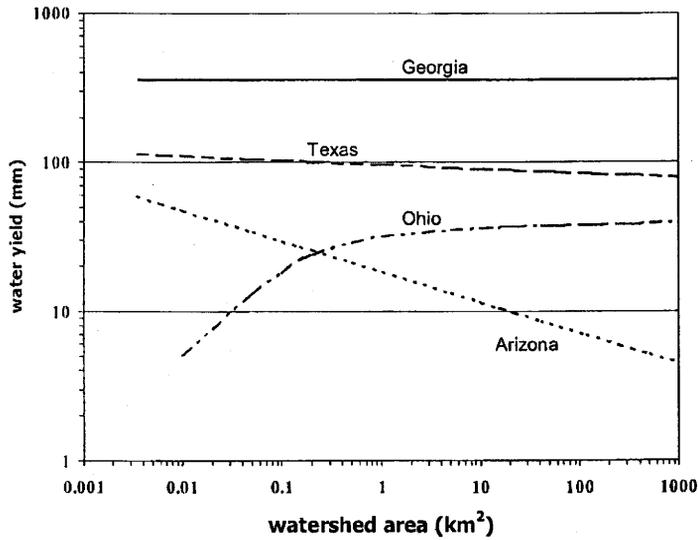


Figure 6--Relationship between watershed yield and area for several regions in the U.S.

Table 4. Watersheds used to develop the generalized watershed area to annual flow response curve for the Coastal Plain Region.

Site	Area (km ²)	Average Annual Flow (mm)	Ratio of Annual Flow to Annual Precipitation	Data Source
Station Z, Tifton, Georgia	0.003	361	0.29	Shirmohammadi and others (1984)
Ahoskie Creek WA1, NC	147.6	395	0.37	SEWRL (1977)
Ahoskie Creek WA2, NC	62.2	339	0.33	SEWRL (1977)
Ahoskie Creek WA3, NC	9.6	265	0.25	SEWRL (1977)
Taylor Creek W3, FL	40.7	316	0.25	Knisel and others (1985)
Taylor Creek W2, FL	255.6	443	0.34	Knisel and others (1985)
Alapaha River at Statenville, GA	3626	397	0.33	USGS*
Withalacoochee River at Bemis, GA	1300	356	0.29	USGS*
Little River at Adel, GA	1494	302	0.25	USGS*
Okapilko River at Quitman, GA	697	319	0.26	USGS*

* Data obtained from the U.S. Geological Survey, <http://waterdata.usgs.gov/nwis>

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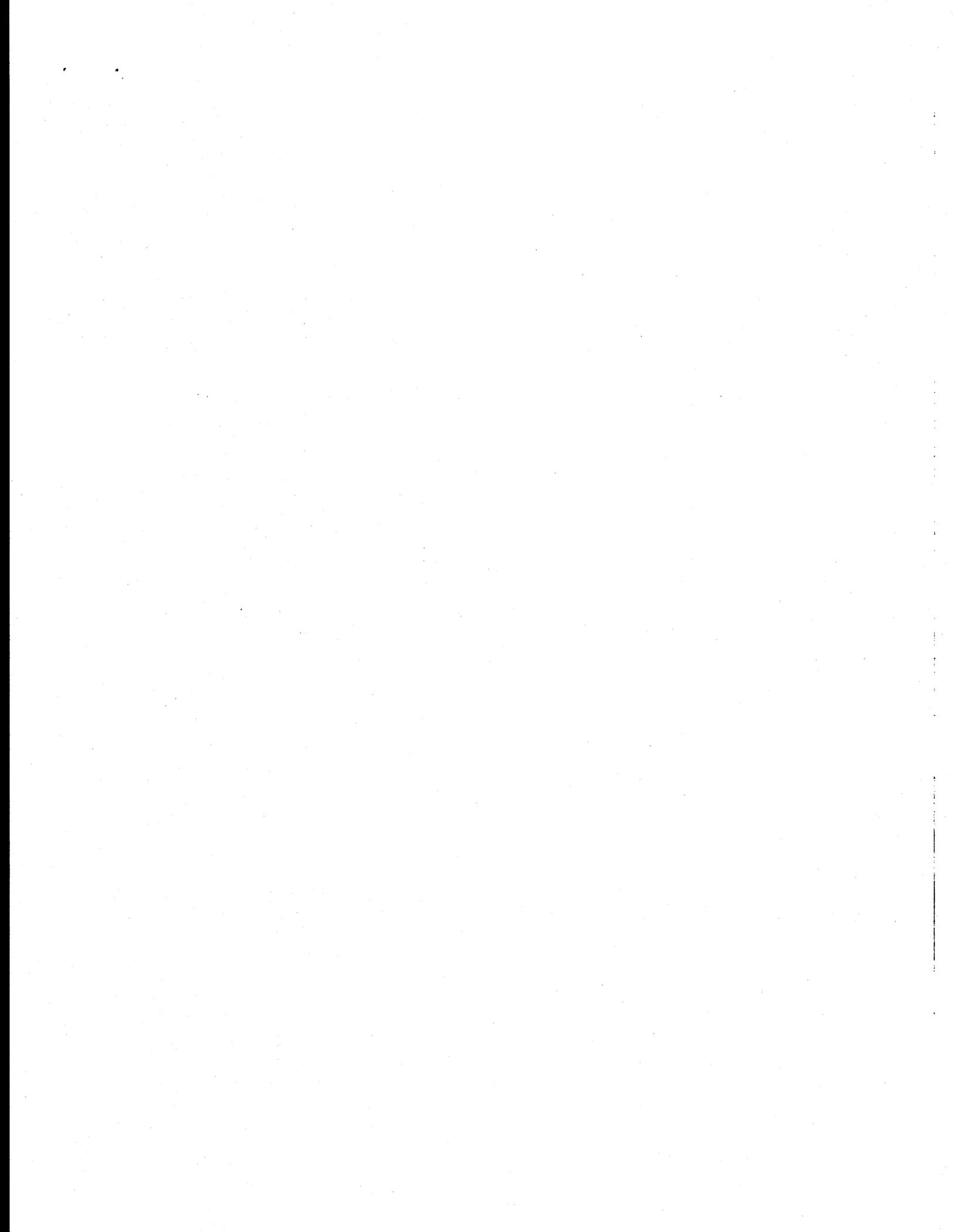
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QUANTIFYING TRAIL EROSION AND STREAM SEDIMENTATION WITH SEDIMENT TRACERS

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Abstract--The impacts of forest disturbance and roads on stream sedimentation have been rigorously investigated and documented. While historical research on turbidity and suspended sediments has been thorough, studies of stream bed sedimentation have typically relied on semi-quantitative measures such as embeddedness or marginal pool depth. To directly quantify the impacts of a functioning off-highway vehicle (OHV) trail on stream sedimentation, we employed a marked-recapture sediment tracer approach that allowed us to directly measure the movement of sand eroded from the trail and transported through the stream. We seeded a controlled section of an operating OHV trail with manufactured limestone sand (MLS). Fine fractions of the MLS were washed from the road and increased stream water calcium concentrations, [Ca²⁺]. Stream water [Ca²⁺] began to return to pre-treatment levels within 12 weeks. Coarser fractions, greater than 0.5 mm, were eroded from the road with rain events and moved along the study reach in pulses. Much of the coarse sediment appeared to be within the study reach eight weeks following application of the tracer. Tracer results and estimated stream bed sediment transport times indicated the small section of OHV trail had contributed at least 2.45 kg (302 kg/ha) of coarse sediment to the stream bed in 8 weeks (1,960 kg/ha/yr).

INTRODUCTION

Multiple use management of USDA Forest Service (USFS) National Forests allows for public use of off-highway vehicles (OHV) on designated trails. The National Forests in the southern Appalachian Mountains are within a few hours drive for millions of potential users and provide a wide selection of OHV recreation opportunities on hundreds of kilometers of trails. Roads have been identified as a significant source of sediment in southern Appalachian streams (Riedel, and others, 2003) however, the influences of OHV trails and OHV use on stream sedimentation have not been documented. While OHV trails are similar to roads, OHV trails have not been regularly maintained. This is an important difference in the southern Appalachian Mountains where average annual rainfall often exceeds 230 cm per year (Riedel, 2006a) and fine grained micaceous soils are highly erosive (Van Lear and others, 1995). Although anecdotal evidence indicates OHV trails cause significant soil erosion and stream sedimentation in the southern Appalachian Mountains (Figures 1 and 2, Riedel, 2006b), the long-term effects of OHV trails on stream bed sedimentation have not been documented. We conducted a research trial of a sediment tracer methodology on an OHV Trail in the Nantahala Mountains of northeastern Georgia. The purpose of this study was to test the development, application, and utility of using manufactured limestone sand (MLS) as a coarse sediment (0.5mm to 2mm) tracer. Preliminary results are reported.



Figure 1: Typical erosion of an OHV trail in the southern Appalachians.

Figure 2: Resultant stream sedimentation - sediments included fine to coarse grained sand (0.5mm – 2mm).

METHODS

A portion of the USDA Forest Service Oakey Mountain OHV trail system in the Chattahoochee National Forest of NE Georgia was selected for this study (Figure 3). Though not closed during the study, trail use was typically limited to small groups of a few riders during weekends because of the remote site location and difficult site access. Vehicles driven on the trail included "4-wheelers", "moto-cross bikes", and the 4-wheel drive "mule" used by study personnel; large boulder and log obstructions at the visitor access points (approximately 45 minutes away by OHV travel) prevented access by larger vehicles. A field inspection of the watershed revealed no other trails, roads, human induced forest disturbance, or illegal trails affected the study site. A ford allowed OHV users to cross the stream at a low point in the trail (Figure 4).

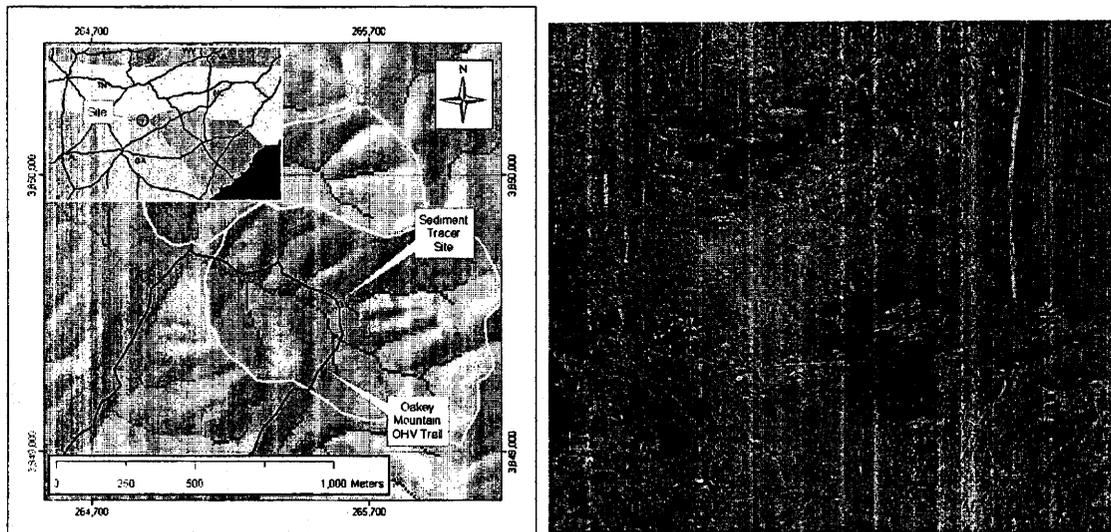


Figure 3: Location of OHV sediment tracer study site. The OHV trail loop delineated by the black line is part of the Chattahoochee National Forest Oakey Mountain OHV trail system.

Figure 4: Study site showing OHV trail and stream ford.

Discharge, Precipitation, and Stream Water Chemistry

Site location and installation of an automated stream pumping sampler in June, 2005 followed standard federal protocols (Wagner, and others, 2000). A pressure transducer, used to log stream stage, was placed in a PVC stilling well to minimize interference from wave action. The sampler logged stage on ten minute intervals. Stage readings were validated weekly by manually surveying stage and measuring discharge (Buchanan and Somers, 1969). A stage-discharge regression was developed and used to convert stage data to time series discharge data. Precipitation data were obtained from the National Weather Service weather station in Clayton, GA; this was approximately 6 km northeast of the study site. Flow data were processed following standard protocols (Riedel and others, 2004, Hibbert and Cunningham, 1967). The sampler was also used to collect stream water samples on a flow proportional basis; sampling frequency increased with flow. The fixed sampling inlet was anchored to a 1 m rebar pin in the streambed. While sampler capacity was limited to 24 one liter samples, composite sampling was used to draw four discrete, 250ml samples per bottle at a proportionately higher sampling frequency. Between storm events, an average of four to six one liter composite samples (16 to 24 250ml samples) were collected daily. Water quality samples were analyzed at the USDA Forest Service Coweeta Hydrologic Laboratory using a Perkin Elmer 2100 Atomic Absorption Spectrophotometer and following standard methods for total cation determinations (K^+ , Na^+ , Ca^{2+} , Mg^{2+}) (Deal and others, 1996).

Trail and Stream Bed Sediment

We used a rod and self leveling-transit to survey the stream and trail. The trail descended to the stream from both valley sides and crossed at a ford (Figure 5). Average trail slope was 14 % and the runoff contributing area was 0.0081 ha. The stream survey began upstream of the trail crossing and continued downstream to the confluence with Raper Creek (Figure 6). Perpendicular stream channel cross sections were surveyed at 7 m intervals along the stream. The first cross section, zero, was upstream of the ford. Subsequent cross sections began immediately

downstream of the trail ford (cross section 1) and continued to the sampler just upstream of Raper Creek. The stream featured step-pool morphology and an average slope of 6 %.

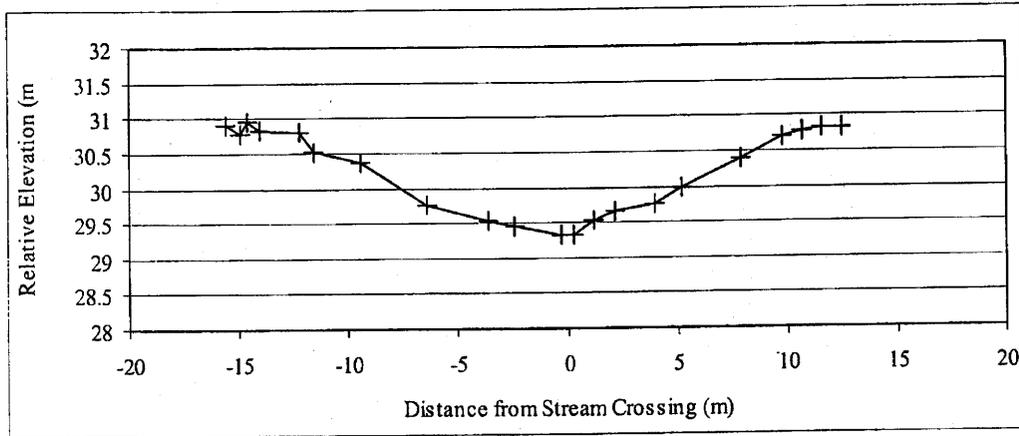


Figure 5: Survey of OHV Trail showing location of stream crossing at zero. The crests at each end of the trail determined the area of the trail that contributed runoff to the stream.

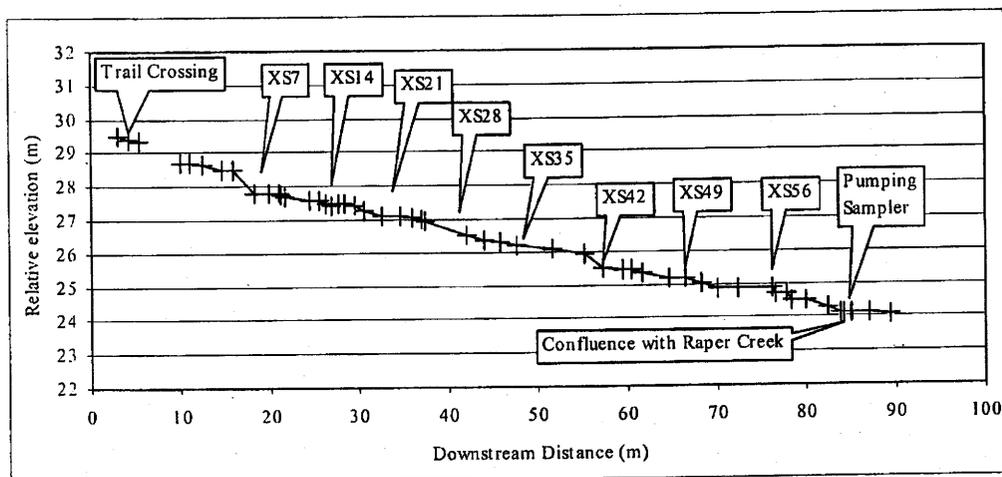


Figure 6: Longitudinal survey of study reach showing approximate locations of sediment sampling locations, trail crossing and automated pumping sampler. Average stream slope was 6%.

Samples of sediment were gathered from the trail and stream before and after the placement of the MLS sediment tracer (described below). Trail sampling was conducted by gathering replicate cores along the trail, on both sides of the ford. Stream bed sediment samples were composites of five replicate samples at each cross section. These consisted of one sample at the cross section with four others spaced at 1 m intervals upstream and downstream of the cross section. Hence, the samples represented 5 m long sections of the streambed centered on each cross section. Each of the five individual samples at a cross section was collected as a series of smaller samples taken across the width of the stream bed. Sediment sample volume was one liter and samples were taken to a depth of approximately 10 cm. Initial sampling was conducted immediately before the placement of the MLS on July 15, 2005. Subsequent samples were gathered following storm events. While sampling continued through December, 2005, analyses have only been completed through September. Laboratory analyses of the sediment samples provided calcium concentrations ($[Ca^{2+}]$, the "tracer") in parts per million (ppm).

We used [Ca²⁺] to compute stream bed sediment budgets and bed material residence times. This allowed for the determination of OHV induced stream bed sedimentation magnitude and duration in the study reach. The computational methods used mass conservation as applied in "marked capture-recapture" studies (e.g. soil erosion or wildlife population studies, see Zhang and others, 2001, Bergstedt and Bergersen, 1997, Wilcock, 1997, Bavley, 1993, Arkell and others, 1983) and required typical assumptions;

1. Full mixing of MLS (tracer) and "native" OHV trail sediments,
2. equal mobility of similarly sized MLS and "native" OHV sediments hence,
3. "native" sediment and MLS left the trail in direct proportion to the seeding rate (concentration).

The percentage of stream bed sediments that originated from the OHV trail was computed as the ratio of post-seeding stream bed [Ca²⁺] to the average background [Ca²⁺] at the site. This percentage was divided by the initial MLS tracer concentrations on the OHV trail to estimate the mass of sediments in the stream that originated from the OHV trail. Bed material residence times were estimated as the amount of time it took the initial MLS tracer pulses to be detected at each transect.

MLS Sediment Tracer

MLS was employed as a tracer in this study for two reasons. First, the [Ca²⁺] of the MLS was orders of magnitude higher than background levels from highly weather feldspars found in OHV trail, stream bed and soil samples. This degree of difference has been necessary for a tracer to provide sufficient enrichment and allow differentiation between introduced and background sediment sources (Zhang and others, 2001). Field sediment samples were analyzed for [Ca²⁺] by hot-plate/hydrogen peroxide consumption. Numerous samples of the MLS, estuarine standards, and blanks were analyzed during the study to characterize calcium content and to document the reliability and stability of the laboratory methodology and MLS tracer. From six independent sets of analyses spread over 6 months, the MLS and estuarine standard averaged 23.5 % and 0.32 % calcium, respectively ($n = 25, 15$; $s_x = 1.2 \%, 0.05 \%$). Background [Ca²⁺] from the stream, trail and soils ranged from 0.2 to 0.3 %.

Second, the particle size distribution of the MLS and particle density of the coarse fraction (0.5 mm to 2 mm sand) were very similar to that of the native soils and stream bed sediments. This was an important requirement as similitude between introduced MLS tracer and native sediments was necessary to maintain similar hydraulic behavior, sediment availability and transport characteristics. The fraction of fines in the MLS was intermediate between those of the stream bed and OHV trails sediments.

The MLS was scattered (seeded) on the OHV trails with a spreader and gently spread to assure uniform distribution. While efforts were made to minimize disturbance of the trail surface, OHV traffic was frequent enough to maintain a disturbed soil surface on the trail and mix the MLS into the trail surface. While no OHV traffic occurred during spreading of the tracer, it did occur immediately afterward and during subsequent site visits. Sediment samples collected before and immediately following application of the MLS verified uniform application.

RESULTS

Discharge, Precipitation, and Stream Water Chemistry

Stream flow and precipitation were typical for the region with a few larger storm events interspersed among numerous small storm events (Swift and others, 1989). While streamflow generally responded to precipitation events, there were a couple of localized rain events not captured by the rain gauge (Figure 7). Average post-treatment, stream water [Ca²⁺], 1.00 mg/l, were higher than pre-treatment, 0.65 mg/l ($p < 0.01$). During relatively large events, stream water [Ca²⁺] decreased during storm flow and then increased significantly after the events.

Bed Material Sediment

[Ca²⁺] of trail and stream bed sediments responded to the application of the MLS (Figure 8). Pre-treatment stream sediment [Ca²⁺] averaged 2.56 ppm. Post-treatment [Ca²⁺] varied significantly with distance from the trail. Spikes in [Ca²⁺] three orders of magnitude greater than background occurred near the OHV trail whereas no [Ca²⁺] response was observed farthest downstream. For the first few weeks following tracer application, stream bed [Ca²⁺] was elevated above background levels in only the first four transects. An increase in both magnitude and the number of transects with elevated [Ca²⁺] occurred following relatively large rain events in August. Upstream [Ca²⁺] before and after tracer application were not different.

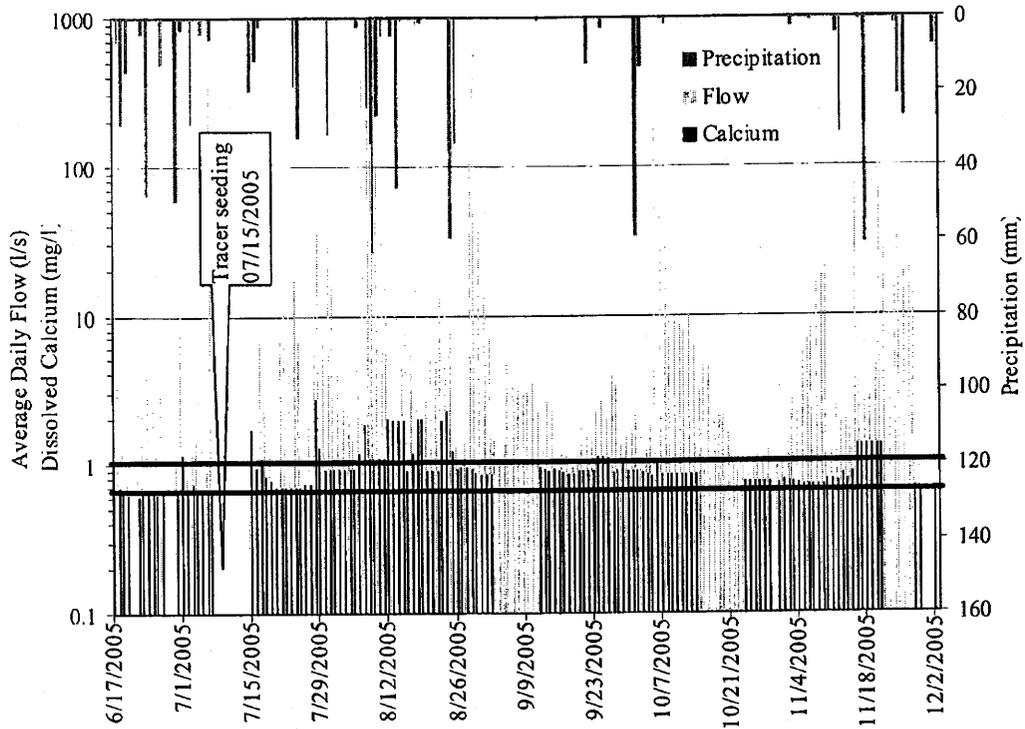


Figure 7: Precipitation, discharge and flow weighted Calcium concentration during the period of the study. Limestone tracer sand was applied 7/15/2005. Black lines at 0.65 mg/l and 1.00 mg/l denote average pre-treatment and post-treatment calcium concentration in stream water.

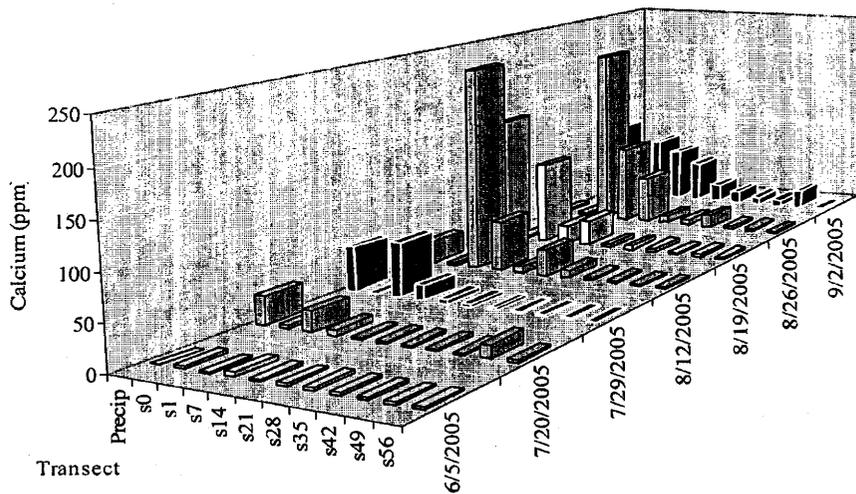


Figure 8: Total storm precipitation and post-treatment stream bed sediment calcium concentrations by sampling date. Transect s0 represents natural background concentrations.

The cumulative stream bed sedimentation attributable to the MLS ranged from 400 g to over 600 g per event (Figure 9). Declines of MLS in the stream beds on July 27th, August 19th and September 2nd were precluded by large rain events. These events removed more MLS from the stream bed than was introduced from the OHV trail. The cumulative storage of OHV sediments within the stream bed fluctuated as sediments were alternately transported into and out of the study reach (Figure 10). Given the original MLS application rate on the OHV trail was 24.5% and the previously mentioned assumptions, the OHV trail produced at least 2.45 kg of the observed stream bed sedimentation during this study. This is equivalent to 302 kg/ha or 1,900 kg/ha/yr of coarse sediment yield (0.5 mm to 2 mm) to the stream bed.

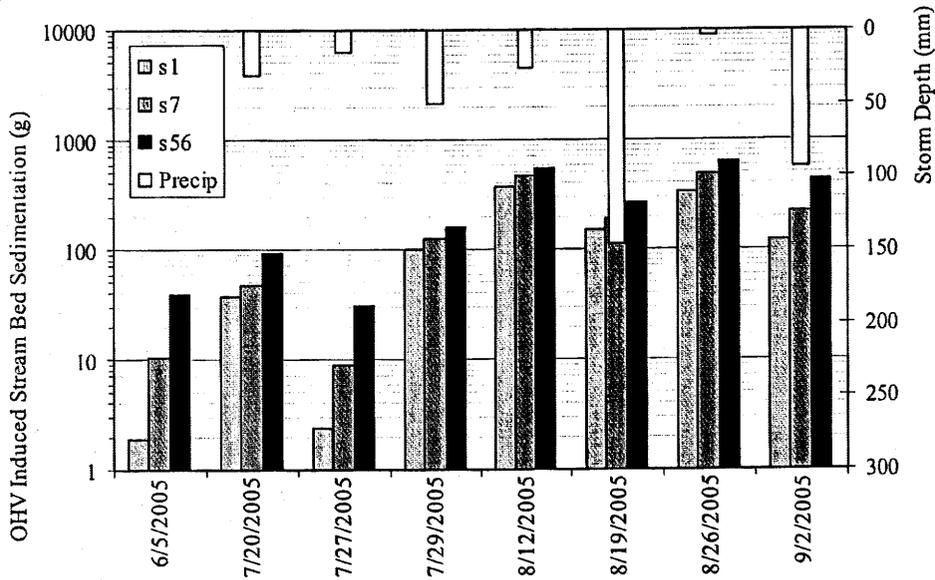


Figure 9: Cumulative stream bed sedimentation along the study reach as estimated from Calcium concentrations in stream bed sediments. Values do not include sediment that passed through the entire length of the reach during a single storm event.

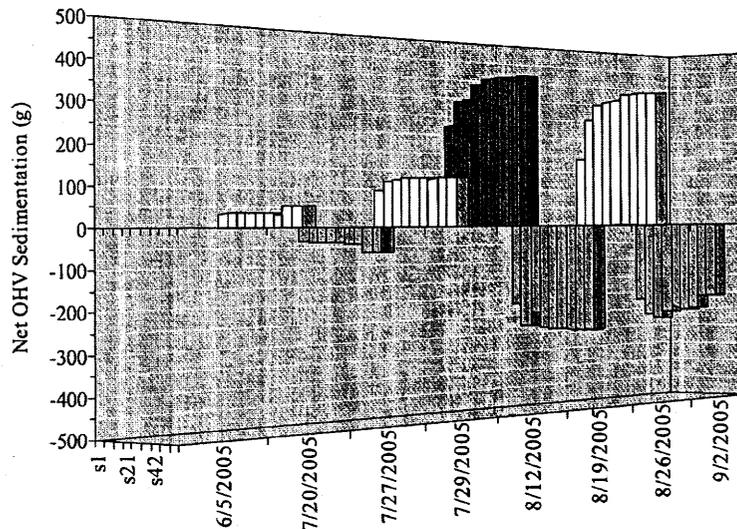


Figure 10: Cumulative change in OHV induced stream sedimentation along the study reach as estimated from changes in MLS concentrations. Positive values indicate net influx of OHV sediments while negative values indicate net efflux of OHV sediments.

DISCUSSION

Discharge, Precipitation, and Stream Water Chemistry

Precipitation, discharge and stream water [Ca²⁺] data generally followed expected patterns following the introduction of the MLS tracer; discharge and [Ca²⁺] generally increased following rain events. Relative to the duration of increased discharge in response to storm events, the spikes in stream water [Ca²⁺] were very short in duration. This could most likely be attributed to the short period over which overland flow from the OHV trail was running into the stream. With the cessation of precipitation, no additional MSL would be introduced to the stream even though discharge remained elevated. Following this spike, [Ca²⁺] declined rapidly. After about six weeks, stream water [Ca²⁺] began to decrease as discharge increased and only returned to higher levels as flows subsided. It is likely that by this time, the finer fractions of the MLS had been washed from the OHV trail and flushed through the stream. Thus, the study reach began to exhibit more typical water quality response to storm events in this region – a dilution of cations as the relative proportion of stream flow sources shifted from ground water to quickflow fed by runoff and shallow soil water. There were a couple of storm flow events unassociated with measured precipitation, and vice versa. This was due to the highly variable spatial distribution of late summer storm events in the mountains; while the rain gauge was a relatively short distance from the study site, it was not within the immediate area.

Bed Material Sediment

Sedimentation of the streambed in response to the MLS tracer application developed over time and occurred in pulses. Runoff from individual rain events washed new sediments into the stream while those from previous storms were transported short distances down stream or flushed entirely from the reach. As no MLS was detected immediately downstream of the junction between the study reach and Raper Creek, these events must have flushed the MLS sediments beyond the study boundary. Despite these large flushing events, additional MLS was still being transported from the OHV trail to the stream at rates approximately 400 times greater than background levels (100 ppm vs. 0.25ppm). This was evident as subsequent pulses of the MLS were detected in stream bed sediments. This result suggested there was a mechanism that influenced the availability of coarse sediments for erosion from the trail. The Author suggests the most likely mechanism would be OHV traffic and disturbance of the trail surface.

While one potential use of the MLS method would be to estimate bed material transport rates, sampling intervals were not frequent enough given the highly dynamic nature of the stream bed. With the efflux of OHV sediments out of the stream, it was not possible to discern the MLS mass centroid needed to estimate transport rates and residence.

CONCLUSIONS

The results of this study were preliminary in nature as they have not been replicated. Despite this, they indicated the OHV trail was having enormous impacts on water quality, sediment yield and stream bed sedimentation in the study reach. Most importantly, the use of MLS as a sediment tracer showed great promise as a tool to document soil erosion impacts on stream water quality and stream bed sedimentation. This method allowed for the direct quantification of sand and fine gravel transport into, through, and out of the stream bed. These processes define stream bed sedimentation and strongly influence stream ecology by affecting nutrient cycling (Boulton and others, 1998), development of aquatic invertebrate communities (Chiao and Wallace, 2003), and ultimately the survival and reproduction of numerous river fishes (Suttle and others, 2004).

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Atmospheric/Oceanic Influences on Climate in the Southern Appalachians

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Abstract--Despite a wealth of research, scientists still disagree about the existence, magnitude, duration and potential causes of global warming and climate change. For example, only recently have we recognized that, given historical global climate patterns, much of the global warming trend we are experiencing appears to be natural. We analyzed long-term climatologic records from Coweeta Hydrologic Laboratory (1934 to present). There is strong annual and decadal cycling in temperatures and rainfall patterns. These are confounded by a significant amount of natural climatic variability in the southern Appalachians. The natural variability is closely linked to fluctuations in the North Atlantic Oscillation (NAO). For example, the record drought in the southeastern United States, while extreme, was not unusual given historical patterns of alternating wet and dry cycles. These cycles are characteristically precluded by phase shifts in the NAO. The breaking of the drought by Hurricane Isidore and Tropical Storm Kyle (Sept. 2002) was also consistent with past drought cessation in this region. Apparent trends toward cooler and wetter conditions for this region are consistent with observed behavior in the NAO. While the highly variable nature of climate in this region makes it difficult to identify climate trends, nighttime temperatures (minimum daily) have increased over the past fifty years.

INTRODUCTION

Coweeta Hydrologic Laboratory (Coweeta), est. 1934, is one of the oldest operating experimental watersheds of the USDA Forest Service (Figure 1). Coweeta is a forested, 2185 ha watershed near the southern end of the Appalachian Mountains in western North Carolina. The precipitation and stream gauging network of Coweeta provides one of the oldest and most complete watershed scale hydrologic records in the world. The research mission is to evaluate, explain, and predict how water, soil, and forest resources respond to ecosystem management practices, natural disturbances, and the atmospheric environment; and to identify practices that mitigate impacts on these watershed resources. Coweeta, an Experimental Ecological Reserve and Long-Term Ecological Research site (<http://coweeta.ecology.uga.edu>), shares climatic and meteorological data and research with the National Climatic Data Center (Asheville, NC), the National Weather Service (Greenville-Spartanburg, SC) and the National Atmospheric Deposition Program.

BACKGROUND

This manuscript summarizes historical patterns in precipitation, temperature, and streamflow observed at Coweeta in the context of the primary regional climatic driver known as the North Atlantic Oscillation (NAO). Reported data are from the main climatic station, CS01, and the main weir on Ball Creek, WS08 (Figure 1). Data are available from the USDA Forest Service experimental watershed climate and hydrology databases, CLIMDB and HYDRODB (<http://www.fsl.orst.edu/hydrodb/>). Raw NAO data were obtained from the database of Dr. James Hurrell, National Center for Atmospheric Research, Boulder, CO (<http://www.cgd.ucar.edu/cas/jhurrell/indices.html>).

NAO

The North Atlantic Oscillation (NAO) is an atmospheric pressure gradient flux between a persistent equatorial high pressure system and an Icelandic low pressure system that drives climatic variability in the southeastern and coastal Atlantic United States (Hurrell, et al, 2003). The NAO is caused by, and interacts with, sea surface temperatures and wind patterns over the North Atlantic. It is similar to the Pacific Oscillation that drives the El Nino/La Nina cycle. Relative changes in strength between these systems cause large scale changes in atmospheric mass and subsequently exhibit control of general wind and weather patterns over the maritime climatic regions of the Atlantic Ocean. As such, it plays a dominant role in influencing climatic trends and variability from central North America to Europe. The winter NAO index, computed as the difference between the polar low and the subtropical high during the winter season (December through March), strongly influences climate and while varying from year to year, exhibits a tendency to remain in positive or negative "phases" for several years. The Positive NAO phase features a relatively larger pressure gradient with a stronger high and stronger low pressure centers. Winter conditions are generally mild and wet in the Atlantic coastal areas (Figure 2). Conversely, the negative phase has a weaker high and a weak Icelandic low. This allows deeper penetration of polar air masses to the United States and generally produces lower temperatures and more snow. See Hurrell, et. al. (2003) for a comprehensive NAO review.

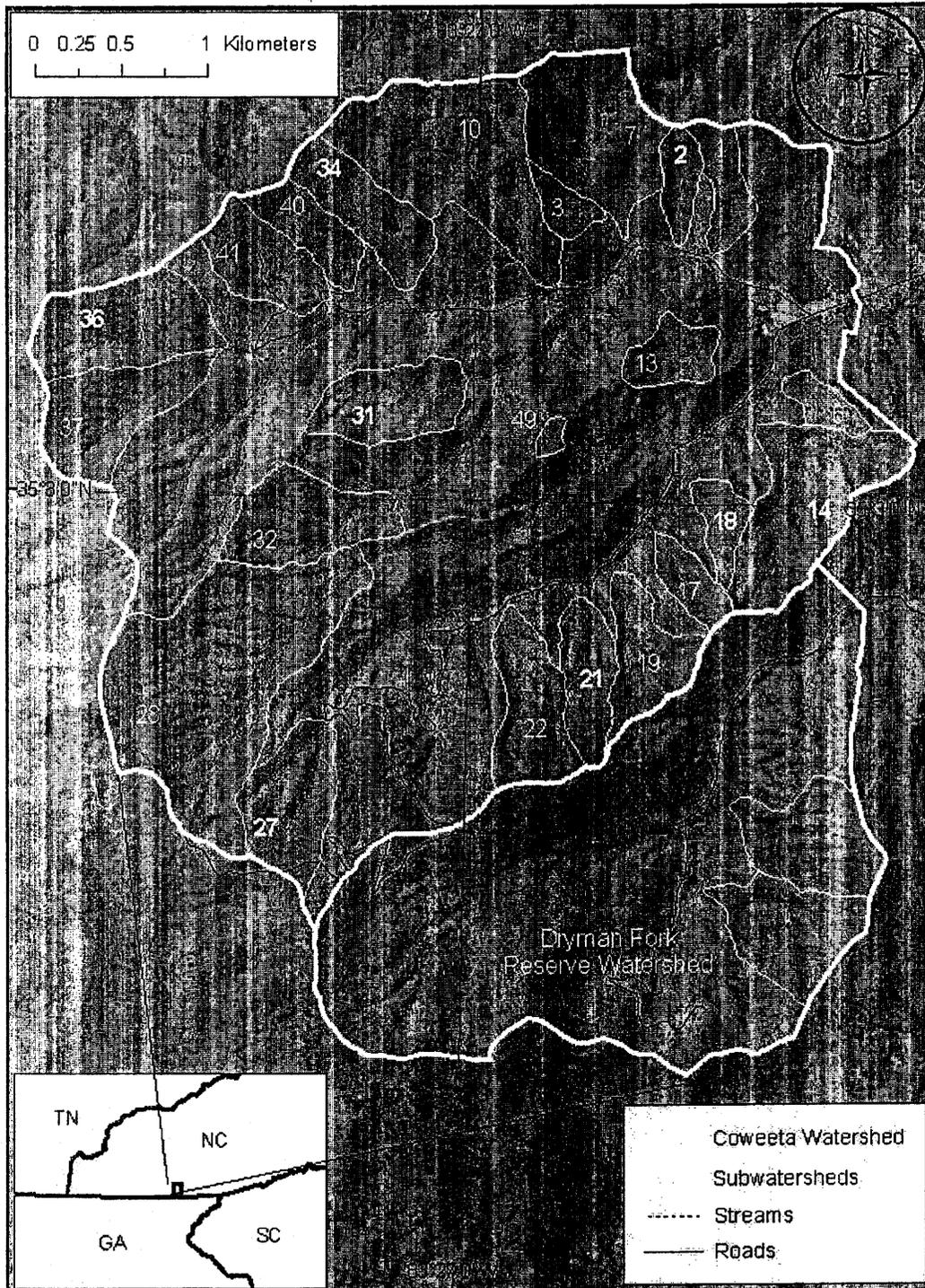


Figure 1: Coweeta Hydrologic Laboratory, Southern Research Station, USDA Forest Service, Otto, NC.

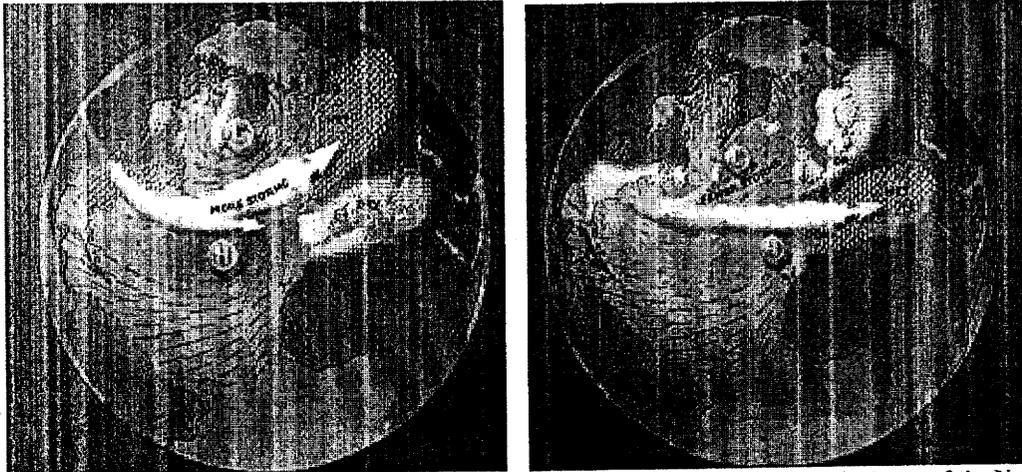


Figure 2: Generalized weather conditions during the positive (left) and negative (right) phases of the North Atlantic Oscillation (images courtesy of NOAA & Lamont-Doherty Earth Observatory, 2001).

Site Description

The Blue Ridge Mountains extend from southern Pennsylvania to northern Georgia, and feature narrow ridges, hilly plateaus, mountains, and high peaks. Bedrock is generally igneous and metamorphic and soils are typically of quartz-rich gneiss, mica-shist, and granitic origin. Elevations at Coweeta range from 675 m near the eastern outlet to 1592 m along the western ridge of the watershed. Slopes range from 30 to over 200 percent. Average rainfall is 230 cm/yr at upper elevations (1600 m) and 180 cm/yr at CS01 (685 m) (Swift, et al, 1988). Mean annual temperature is 12.6 C and ranges from an average of 11.7 C in winter to 21.6 C in the summer. Frequent rain, over 130 storms distributed throughout the year, sustain streamflow (127 cm/yr) and high evapotranspiration rates. The region is categorized as having a maritime, humid, temperate climate (Swift, et al, 1988).

METHODS

Data processing methods follow federal standards and were previously reported (Swift and Cunningham, 1986; Hibbert and Cunningham, 1966). Instrumentation, operation, and maintenance at CS01 (along with all climate stations at Coweeta) were consistent with National Weather Service standard methods. While all meteorological variables were measured at CS01, descriptions here are limited to precipitation and temperature instruments. All readings were taken at 0900 United States Standard Time, eastern time zone. Total precipitation was collected in a standard precipitation gauge and measured to the nearest 1/100th of an inch. Precipitation intensity was recorded using a Belfort Gravimetric recording precipitation gauge. Temperature was measured inside a standard shelter with maximum/minimum thermometers and a hygrothermograph. Streamflow at WS08 was measured by recording the water surface elevation (stage) on 5 minute intervals in the ponding basin of a calibrated, twelve foot Cipolletti weir (Figure 3). Time series and trend analyses of the data were conducted with standardized departure (anomaly) approach (McCabe and Wolock, 2002) by comparing individual standardized values over the period of record.

RESULTS

Temperature

Annual minimum and maximum record temperatures exhibited no trends over the period of record (Figure 4a). Average maximum (daily) temperatures showed no trend while minimum (nighttime) temperatures increased (Figure 4b). Curvature in the data suggested a polynomial trend however, the record was too short to test the statistical significance of this apparent trend. Average daily temperatures, (minimum + maximum)/2, exhibited a periodicity consistent with Winter NAO data (Figure 5); NAO and temperatures were unbiased through the 1950's, below average during the 1960's and early 1970's, and positively biased since.

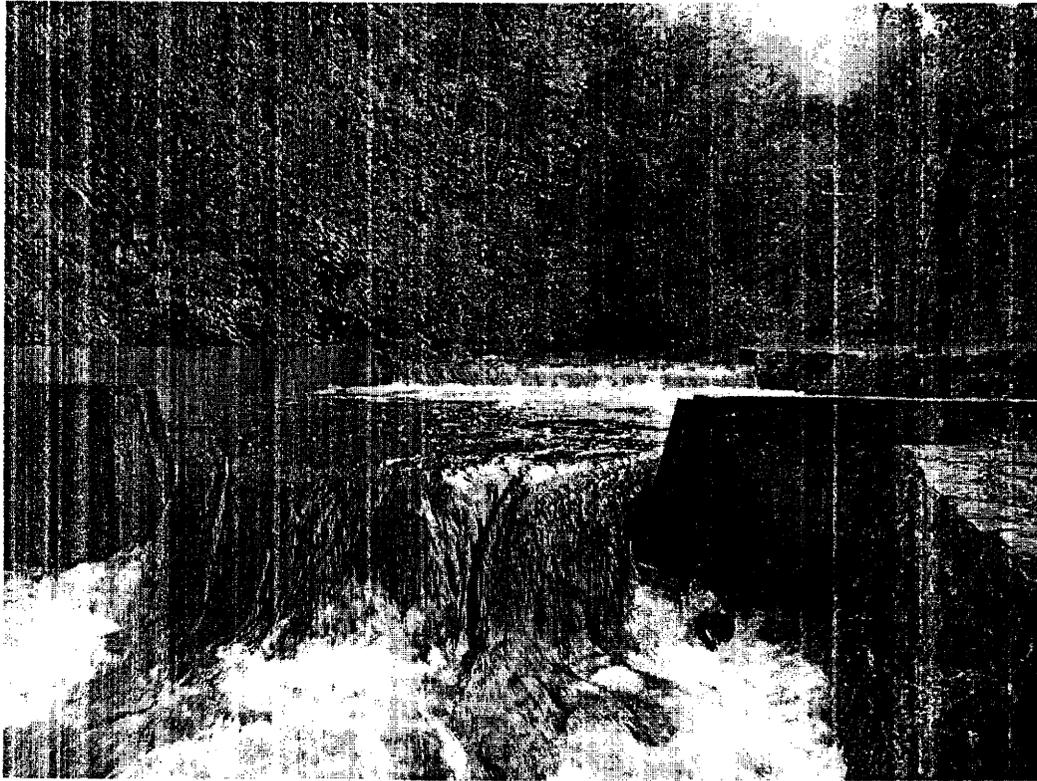


Figure 3: Twelve foot Cipolletti weir at WS08 (760 ha) during Tropical Storm Isidore, September, 2002.

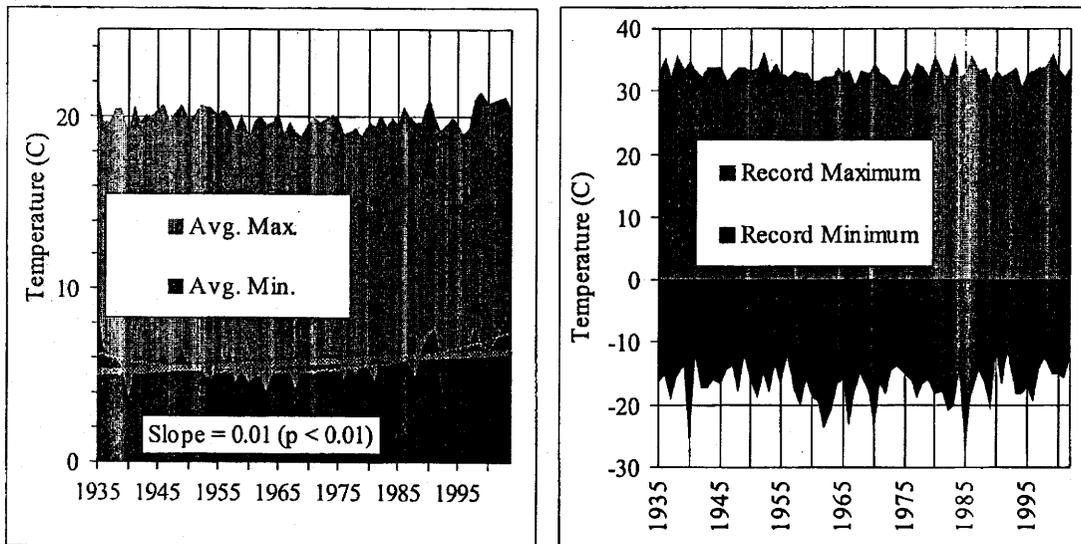


Figure 4a: Record temperatures at Coweeta exhibited no trend.

Figure 4b: Long term average minimum temperatures (nighttime) were increasing. Curvature in data suggested a polynomial or wave function however, the period was too long to identify with existing data.

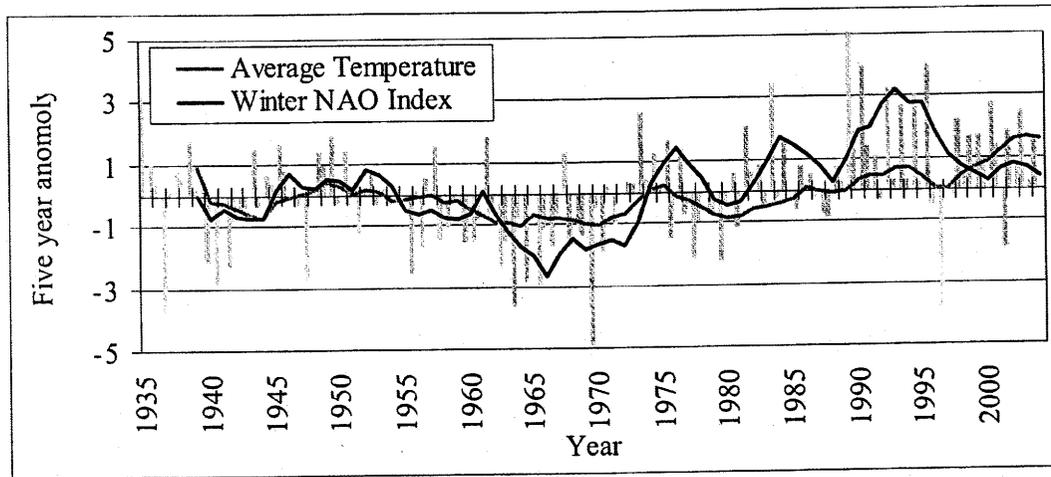


Figure 5: Average annual temperature and winter NAO anomaly (bars) and five year average trend lines.

Precipitation and Stream flow

Annual precipitation (Figure 6a) and stream flow (as water yield, Figure 6b) exhibited no trends over the period of record. Conversely, when precipitation and stream flow were compared with the winter NAO, dry and wet trends were evident (Figure 7). Wet and dry periods, indicated by 5 year moving average, were preceded by positive and negative phases of the NAO.

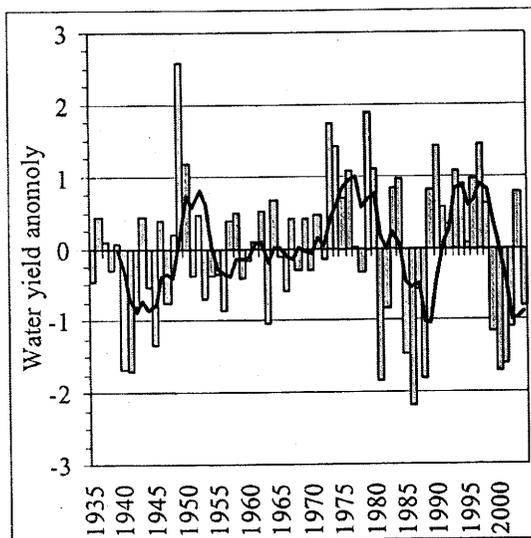


Figure 6a: Annual precipitation anomaly and five year trend at Coweeta.

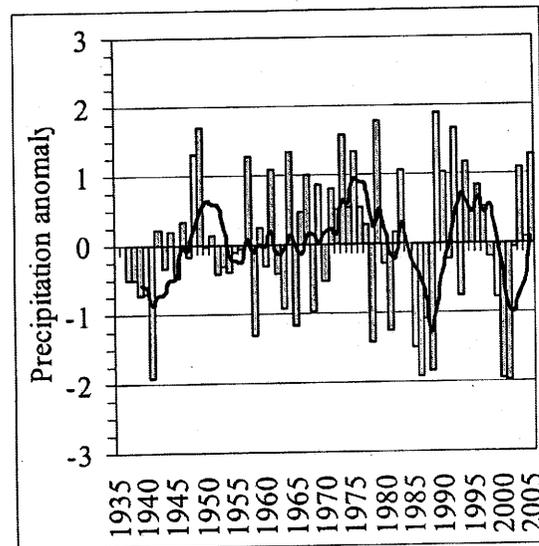


Figure 6b: Annual stream flow (water yield) anomaly and five year trend at Coweeta.

DISCUSSION AND CONCLUSIONS

Analyses of core climatic data from the Coweeta Hydrologic Laboratory indicated long-term positive increase in minimum, nighttime temperatures and periodicity in average annual temperature and total annual precipitation and streamflow. This periodicity was consistent with phase shifts in the winter NAO. The winter NAO plays an important role in controlling the track of air masses across the region hence, it may steer general tendencies in regional climate (Portis, et al., 2001; Hurrell, 2000). Boyles and Raman (2003) reported recent trends in temperature and precipitation; however, these were generalized and did not address linkages with the NAO. While

similar patterns and linkages exist for other maritime climatic regions in closer proximity to the Atlantic Ocean, (Karabörk, et al., 2005; Hayden and Hayden, 2003; Higgins, et al., 2000), Coweeta data indicated the influence of the NAO extend far inland. Previous research found no relation between El Nino/La Nina (ENSO) oscillation in the Pacific and Coweeta data (Greenland and Kittel, 2002; Greenland, 2001). This is likely because the southern Appalachians are too far away from the southern Pacific to be affected by its climatic influences.

Long-term climatic monitoring is very important because it helps explain current weather in terms of what has happened in the past and what may happen in the future. There is a significant amount of natural climatic variability in the southern Appalachians. The natural variability is closely linked to variability in the NAO. For example, the record drought in the southeastern United States (1999 - 2002), while extreme, was not unusual when considered in the context of historical patterns of alternating wet and dry cycles. These cycles were linked to shifts in the NAO. Similarly, the breaking of the drought by Hurricane Isidore and Tropical Storm Kyle (Sept. 2002) was typical for this region as past droughts have been broken by hurricanes or tropical storms. While recent trends toward warmer and wetter conditions for this region were consistent with observed behavior in the NAO, the warming trend was stronger than expected, given historical patterns; the NAO has had a positive bias from the early 1970's through present. These trends in core climatic data were consistent observed NAO behavior however, predicting future behavior is difficult due to interdependency between global climate trends, the NAO and sea surface temperatures (Eden and Jung, 2001).

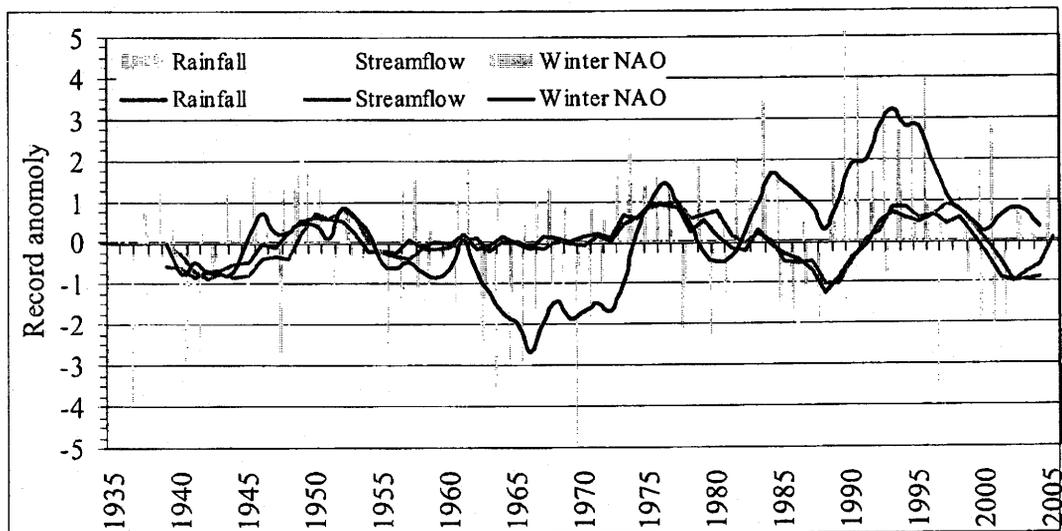


Figure 7: Average annual precipitation, streamflow, and winter NAO index anomaly (bars) and five year moving average trend lines.

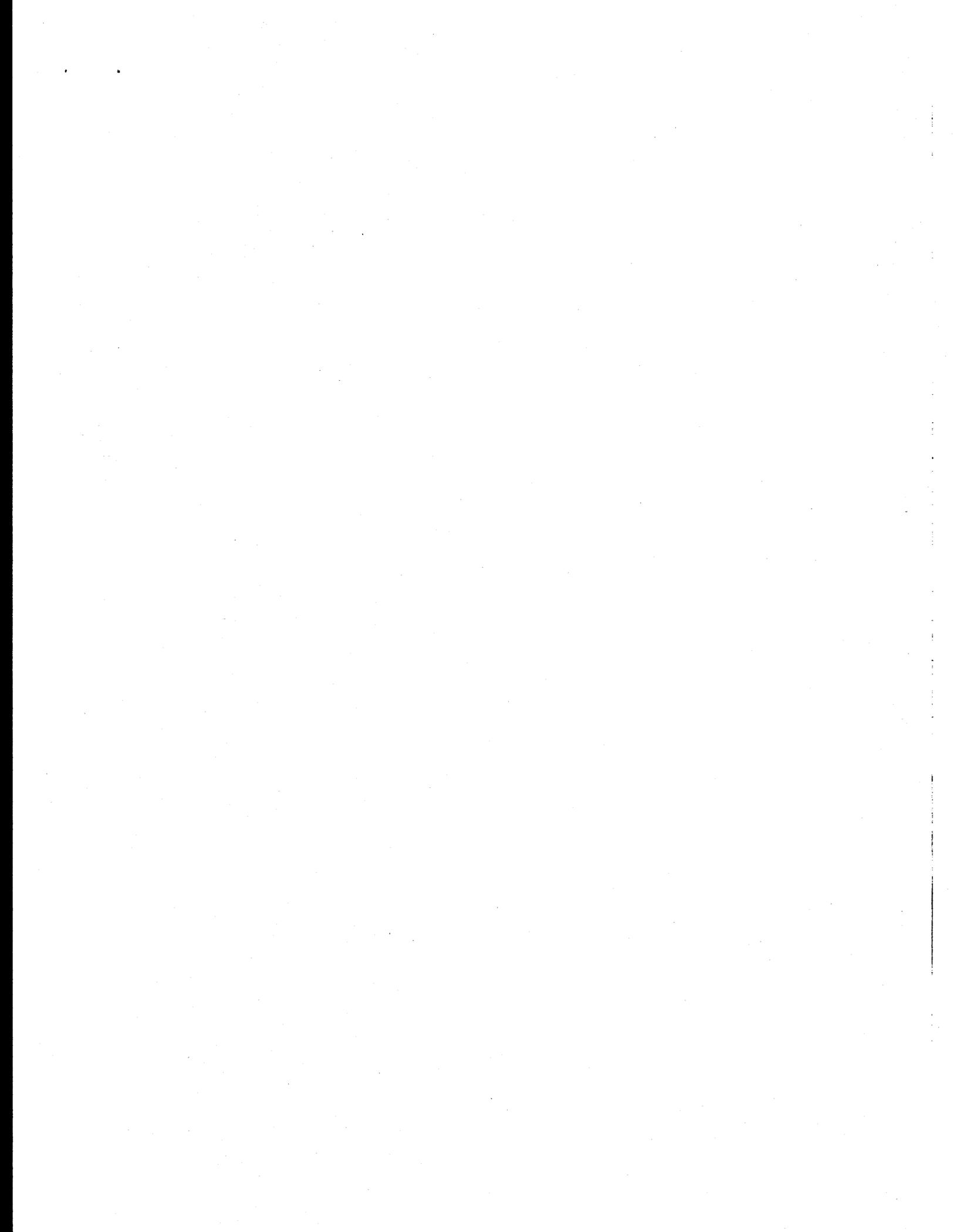
ACKNOWLEDGEMENTS

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INCORPORATION OF VARIABLE-SOURCE-AREA HYDROLOGY IN THE PHOSPHORUS INDEX: A PARADIGM FOR IMPROVING RELEVANCY OF WATERSHED RESEARCH

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Abstract--USDA-ARS at University Park, PA has been conducting research on variable-source-area (VSA) hydrology for almost 40 years. A major emphasis of this research has been on implications of VSA hydrology for water quality, specifically, land management to control nutrient loss from agriculture. Recently, the authors were involved in providing improvements to the Phosphorus Index, a user-oriented tool originally formulated by NRCS to rank vulnerability of fields for potential phosphorus (P) loss. The initial formulation of the P Index contained simple factors reflecting P transport potential, but its developers did not consider the impact of a VSA type of watershed response on P loss. After demonstrating the importance of VSA hydrology, the authors succeeded in having a VSA-related transport factor incorporated in many of the State-level PIs now being used for nutrient management. Drawing on our experiences in development and application of the PI approach to land management, including inclusion of a VSA-reflective transport factor, we discuss the more general areas of integrating research results from the watershed hydrology discipline with those from the disciplines of soil chemistry, nutrient dynamics, and nutrient transport, and then incorporating the integration into planning tools that are both usable and acceptable to planners and farmers. We address needs, limitations, and compromises necessary to accomplish this, with the ultimate purpose of improving the utility of basic watershed research.

INTRODUCTION

In areas of intensive agricultural production, continuing inputs of phosphorus (P) from fertilizer and manure to the landscape often exceed its output in crop and animal produce (Kellogg and others 2000, Lanyon 2000). This condition can result in increased losses of P in runoff which may contribute to eutrophication of freshwater receiving bodies (Carpenter and others 1998, National Research Council 2000). Attempts to reduce P loss from agriculture have followed an evolutionary process as our understanding of the controlling processes and the system being impacted progressed. Subsequent to the problem initially being recognized and documented (Ryden and others 1973, Omernik 1977), research emphasis was directed toward quantifying soil P-runoff water interactions at laboratory and plot scales (Sharpley 1985, Sharpley and others 1981, Taylor and Kunishi 1971). Similar research continues today as we attempt to better understand the controlling processes and more accurately quantify P losses, especially from systems impacted by manure (McDowell and Sharpley 2002, Vadas and others 2005). As our understanding of the natural system increased, and the P problem became more acute, high soil P levels were perceived to be the concern, so management efforts were directed toward minimizing build-up and availability of soil P by using soil test P recommendations to guide fertilizer and manure applications (Sharpley and others 1996). While relatively easily implemented, this approach was found to have only limited success in reducing P loss (Sims and Kleinman 2005, Sharpley and others 1994). Consequently, the emphasis on P management has recently shifted to address the more detailed interactions between P source terms, as manifested by soil P levels and fertilizer and manure additions, and the potential P transport mechanisms of runoff and erosion (Gburek and others 2000, Pote and others 1996, Pionke and others 2000).

As a result of this evolution, we have developed a detailed understanding of the dynamic interactions between soil P and water that control P levels in runoff at point or plot scales. However, extension of this knowledge to multi-field systems or landscape scales remains problematic, since the spatially variable P sources, sinks, and transport processes linked by the watershed-scale flow system have not been investigated and quantified at the same level of detail or at the appropriate time and space scales, i.e., the farm, the landscape, and the watershed. Yet, understanding P-flow interactions at these scales is critical, since any comprehensive P management strategy must address downgradient water quality impacts – this is where the results of the P management efforts applied to fields and farms will be evaluated. To be effective, a P management strategy must integrate effects at the local scale, where specific management practices are implemented (i.e., the field), with the scale of the logical management unit (i.e., the farm), and finally with the larger scale at which results of the strategy are evaluated (i.e., the watershed).

Further, control measures implemented within such an approach to P management will reduce losses most effectively if they are targeted to critical source-areas (CSA's), identifiable areas of the landscape that are most

vulnerable to P loss in runoff (Prato and Wu 1991, Heathwaite and Johnes 1996, Gburek and others 1996, Gburek and Sharpley 1998). CSAs are formed by the coincidence of two factors, referred to as source factors (functions of soil, crop, and management) and transport factors (surface runoff, erosion, and channel processes). Source factors relate to fields or watershed areas that have a high potential to contribute to P export – these are typically well defined and reflect land use patterns as related to soil P status and fertilizer and manure P inputs (Pionke and others 1997, Gburek and Sharpley 1998). The transport factors on the other hand, runoff, erosion, and channel processes, are what transform the potential P source areas into actual P losses from a field or watershed – these are not as easily defined, reflecting natural watershed properties and processes, and being a function of spatial and temporal variabilities in such natural factors as climate, topography, soils, and geology. So to effectively and economically manage P loss at the watershed scale, we must develop techniques to easily and objectively define these CSAs, and also bring their definition into whatever P management strategy is developed – this charge appears to be well within the present-day research bailiwick of watershed hydrologists.

As implied, widespread recognition of the role of watershed processes in quantifying and managing P loss from agriculture has occurred only recently, but a limited number of earlier references referring to the need to consider CSAs (in different terms) for water quality management are found in the more traditional hydrology-related literature. For instance, Dunne (1978), one of the original field-oriented investigators of the variable-source-area (VSA) approach to describing runoff generation, stated, “The transport of fertilizers, herbicides, or animal wastes, for example, can be highly dependent upon where the material is placed in relation to the runoff source areas.” Note Dunne’s specific use of the terms transport and runoff source areas. In a study in Vermont, Kunkle (as referenced in Betson and Ardis 1978) concluded that, “...because of the runoff processes involved, upland contributions of bacteria to streams were small compared to contributions from land surfaces near channels, the channel itself, or direct inputs.” Kunkle provides more indirect, but still obvious, references to transport and source areas.

To summarize, it is critical that hydrologists understand and be able to portray the transport component of CSAs within and from a watershed, because transport processes are what move P (or any pollutant, for that matter) from its source, through the watershed, and finally to its point of impact. Further, for ultimately coupling the hydrology-based portrayal of transport with related research defining P source areas so as to develop effective P management tools or techniques, it is critical that the hydrologist work to develop the capability to represent P transport at the watershed scale simply and objectively. In this context, we discuss the generalities of incorporating VSA hydrology into the P management arena, and the problems and limitations associated with this incorporation from hydrologic, soil chemistry, and agricultural extension disciplinary viewpoints.

BACKGROUND

The authors contributing to this paper, co-located on the campus of the Pennsylvania State University, have each had successful careers within their individual research specialty areas, watershed hydrology with an emphasis on nutrient transport, soil chemistry with an emphasis on P, and agronomy with an emphasis on practical application of research findings via agricultural extension, respectively. More recently, we worked together on a simple, practical, user-oriented tool for P management called the Phosphorus Index (Lemunyon and Gilbert 1993). Within this arena, we collaborated with a number of other scientists to develop a national-scale approach to P Indexing for the Natural Resource Conservation Service (NRCS, Sharpley and others 2003), and also worked in a more self-contained mode to help develop and implement a P Index specific to Pennsylvania (Weld and others 2003). Because of this collaborative involvement, we have been associated with all phases of the P Index process, and relevant to this paper, specifically with incorporation of “watershed hydrology” into a practical P-management tool.

The NRCS envisioned the P Index to be used by field staff and farmers for ranking the vulnerability of fields as sources of P lost to runoff. As first formulated by personnel mostly from soil science and agricultural extension disciplines (Lemunyon and Gilbert 1993), the P Index was intended to be an edge-of-field screening tool. Edge-of-field P losses though, while important, must be evaluated with respect to their potential impact on a receiving water body, because there is where effects of excess P application are manifested and results of P management efforts will be evaluated. For instance, when the original P Index was applied to a research watershed in Pennsylvania where ARS had mapped P source areas in detail and identified the variable source-areas of runoff production, its field rankings did not reflect the watershed areas having combinations of high soil P levels and high probabilities of runoff thought to impact the stream draining the watershed (Gburek and others 1996, McDowell and others 2001). These studies, and successive research efforts applied to the same watershed (Gburek and Sharpley 1998, Gburek and others 2000, Gburek and others 2002, Sharpley and others 2001b), coupled with the extensive VSA-related

literature, provided the foundation for development of modifications to the original P Indexing approach reflecting P transport specifically from hydrologically active areas. The modified P Index concept, recognizing the interactions between transport and source factors controlling P loss at the field-scale using a landscape-scale representation of VSA runoff generation, has been incorporated in many State-specific P Indices (Sharpley and others 2003).

Following are brief descriptions of watershed hydrology, soil chemistry, and agricultural extension in context of P management issues, and integration of these disciplines related to incorporating watershed hydrology concepts into the P Index. Based on these experiences, recommendations for watershed research in the more generic sense are developed to help the research program continue to remain vital and relevant as the world and its problems change.

WATERSHED HYDROLOGY

The science of watershed hydrology, specifically that component related to generation of the storm hydrograph, was founded on the basis of quantifying a watershed-scale response to precipitation. Water quantity, not quality, was originally of import, so the earliest hydrologists were generally unconcerned with details of within-watershed spatial variability when developing tools to quantify this response – they were simply concerned with representing a response at the watershed outlet, often in a risk or probability format. This view of a watershed's rainfall-runoff performance ultimately developed as its foundation an infiltration-excess (IE) approach to surface runoff generation, i.e., a Hortonian concept of runoff production (Horton 1933). Here, surface runoff is thought to be generated more or less uniformly from all locations within a watershed, with the runoff generated assumed to be the result of rainfall intensity exceeding the time-varying infiltration capacity of the soil – the soil becomes saturated from the top down. In current views of IE runoff generation, surface runoff can be expressed on a whole-storm basis (e.g., the Curve Number approach, NRCS 1985), or on a within-storm, time-variable basis (e.g., Philip 1957).

As the IE approach to quantifying surface runoff continued to evolve, hydrologists incorporated a spatially variable nature to the runoff production mechanisms – this was generally expressed as a function of soils distribution and land use, and is exemplified by how the Curve Number (CN) approach to predicting runoff is now typically applied to a watershed, area-weighting CNs from all combinations of soil and land use. However, the underlying CN assumption remains – a given soil-land use combination produces the same runoff depth from a specific precipitation event no matter where within the watershed or landscape it is positioned.

In the mid 1960's, an alternative to the IE-based stormflow generation model began to appear in the hydrology literature, what has since been termed variable-source-area (VSA) hydrology. One significant early presentation of the VSA hydrologic response was derived from research conducted at the Coweeta Watershed facility (Hewlett and Hibbert 1965). The observations reported were made under forested conditions, with subsurface stormflow thought to provide most, if not all, of the increased flow observed during a storm hydrograph. Most of the subsequent research reported was also conducted in the humid-climate, forested environment (Ward 1984, Bernier 1985, Hibbert and Troendle 1988, Bonell 1993), and consensus developed that stormflow (not strictly surface runoff) forming the storm hydrograph is generated from localized, typically near-stream, areas of the landscape. VSA stormflow results partly from rainfall on soils having little or no available storage, thus precluding infiltration, rather than on soils with a limited infiltration capacity. Controls on VSA stormflow generation are from below the soil surface (typically high soil water or ground water levels) rather than above (rain intensity > infiltration capacity at the soil surface), and the mechanism controlling surface runoff generation has come to be referred to as saturation excess, rather than infiltration excess.

The basic premise of VSA hydrology is existence of a dynamic runoff-contributing subwatershed within the topographically defined watershed that expands and contracts during a storm as a function of precipitation, topography, soil type, geology, ground water levels, and watershed moisture status. The runoff-contributing subwatershed produces an ill-defined combination of saturation excess overland flow and subsurface stormflow to the stream. This concept has caused watershed hydrologists, especially those in humid-climate regions where VSA hydrology is thought to be the dominant runoff-generating mechanism, to consider not only soils and their distribution, but also the influential and interactive roles of topography, geology, and saturated and unsaturated flow regimes, when attempting to explain a watershed's response to rain fall (Dunne and others 1975, Burt and Butcher 1985, Srinivasan and others 2002). Obviously, this suggests that the hydrologic discipline focus more on the landscape scale, and less on plot-type studies of infiltration and runoff or on integrated watershed-scale responses.

Coincident with emergence of VSA hydrology was increasing recognition of water quality problems within the U.S.,

many of these associated with agriculture. Watershed hydrologists and other environmental researchers began addressing these problems by developing "integrated" watershed hydrology-water quality models. A number of field- and watershed-scale hydrology-water quality models have been developed over the years, but two are notable for their current widespread application. The Hydrologic Simulation Program-Fortran (HSPF, Donigan and others 1984) and the Soil and Water Assessment Tool (SWAT, Arnold and others 1998) are being used routinely to assess water quality problems at the watershed scale. However, both can be described more as water quality add-ons to previously existing watershed hydrology models rather than a truly integrated model where all components, both hydrologic and chemical, were developed and included strictly with the underlying water quality objectives in mind. Quantifying nutrient transport within and from a watershed though, requires knowledge of flow pathways and/or differential sources of runoff within the watershed, since moving water provides the mechanisms for nutrient transport. If we consider these two watershed models in context of a VSA/CSA view of watershed water quality performance, we see there is a disconnect, because the models rely on an infiltration excess view of the world, using variations of CN technology to simulate runoff and related nutrient loss from over a watershed.

This disconnect may be partly due to the hydrologists conducting VSA research. While we have realized for some time that many watersheds exhibit a VSA type of response, not only haven't we worked to include VSA hydrology in hydrology-water quality models, we haven't yet developed methodologies to express a VSA type of response similar to that of the Curve Number – that is, a simple, acceptable, and regionally applicable method to identify VSAs based on readily available datasets. There has been substantial research into the VSA watershed response, but there are few references reflecting landscape- or watershed-scale identification and quantification of VSAs, and even more telling, VSA hydrologic response is currently not part of any recognized, routinely applied watershed-water quality model. This may not be a problem when simply attempting to simulate nutrient output based on the variety of land uses within a watershed when the model can undergo extensive calibration and verification (e.g., SWAT and HSPF), but it becomes a problem when we attempt to look within the watershed and identify CSAs, for instance, that are the cause of P loss over the watershed for management and remediation purposes. If we extend this limitation to the more general field of watershed hydrology, we may find that our field is more insular than it should be, not having reacted to needs in problem areas related to, but outside the realm of, watershed hydrology.

The science of watershed hydrology has explored new boundaries in promoting the importance of the VSA concept. While the concept may not be quantified to the extent needed, the related concept of CSAs has been accepted by, and become a challenge to, other environmental scientists and land managers when considering impacts of land management on the forms, amounts, and environmental availability of P applied over the landscape.

SOIL (PHOSPHORUS) CHEMISTRY

Since the 1960's, when the environmental consequences of excess P were becoming apparent, there has been a shift in the philosophy of soil P chemistry from a concern with P build-up, to one of P maintenance, and finally to a balanced approach in terms of P being added to the farming system of interest verses that lost in produce. To a large extent, this expanded our research emphases from a focus on basic soil mineralogical, physical, and biological interactions, to more of a concern with soil P reactions, forms, and controls, as well as the fate of P added in fertilizer. Eventually, fertilizer-type studies were expanded to encompass impacts of application of biosolids and manures, which have recently become more environmentally important. These studies mainly involved agronomic response trials using field plots and adsorption of P by soil using batch experiments in the laboratory.

The change in agriculture in the mid 1980's from production-only concerns to those encompassing environmental consequences, led to a shift in research direction, with increased emphases on applied soil chemistry. Studies of P desorption, or release from soil, directed research away from P adsorption, showing that soils can be a source of P under some conditions (Pote and others 1996, Sharpley 1995). Conventional wisdom that soil is an infinite sink for P was questioned by this research, especially for soils with high P concentrations where P desorption can be both rapid and large enough to exceed eutrophication thresholds (Vollenweider 1976). At the same time, agriculture was undergoing another transition – by the mid 1990's, crop and livestock operations had become spatially separated and increasingly intensive, resulting in an even more rapid build up of P in soil (Lander and others 1998, Lanyon 2000). This progression of agriculture, coupled with the knowledge that soil P could actually be released back to water and increase the incidence of algal blooms in fresh water systems, shifted the soil P chemistry paradigm from production to environmental drivers (Boesch and others 2001, Carpenter and others 1998, Foy 2005).

As land management has intensified and increased amounts of manure are being applied based on crop nitrogen

needs, it has become all too apparent that potential for enrichment of runoff by P is increasing (Kellogg and others 2000). This was initially reflected by increases in soil test P levels encountered by testing laboratories (Beegle 2005, Sharpley and others 2003) by application of environmental P thresholds developed from the extraction of soil P with water and iron-oxide strips (environmental tests), and by limited experimentation using soil packed in trays and subjected to simulated rainfall (Kleinman and others 2004, Sharpley 1995). However, there was very little field- or watershed-scale evaluation of P loss from agriculture until the early 1980's. Field studies then showed that land management was a primary factor determining P loss, and in many situations, broadcast fertilizer or manure applications could mask or even override soil P effects on runoff P (Andraski and Bundy 2003, Sharpley and others 2001a). It was a logical progression for soil P chemists to develop experiments showing that land and fertilizer or manure management affected runoff P (Andraski and others 2003, Ebeling and others 2002, McDowell and Sharpley 2002). In the specific case of manures, they have showed that rate, timing, method of application, and even source of manure (i.e., animal type) all influenced runoff P levels (Edwards and Daniel 1993, Withers and others 2001).

The shift from agricultural production to inclusion of environmental soil P chemistry has led to changes in how we perceive the "world" of soil P. The first basic change, and probably most challenging, was an increase in scale of study from laboratory test tubes and small plots to landscape processes, and consequently having to deal with variability of soil P within agricultural watersheds (McDowell and others 2001, Sharpley and others 2001a). Associated with this change was the realization that P is a valuable resource at the point of application, and remains so until it is moved away via surface or subsurface transport. Like the definition, "A weed is a plant in the wrong place," P changes from a resource to a pollutant when it enters surface waters in quantities sufficient to promote eutrophication. This realization then led the need to consider inclusion of hydrologic processes when considering research in soil P chemistry. When integrated, these two disciplines help define whether P remains a resource or transforms to a potential pollutant (Gburek and Sharpley 1998, Gburek and others 2005, Simard and others 2000). The integration also provided the primary impetus for development of the most recent versions of the P Index being applied (Sharpley and others 2003). As importantly, in terms of watershed-oriented P research, it has fostered a closer collaboration with agricultural extension in order to transfer this research to end-users.

AGRICULTURAL EXTENSION

Dissemination of information and advice to farmers has had a long history, beginning even before the emergence of modern agricultural extension. The first known example of an extension-type activity was in Mesopotamia around 1800 B.C. Archaeologists have unearthed clay tablets from that time on which was inscribed advice for watering crops and getting rid of rats (Jones and Garforth 1997). In the U.S., the formalization of extension activities culminated in 1914 with the passage of the Smith-Lever Act establishing the Cooperative Extension Service, a tripartite cooperation between federal, state, and county governments, with the local state colleges as the extension agencies. This was done "to aid in diffusing among the people of the United States, useful and practical information on subjects relating to agriculture and home economics, and to encourage the application of the same." (Jones and Garforth 1997). The formal link of extension with Land Grant Universities has established a strong and continuing connection between research and application related to agriculture. Extension provides not only a mechanism to disseminate research results, but also provides a channel to communicate research needs back to the scientists.

This two-way linkage is one of the key strengths of the system. The emphasis of extension was founded on increased agricultural production and improvement of rural life. Over the years, the mission of extension has diversified significantly, most notably in the emerging problem areas of socioeconomic and the environment. While the overall extension program has remained highly diverse, there has been a shift in recent times to extension providing increased "public good" education, while the private sector has assumed more of the agricultural production-related activities. A classic example of this shift is the work that extension now does in environmental education. Fortunately, even though there has been some change in subject matter emphasis, the research-based education paradigm has remained at the center of extension programs. Extension continues to be a catalyst for connecting basic and applied research as the result of its mission to provide the end-users with helpful and applicable information – applied research efforts directed toward development of practical applications from results of basic research have often been initiated from needs identified through extension programs. Further, as a result of this process, many basic research disciplines have learned the importance of fostering linkages to applied and adaptive research, and ultimately to dissemination of the results of their work for end-users.

Extension has generally focused on problems at the farm-scale level, while agricultural research has been largely reductionist in nature, looking at small, relatively easily quantifiable pieces of the problem, usually at plot and field

scales. Shifts in this approach have occurred though, with extension expanding its role to encompass larger systems, such as whole industries, communities, and watersheds. Likewise, and perhaps partly as a result of this shift in extension's needs, there has been an increased emphasis on systems-level research, and a corresponding increase in interdisciplinary research (e.g., soil P chemistry and watershed hydrology), all addressing problems at a variety of scales. Both of these changes are seen as critical since the issues faced by agriculture have become more global in nature. For example, environmental problems historically related to nutrient management have been addressed through research and education related to modification of farm- and field-level management practices. However, contemporary nutrient management problems have been shown to be not so much related to farm management, but rather as to how agriculture is responding to global economic forces and to the accumulated impacts of the overall structure of farming systems across broad areas, such as the Chesapeake Bay Watershed. This implies that that extension must broaden its scope even further, to include not only results of traditional agricultural research, but also results from research on economics, public policy, and watershed hydrology-water quality interactions.

INTEGRATION – THE PHOSPHORUS INDEX

Development and implementation of the P Index approach has been a classic example of how the connection between research and extension provides the unique synergy needed to address critical problems facing modern agriculture. The initial P Index framework was developed in 1991 during an NRCS-sponsored meeting of seven researchers experienced in soil P chemistry, the release of P from soil to water, and its loss in surface runoff. The format of the initial P Index was a direct reflection of the expertise of those involved in its development, with hydrologic principles and processes included at rudimentary levels, incorporating only NRCS soil runoff classes and the Universal Soil Loss Equation to represent P transport.

It later became evident from field studies though, that movement of P from point of application into a stream is an extremely complex function of surface soil physiochemical properties, current and historical nutrient management, surface slope, and vegetative cover and type, as well as rainfall intensity, duration, and frequency, in other words, landscape- or watershed-scale hydrologic processes. Thus, a tool reflecting the risk of P loss from agricultural land, such as a P Index, should represent all these processes in a way that reflects their relative importance in the field. This stimulated the shift in the soil P chemistry/watershed hydrology paradigm toward representing spatial and temporal variabilities in surface and subsurface hydrologic processes as related to their controls on P transport, and the P Index went through another round of iterations, this time with watershed hydrology expertise and extension added to the mix. Soil scientists have a history of adapting their research to field-scale applications, but such adaptation has been less commonly related to landscape-scale issues like those studied by watershed hydrologists. The improved P Index was a catalyst for integrating these somewhat disparate disciplines to address a critical contemporary management issue at common scientific scales and similar levels of detail. Extension provided the input necessary to keep the form of the improved P Index focused on the desired result, an estimate of the risk of P loss from a landscape, and to keep it simple enough to be used effectively by farmers and farm advisors. This last factor is critical if the P Index is to be effective in helping reduce the risk of P loss to the environment.

The integration of soil P chemistry and watershed hydrology principles to better represent P loss from agricultural land uses, coupled with extension's insight in transferring this technology to users, has formed the P Indexing approach we are now embracing. This effort may eventually come to be recognized as one of the more significant instances of watershed-hydrology contributing significantly to a multidisciplinary, user-oriented tool, enhancing the relevancy of watershed research in areas of Comprehensive Nutrient Management Planning, Total Maximum Daily Load, and Nutrient Trading strategies, all in various stages of national adoption.

IMPLICATIONS FOR WATERSHED RESEARCH

Based on the previous sections, we can extract some suggestions for improving the utility of watershed research programs, such as those being conducted by the Agricultural Research Service, the Forest Service, Land Grant Universities, and the U.S. Geological Survey. Perhaps most relevant to defining the potential role of the watershed research community is a conclusion directed toward P Index development by Sharpley and others (2003): At some point, watershed-scale validation will be required of the P Index, leading to a number of questions that must be addressed. For instance, are the areas identified by an Index to be at greatest risk for P loss actually the sources of most of the P exported? In the same vein, will remediation of the high-risk areas identified by the Index decrease P export in stream flow from a watershed? Conversely, can low vulnerability areas receive more liberal P management without increasing P export from the watershed? The questions included, specifically related to the P Index but also easily generalized, are well within the research

realm of watershed hydrologists – incorporating spatial variability in watershed response, developing model testing and application paradigms and relating them to field data, expressing naturally occurring and managed events in terms of risk and/or probability, and evaluating impact of land management alternatives. To make watershed research relevant though, not only as related to something as simple and straightforward as a P Index, but also to the wider variety of real-world environmental problems along with their economic, social, and political ramifications, hydrologists must be willing and able to redirect, or even modify, their way of defining research objectives, conducting research, and expressing research results.

To help streamline its application, the next evolution of the P Indexing approach will likely be more toward an interactive tool closely integrated with available parameter databases. These databases may include, for example, digitized soil survey information, digital elevation models, and County-level GIS-based maps of field boundaries and land use. This suggests that watershed scientists should be thinking about how these same types of datasets can be used to enhance and/or extrapolate their research results when extended to real-world problems. While proceeding in this direction though, we must also be wary of not developing a cadre of “computer” hydrologists. We must remain aware of the vagaries and variabilities of the field, ensuring that the models and techniques we develop incorporate those that are relevant to the problems of concern.

Watershed hydrologists, as well as research soil scientists, typically feel the need to development more rigorous, detailed, mechanistic models. While these may provide a better framework for process-based understanding and management of potential P losses, for instance, such models are typically not practical for routine field application. Consequently, a challenge to watershed scientists who want their work to be applied is to extract the critical parameters from the mechanistic models and incorporate them into user-oriented tools, such as a P Index. A P Index does not rigorously simulate the mechanisms of P loss, but if developed properly, its algorithms capture the basics of watershed performance controlling this loss that are generally applicable to wide geographic areas. Proper development of such a tool, a simplification of the rigor of all contributing disciplines, requires a considerable amount of give and take between disciplines as the appropriate balance is sought between scales, assumptions, and interdependent processes – and also between research rigor and extension-level application. If done successfully, the result will be a tool like the P Index that is first off, scientifically defensible, and as importantly, can be effectively and successfully utilized in the field.

Most of the soil P and management factors influencing P loss included within the P Index have been quantified by cause-and-effect studies – not so with the landscape-scale transport factors however, because of the temporal and spatial variability of hydrologic processes occurring over a watershed resulting from the complex interactions among climate, topography, soils, and geology. Consequently, results of the integration of soil P chemistry and applied watershed hydrology as expressed by the P Index are difficult to evaluate. As the spatial scale of concern increases, uncertainties and unknowns increase too. In the case of the P Index, it becomes increasingly difficult to quantify controls on P transport as we move from lab to plot to landscape. Here, the watershed hydrologist can have significant impact, helping incorporate the concepts of uncertainty, probability, and risk – areas in which watershed hydrologists have traditionally worked – into a management tool. Concurrently, it becomes incumbent on the watershed hydrologist to support this probability- or risk-based approach, because in the P Index, for instance, this approach is a consequence of the transport factors being less-well quantified than are the source factors. Many watershed hydrologists come from an engineering background; they can offer engineering judgment to provide the necessary support, but they must also be willing to stand behind the methodologies developed and “sell” them to the other disciplines involved. The methodologies must be logical, simple, and in line with the other components of the management tool, and they must be understandable by, and make sense to, the other researchers involved, as well as to the extension personnel who will be extending the methodology to the users and to the users themselves.

Truly integrative watershed hydrology-water quality research in support of tools such as the P Index is being conducted only at a limited number of locations. To make their research more relevant, watershed scientists must work more closely with related disciplines, not just on paper but in fact, designing field experiments, instrumentation, and data collection and analyses protocols that properly address multidisciplinary studies with integrative objectives. Additionally, the scientists must involve agricultural extension, or other appropriate technology transfer personnel, in their research to ensure that the controlling processes being investigated, the levels of accuracy being targeted, and the way the results are expressed are appropriate to the problem being addressed.

Lastly, and perhaps most importantly, a critical part of any research program directed toward development of

practical applications involves educating the users – farmers, farm advisors, action agencies, policy makers, and regulators. They must be given a basic understanding of the science behind any such application so that they are comfortable with the approach and willing to accept it as a component of a management program. Extension takes the lead in this educational process, but for it to be fully effective, the contributing researchers must also be involved, providing background material for educational programs and directly participating in these programs. The legitimacy of a program is enhanced when clientele can listen to and directly question the scientists. Again, the P Index development and implementation is a good example of this. It is a practical, user- and management-oriented tool developed by an integrated research approach coupled with an extension-led and scientist-participating education oversight that is being used effectively in a management and environmental policy development effort.

SUMMARY

It is clear from our combined experiences with the P Index that integrating basic and applied research in watershed hydrology and soil science is critical to addressing the many challenges in production and environmental protection faced by agriculture today. Having extension involved in the process helps ensure a balance between scientific rigor and end-use practicality. A “perfect” watershed model might be satisfying to the watershed hydrologist, but if it is too complicated to be used as the hydrologic basis for a P management tool, for instance, it will have minimal impact on real-world problems. Hydrologists, as well as other researchers, must understand that when developing applied applications based on their results, compromise will be necessary. In applying research results, we must determine what will give us the most return for the effort expended, while still keeping our research meaningful. This is analogous to a diminishing-return crop-response curve. We know what it takes to achieve maximum yield, but there is an economic optimum that is lower than the maximum where the last input just returns an equal value of crop response. We must apply this same approach to incorporating research results from individual contributing disciplines when developing practical, user-friendly, and effective management tools.

The bottom line is that now, more than ever, while we continue to develop fundamental knowledge in watershed hydrology, we must also be looking at how to integrate this knowledge with that from other disciplines so as to develop applied, accepted, and effective environmentally oriented management tools. Watershed scientists must work closely with scientists representing all disciplines contributing to a problem area to insure that their discipline-specific assumptions, scales, and levels of detail are integrated properly to provide the desired result. As importantly, the scientists must all work closely with extension educators to develop and implement tools that have the appropriate balance between scientific rigor and practical application.

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LONG-TERM HYDROLOGIC DATABASE: GOODWATER CREEK, MISSOURI

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Abstract-- One of the more valuable ARS assets is the long-term record from their permanent watershed network. However, these data have not always been adequately documented, nor made easily available to the public. This report describes ongoing research at the 72-km² Goodwater Creek (GWC) watershed in north-central Missouri with objectives to 1) fully document; 2) provide quality assurance, and 3) make available in web-compatible form historical GWC data. This watershed, instrumented for hydrology in 1971 and water quality in 1993, is dominated by surface runoff. The original installation included three stream weirs, a rain gauge network, piezometers, and climate instruments. The later period included three field-sized watersheds and an automated weather station, plus autosamplers at all weirs. The GWC database will be made available through the ARS STEWARDS (Sustaining the Earth's Watersheds, Agricultural Research Data System) when it is released.

INTRODUCTION

USDA research, in cooperation with University of Missouri Agricultural Engineering Department, dates at least to 1929, with erosion research tied to the Bureau of Chemistry and Soils and then to the Soil Erosion Service. By 1935, this cooperation was identified with the recently established Soil Conservation Service, and in 1939, plans were made to establish forty 88-m² runoff plots for research into soil and water management at the Midwest Claypan Experiment Farm near Kingdom City MO. These plots were continuously monitored from 1941-1998, during which time the research arm of the Soil Conservation Service was transferred into the new Agricultural Research Service in 1953. Larger-scale watershed research by USDA-ARS at the Columbia location dates to the formation of the North Central Watershed Hydrology Research Unit in the fall of 1961, as the fourth watershed unit authorized by Senate Bill 59 (after Boise ID, Chickasha OK, and Tucson AZ). Related research in the region was conducted by scientists in this unit, including field-sized watersheds at Treynor IA and Kingdom City MO, the erosion research mentioned above, and sedimentation and hydrologic balance research on several regional reservoirs.

In 1969, plans were made to install a series of weirs on Goodwater Creek, north of Centralia MO. These weirs drained areas from 13 to 72 km², and represented nearly half of the drained area of Young's Creek, which had a long-term USGS gauging station draining 172 km². A dense precipitation gauging network was installed concurrent with the weir installation. Collateral data included groundwater elevations, intermittent sediment and nutrient transport, temperature and humidity, and pan evaporation. This configuration of instruments remained in place until approximately 1990, when the scope of monitoring was significantly increased.

At that time, additional funds were provided to add water quality measurements to the instrumentation. Automated samplers at the three stream weirs obtained samples during storm events, and periodic grab samples were taken under baseflow conditions. These samples were analyzed for sediment, nutrients, pesticides, and common pesticide metabolites. Three field-scale watersheds ranging from 7 to 35 ha were instrumented with weirs and automated samplers during 1992. These fields were farmed with contrasting practices, including mulch tillage and no tillage. Groundwater well nests were installed in these fields and elsewhere in the watershed to sample for nutrients and pesticides. A set of 30 plots, 0.37 ha each, was installed in 1991 as 10 plots in each of three replicates, on which several common farming practices were established. Fourteen of these plots were instrumented with flumes and walls and autosamplers to channel and sample runoff during the period 1996-2002. A modern weather station was installed in 1991. Strip chart recorders were augmented with electronic measurements for stream stage heights and rain gauges.

During this period of extensive watershed research, documentation of the instruments and data was done when research using them was published. For instance, a subset of flow data from the smallest of the three nested watersheds (W-11 in Table 1) was extracted by Hjelmfelt and Kramer (1988) and used repeatedly for unit hydrograph analysis (Hjelmfelt and Wang, 1994) and also for evaluating a DEM-based overland flow routing model (Wang and Hjelmfelt, 1998). Ward and others (1994) presented limited field groundwater and stream water quality data in an overview of the multi-state project. Alberts and others (1995a) described herbicide transport in surface

runoff from the field sites (F1-F3 in Table 1), and in Alberts and others (1995b) they included data from the weir locations as well. Lerch and others (1995) reported hydroxyatrazine loads and concentrations of atrazine and several of its metabolites at W-1 from 1992 to 1994. Kitchen and others (1997) described preliminary effects of farming systems on groundwater nitrate using sampling wells at the three field sites. Blanchard and Donald (1997) reported herbicide contamination of ground water in the fields and the watershed. Hjelmfelt and Wang (1999) used observations of runoff, sediment, and atrazine loss for 7 events on F1 to predict possible environmental effects of placing a grassed waterway in the drainage channel of the field. Wang and others (1999) calibrated the HSPF sediment model on 2 years of F1 data and tested it on data from W-1. Fraisse and others (2001) reported a simulation study for corn at 7 sites in F1 during 1997. Drummond and others (2003) described statistical and neural methods to predict site-specific yield on F1 and two additional fields just outside the watershed boundary. Alberts and others (2003) compared nutrient and herbicide concentrations in Goodwater Creek to those in Walnut Creek, IA. Wang and others (2003) used 3 years of weather data and field observations from F1 to evaluate the capability of the CROPGRO-soybean model to simulate within-field variability. Ghidry and others (2005) presented event-total runoff and pesticide concentrations and losses for a total of 36 site-years on 14 of the 0.37-ha plots. Lerch and others (2005) described the development of a conservation-oriented precision agriculture farming system on F1, and Kitchen and others (2005) explained the crop production assessments and the development of the system. These last two citations also present the history of F1, which includes several other research articles that describe soil- and crop-based data collected in F1.

However, little documentation of the watershed as a whole has been published, and only limited site descriptions have been collected into internal documents. One abstract described background information prior to the activity in the early 1990's (Hjelmfelt and others, 1990), and Ward and others (1993) listed the suite of water quality data collected during 1991-1992. Consequently, information is contained in locations specific to research topics and in forms prevalent at the time of the research. While quality assurance procedures were usually performed by the staff collecting the data, anomalies were often corrected without sufficient documentation to determine the problem or the solution. Consequently, in 2003, the Cropping Systems and Water Quality Research Unit initiated a comprehensive project to consolidate and document data obtained in the Goodwater Creek Watershed. Nearly coincident with this launch was that of the USDA-ARS database system STEWARDS (Sustaining The Earth's Watersheds Agricultural Research Database System; Steiner and others, 2005). The purposes of both the local and national projects are the same – providing publicly-funded research data to the public.

The objectives of the current paper are to fully document the extent of Goodwater Creek hydrology and water quality data, to describe the ongoing quality assurance project, and to establish a basis for submitting the Goodwater Creek data to the STEWARDS database system.

PHYSICAL DESCRIPTIONS

The Goodwater Creek watershed is located NE of Columbia MO, in a predominantly agricultural area (Figure 1). The area drained at the outlet (W-1) is 72 km², with two smaller watersheds (W-9, W-11) nested along the main channel. Topography is characterized by broad gently sloping divides, with roughly 37 m elevation from divide to outlet, which is at 235m MSL. From the outlet, Goodwater Creek drains into Young's Creek (172 km²), which then drains into Long Branch Creek of the Salt River (466 km²), which drains directly into Mark Twain Lake, which drains an area of approximately 6,000 km². The lake drains into the lower Salt River, which flows into the Mississippi River between Hannibal and St. Louis MO.

Soils in the watershed belong to Major Land Resource Area (MLRA) 113. Soil families include Vertic Epiaqualfs (Mexico Series), Vertic Albaqualfs (Adco series), and Vertic Epiaqualfs (Leonard series). (<http://soils.usda.gov/technical/classification/>, accessed 2/22/2006). In general, the soils are considered poorly drained because of an argillic horizon at depths from 15 to 45 cm, with clay contents >50 percent and of smectitic mineralogy. As a result of this argillic horizon, soils in the watershed are predominantly classified as hydrologic soil groups C and D, the two highest soil runoff categories. Preliminary analyses of storm vs baseflow indicate that storm flow represents an average of 85-95 percent runoff (Alberts and others 1993).

The watershed is dominated by agricultural land uses, but does include approximately half of the town (~170 ha) of Centralia MO on the southern divide. Stream channels generally have narrow, forested riparian corridors. Crops include corn, soybean, grain sorghum, wheat, and pasture/hay. Moderate amounts of livestock, mostly beef

operations, are in the watershed. During the period of record (1971-present), agricultural practices have changed from moldboard plowing to no-till or other conservation tillage practices.

Instrumentation

Locations of all installations, sensors, and recording instruments are shown in the watershed map in Figure 1. Period of record for each is shown in Table 1. For simplicity, the figure and table will not be cited at each mention.

Runoff

At the watershed outlet and those of the two nested subwatersheds, precalibrated, broad-crested 10:1 V-notch weirs were installed during 1971. Rating curves were developed between 1971 and 1986. Stage recorders in stilling wells provided the primary measurements of water level from 1971 to 1995. The stage recorders were installed in tandem, with daily and weekly clocks on separate recorders for backup in case of failure, and to aid in reading when dynamics caused the ink trace to be complicated. Data from these charts were reduced to breakpoint flow. Electronic head measurements and dataloggers (5-min interval) were installed with autosamplers in 1993, and were considered the primary measurement starting in 1995. The weekly chart-based recorders have been maintained as a backup. Baseflow and storm flow at these weirs have been sampled for sediment, nutrients, and pesticides with grab samples and autosamplers since 1993.

At the outlet of the field watersheds (F-1, F2, and F3), precalibrated broad-crested 3:1 V-notch weirs were installed in 1992. Flashiness of events on these small watersheds prevented developing a rating curve, thus theoretical calibrations were taken from tables in the hydrology field manual (Brakensiek and others, 1979, pg 84-85), adjusted for area above the weirs at several heights above the notch. Water levels above these weirs have been measured using electronic equipment since installation.

Precipitation

Weighing, recording rain gauges were installed in 1971 in a network of up to 39 gauges across the entire Young's Creek watershed, as the USGS station at Young's Creek was originally considered part of the nested watershed design. However, after the USGS decommissioned Young's Creek in 1970, the rain gauge network was reduced to include only the area in and near Goodwater Creek. These rain gauges were standard 20-cm weighing gauges with daily and weekly charts. Data from these charts were reduced to breakpoint values. A subset of the network was winterized by removing the funnel and placing ethylene glycol in the bucket to measure rainfall equivalent of snow; the remainder were removed during the winters. In 1997, load cells were installed under the buckets of all rain gauges to automate the measurement on 2-minute intervals. Charts were continued as a backup until confidence was developed in the automated system.

Groundwater Level and Sampling

During 1971, a 2.5-km² area was instrumented with piezometers to measure the elevation of the water table. In 1977, additional piezometers were installed around the periphery of the watershed. In the early 1990's, groundwater sampling well nests were installed at several points in the watershed, including several in each of fields F1, F2, and F3. These were formally established as water quality sampling sites, at depths ranging from 2.9 to 15.7 m, with a screened interval of 1.2 m. Each well consisted of 5-cm PVC pipe with concrete collar and locking cap. Dedicated hand pumps were installed in each. See Kitchen and others (1997) for a more-detailed description of sampling procedures. While the primary purpose of these wells was for sampling groundwater, elevations of the water table were recorded before obtaining samples.

Temperature/Humidity and Evaporation

A hygrothermograph was installed in 1971. Maximum and minimum air temperature and average relative humidity were recorded daily. A Class A evaporation pan was maintained at that site, with daily wind run recorded at the elevation of the pan. This pan was read with a standard hook gauge on work days. These instruments were decommissioned in 2004, with air temperature, humidity, and wind speed being measured at an automated weather station, and evaporation calculable from weather station data.

Weather

An automated weather station was installed in 1991, with confirmed data starting in 1993. Daily maximums and minimums for air temperature and soil temperature, and precipitation were recorded by the datalogger. In addition,

hourly average solar radiation, air temperature, saturated and actual vapor pressure, wind speed, wind direction, soil temperature, soil moisture, and precipitation were recorded and downloaded on a nominal weekly basis.

Sediment

Suspended sediment was determined on runoff and grab samples collected from the creek at various times from 1972 to present. There were sporadic measurements in 1972-73 and continuous data sets from 1979-82 and from 1991-present. The evaporation method (Brakensiek and others 1979) was used over the entire period of record. The method calls for adding a flocculant for removing clay-sized particles from suspension, but instead, samples were allowed to settle over a period of days to weeks, depending upon the clay content of the samples, before gravimetric analysis was performed.

Nutrients

All surface and ground water samples were analyzed for dissolved N and P forms [(NO₃ + NO₂)-N, NH₄-N, and PO₄-P]. Dissolved nutrient analyses were determined on runoff and grab samples collected from GWC at various times from 1979 to present. There were sporadic measurements from 1979-82 and continuous data sets from 1986-89 and 1991-present. From 1991-2004, groundwater samples were collected in spring and fall from approximately 90 wells. From 2005-present, groundwater sampling continued only for the 25 wells located at F1. Samples were filtered through 0.45- μ m nylon filters within 48-72 hours of collection. Dissolved N and P species were determined colorimetrically using a Lachat flow injection system (Lachat Instruments, Loveland, CO) or an Aquakem 200 discrete colorimetric analyzer (Environmental Science Technology, Fairfield, OH).

Herbicides

Since fall 1991, all GWC samples have been analyzed for the following corn and soybean herbicides and selected triazine metabolites: acetochlor, alachlor, atrazine, deethylatrazine, deisopropylatrazine, metolachlor, and metribuzin. From 1991-1997, groundwater samples were collected in spring and fall from approximately 90 wells and analyzed for the same suite of herbicides and metabolites as surface water samples (Blanchard and Donald 1997). From 1997-2004, these same wells were sampled in spring only. Beginning in 2005, only the 25 wells in F1 continue to be sampled in the spring. Herbicides were isolated from water samples by solid-phase extraction (SPE), and terbutylazine was added as a surrogate for quality assurance. Herbicide concentrations were then determined by gas chromatography with either an N-P or mass selective detector (Donald and others 1998; Lerch and Blanchard 2003). Other compounds and triazine metabolites have also been included for selected time periods (Lerch and others 1995; Lerch and Blanchard 2003).

Data Documentation and Quality Assurance Procedures

The initial step in the data documentation procedure was to develop a detailed flow chart describing each data stream for each general instrumental method used during the history of the watershed. Each flow chart will include links to calibrations, example files, data processing programs, or more-detailed flow charts. These flow charts are a first step both toward documentation and toward formal total quality management of the data streams and are a crucial part of the metadata. After preliminary flow charts were developed for each data stream, quality assurance was initiated. This process is not yet completed.

Each data element was compared to its closest analog. For example, from 1993 until the end of the record (continuing for W-1), redundant measurements, electronic and chart-based, were made of stages at the 3 stream weirs. The time series of electronic data was plotted, and suspect points were identified visually. These points were compared to the charts, and the points were either confirmed or a decision was made to replace the automatic measurements with chart-based measurements. This process is complete for W-1. The same process will be used for W-9 and W-11. On the other hand, the chart data from 1971 to 1993 have no electronic counterpart. For that period, the closest analogs for each weir are the other two weirs, with precipitation records adding a more-distant analog. Therefore, plots of flow through the 3 weirs on a per-unit-area basis, overlaid with representative rainfall rates, will be examined visually to add confidence in the evaluation. If possible, this first-level screen will be automated.

The second data stream for which the quality assurance process is complete is that from the automated weather station. For it, the closest analog has been either the University of Missouri South Farm weather station or the Audrain County weather station. For instance, statistical comparisons of daily solar radiation indicated degradation in the calibration of the pyranometer at F1. However, these same comparisons indicated that the calibration change was consistent, which suggested that a ratiometric calibration change could be applied to the F1 data for the period.

Where the sensor completely failed, the South Farm data were used to replace the FI solar radiation data. This process was developed to fully document the database, and is described in more detail below.

In general, for every time series of data, there now exists, or will exist, a table of raw data. In addition, where equipment failures or obviously erroneous raw data are detected, there will be a replacement data table containing the best known parameter estimates to be used as substitutes for the raw data. The intersection of these two tables creates our best data series. Each record with a replaced datum has a flag placed in the data tables and is keyed to a table of exceptions to document the action. Thus, an audit trail back to the raw data is always maintained. These tables of exceptions augment the usual station notes, which describe the data when the system is working properly.

Database descriptions

The data streams are periodically loaded into an Oracle database intended for use by CSWQRU scientists, cooperators, and eventually, the general public. The details of the raw and replacement data will not be stored for public use; however, the station notes and exceptions tables will be included and the details will be provided upon request. This database pre-existed the STEWARDS project, and the steps needed to transform the tables from the Goodwater Creek format to the STEWARDS format will be developed by the time of this conference.

CSWQRU's data store is hosted on the University of Missouri's Oracle database server. The host server runs Red Hat (Linux) Advanced Server 3.0 and Oracle 9i Release 9.2.0.7.0. Our main tablespace currently occupies 200 MB, to which users have read-only access. Currently, the ARS data store consists of 1,151,246 records in 32 tables built using a normalized relational database structure. In addition, 14 data views provide depictions of the contents in formats commonly expected by users.

Future plans

The STEWARDS system is being prototyped in spring 2006, with the intent that part of the Goodwater Creek database will be available for a demonstration in early May. An update is scheduled for October 2006, with phased-in uploads of other ARS Conservation Effects Assessment Project watershed databases following as resources allow.

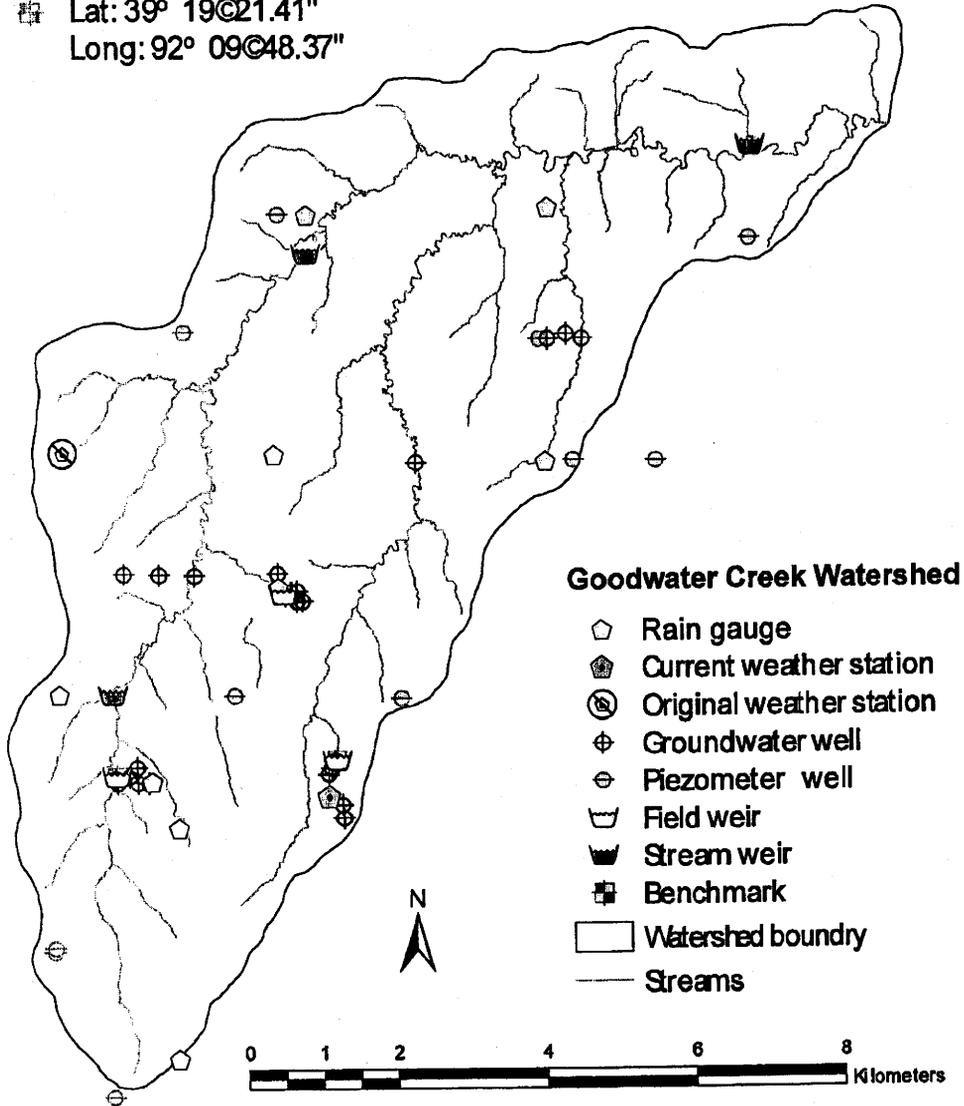
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Figure 1--Goodwater Creek Watershed map with locations of primary installations and instruments.

⊠ Lat: 39° 19' 21.41"
Long: 92° 09' 48.37"



HURRICANE IMPACT ON STREAM FLOW AND NUTRIENT EXPORTS FOR A FIRST-ORDER FORESTED WATERSHED OF THE LOWER COASTAL PLAIN, SOUTH CAROLINA

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Abstract-- A small first-order forested watershed (WS 80) within Forest Service Santee Experimental Forest near Charleston, SC has been instrumented since late-1960s to characterize its long-term hydrology and water quality as baseline information for such unmanaged coastal forests. WS 80 suffered an 80% reduction in overstory trees as a result of Hurricane Hugo (September, 1989). Stream outflows and nutrient concentrations data including calculated nutrient exports were analyzed and compared for 1976-1981 (before) and 1990-1994 (after) Hugo to examine the effects of this large-scale disturbance. Data showed an average increase in outflow by 41% for three years following Hugo, despite only a 9% increase in rainfall. Additionally, event hydrographs showed greater peak flow rates and outflows following the hurricane. Nitrogen and phosphorus exports also increased by 108-154%, primarily due to increased outflows. These data suggest that the loss of vegetation due to the hurricane resulted in reduced evapotranspiration, increasing outflow and nutrient export.

INTRODUCTION

Hydrology plays a crucial role in wetland ecosystems, affecting ecological functions, such as flood control, nutrient storage, water quality remediation, habitat restoration, and values derived by society. Wetlands, especially those in a riverine setting, have the capability to buffer flood events by storing large amounts of precipitation and slowing drainage of flood waters. They can also act as sinks for nutrients and sediments, thus reducing both the flow and nutrient release in runoff. The majority of the wetlands in southern United States is dominated by forests and is located in the coastal plain. Previous studies have indicated that forested watersheds tend to have lower flows and nutrient exports than developed or disturbed watersheds with less vegetation (Binkley and others 2004, Chescheir and others 2003, Shiang-yue 1996, Wahl and others 1997). However, nitrogen (N) and phosphorus (P) loading to water bodies are becoming a public concern; as a result, research efforts have increased to understand the role of wetlands in mediating hydrologic and biogeochemical processes (Mitsch and Gosselink, 2000).

The coastal, forested wetlands of the southern US have developed over time with hurricanes and tropical storms occurring as natural episodic events. Wind damage associated with hurricanes can alter the landscape by changing the canopy and reducing live biomass. These changes can lead to decreased evapotranspiration (ET) and nutrient uptake thereby altering the water and nutrient budgets including the stream dynamics. Hurricane-related events have produced large nutrient loads to rivers and estuaries of North Carolina where total-N and total-P loadings in the 2 months following the storm were nearly equal to the average for an entire year (Shelby et al., 2006). However, very little is actually known about long-term impacts of hurricanes on forested ecosystems and the corresponding hydrologic linkages, due to their unpredictability and lack of data to support such studies (Conner 1998).

Hurricane Hugo, a Category IV storm, made landfall near McClellanville, SC (~50 km north of Charleston) in September of 1989, causing extensive damage to the coastal forests, primarily within the US Forest Service Francis Marion National Forest (FMNF). The Santee Experimental Forest, in the western side of the FMNF, contains several gauged watersheds that have been used to study the effects of forest management practices (Harder and others 2006; Amatya and Radecki-Pawlik 2005). The small reference watershed, WS 80, has been monitored for periods over the last 40 years and the long-term data have been used as a baseline for stream outflow characteristics. The watershed suffered an 80% reduction in overstory trees and canopy as a result of Hurricane Hugo (Binkley 2001; Hook et al., 1991). Accordingly, the historic data available for this watershed provides an opportunity to evaluate the potential effects of a large-scale natural disturbance on an area largely un-impacted from human activities. Studies of this situation can lead to a better understanding of the effects of hurricanes and tropical storms on watershed response, and provide baseline data for other research. For example, data from WS 80 are being utilized as a baseline for calculating reference watershed loads needed for TMDL (Total Maximum Daily Load) development for the Charleston Harbor System (Silong et al. 2005); the East Branch of the Cooper River, one component of the Charleston Harbor System, is within a few kilometers of WS 80.

Nutrient exports were calculated by multiplying periodic (weekly to bi-weekly) N and P concentrations by corresponding amounts of cumulative outflow, and summed up to obtain totals for each of the approximate five years before and after the Hurricane Hugo. Flow weighted concentrations were obtained by dividing the total load by the total outflow for each of the years before and after the Hurricane Hugo.

Effects of Hugo on magnitude and duration of increase in stream outflows were evaluated by comparing the observed and expected annual outflows for the post-Hugo years. The expected annual outflow was calculated by using the pre-Hugo calibration relationship of annual rainfall and runoff with the post-Hugo observed rainfall. For obtaining a stronger relationship twelve years (1969-81) of pre-Hugo rainfall and stream outflows were used.

RESULTS

Stream outflows

Figure 2 represents the daily cumulative rainfall and stream outflow curves for periods before and after Hurricane Hugo. The post-hurricane outflow appears more responsive to rainfall events than the pre-hurricane period. Also, a wider gap between the outflow and rainfall curves for pre-Hugo years compared to the post-Hugo indicates less cumulative total outflow and more evapotranspiration (ET) prior to hurricane Hugo. This was true, especially, for the Days 600 to 800 and 1300 to 1700 of the pre-Hugo period when the outflow was not responsive. Both the cumulative rain curves followed the similar pattern nearly up to 500 days after which the post-Hugo rain deviated with somewhat higher rain than the pre-Hugo period. The largest deviation in rain occurred between approximately Day 600 to 650. The 4th year in both pre- (1979) and post- (1992) Hugo periods had the highest and similar amount (1419 – 1434 mm) of the rain of all the years (Table 1).

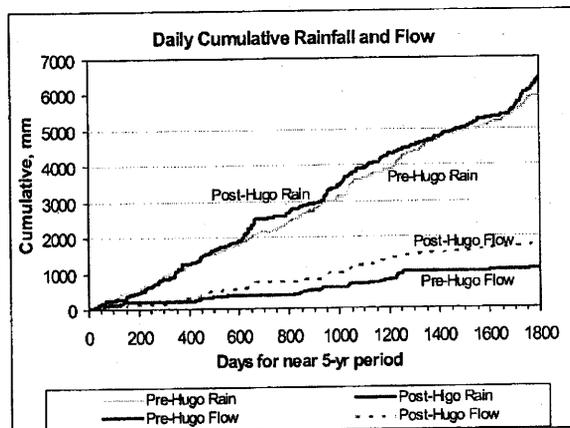


Figure 2—Daily cumulative flow and rainfall curves for WS80 for near five-year periods before (1976-80) and after (1990-94) Hurricane Hugo.

Table 1—Measured Annual rainfall, flow, gross ET calculated as a difference of rain and flow and runoff ratio calculated as a ratio of flow and rainfall for near five-year periods (1800 days) before and after Hurricane Hugo.

Period/Year	Rain	Flow	Difference	Flow/Rainfall	Expected Flow	Flow Increase	Flow Increase	Ratio Increase
	mm	mm	mm		mm	mm	%	%
(Nov 4) 1976	211.7	80.4	131.3	0.38				
1977	1114.3	141.9	972.4	0.13				
1978	1030.8	147.6	883.2	0.14				
1979	1419.2	358.4	1060.8	0.25				
1980	1225.2	299.1	926.1	0.24				
(Oct 8) 1981	906.2	58.9	847.3	0.06				
Pre-Hugo Total	5907.4	1086.4	4821.0	0.18				
Average (77-31)	1139.1	201.2	937.9	0.17				
(Nov 1) 1989	171.6	62.7	108.9	0.37				
1990	1120.9	264.6	856.3	0.24	183.9	80.7	-43.9	-28.4
1991	1351.7	434.8	916.9	0.32	305.4	129.4	-42.3	-74.9
1992	1434.4	481.9	952.6	0.34	349.0	132.9	-38.1	-82.7
1993	1008.4	327.4	681.0	0.32	124.6	202.8	-162.7	-76.5
1994	1348.6	202.3	1146.3	0.15	303.8	-101.5	33.4	18.4
Post-Hugo Total	6435.6	1773.6	4662.0	0.28	1266.8	444.2		
Average (90-34)	1252.8	342.2	910.6	0.27	253.4	88.8	88.8	-48.8

Daily flow frequency duration (FFD) curves obtained using 1800 days of measured data for both the pre- and post-Hugo years are presented in Figure 3. The data clearly indicate that the frequency of daily flows in the range of 5 mm to 40 mm was higher for the post-Hugo periods than for the pre-Hugo. For example, daily flow of 20 mm occurred 0.5% of time (9 days) for post Hugo period compared to only 0.22% or 4 days in the pre-Hugo period. Similarly, for 1% of the time or 18 days daily flows for the post-Hugo period equaled or exceeded 15 mm compared to 7.6 mm for the pre-Hugo. However, a largest flow rate of 55.8 mm day⁻¹ occurred for the pre-Hugo period compared to only 41.5 mm day⁻¹, the largest flow rate for the post-Hugo. All other high flow rates from 99.95th percentile (39.3 mm day⁻¹) to 50th percentile (0.29 mm day⁻¹) were greater for the post-Hugo period than the 31.6 mm day⁻¹ and 0.01 mm day⁻¹, respectively, for the pre-Hugo period. Similarly, the mean daily flow of 0.99 mm for the post-Hugo period was almost 65% greater than the 0.60 mm for the pre-Hugo period.

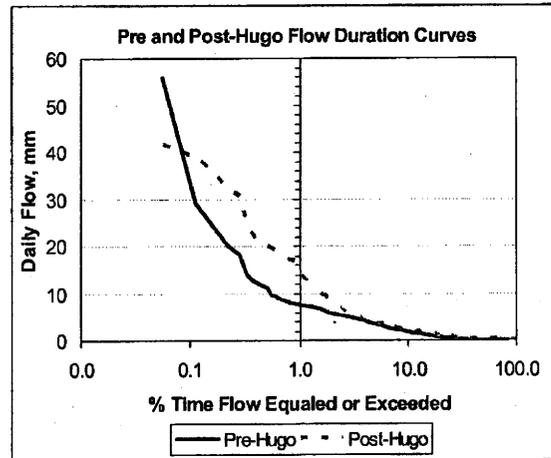


Figure 3—Daily flow frequency duration curves for near five years (1976-81) of pre and (1989-94) post-Hugo outflow data.

Cumulative outflow volume for the post-Hugo years was 1774 mm, compared with 1086 mm for pre-Hugo years, an increase of 63% (Figure 2). But the cumulative total rainfall for pre-Hugo years was 6436 mm compared to only 5907 mm for the pre-Hugo years, an increase of only 9%. The average annual runoff/rainfall ratio, a rainfall normalized outflow parameter, for the post-Hugo years was 28% with a year-to-year variation from 15 to 37%, compared to only 20% with an annual variation from 6 to 38% for 12 years before the hurricane (Figure 4). The 20% ratio for the undisturbed pre-hurricane data was 3% higher than the data (17%) reported for a natural forested wetland in eastern North Carolina (Chescheir and others 2003). The average annual estimate of outflow shows an increase of 115 mm for the period following Hurricane Hugo (Table 1) compared to the pre-hurricane data. On a year-to-year basis the increases in runoff/rainfall ratios were larger in the first three years (1990-93) after Hugo compared to the pre-Hugo average annual ratio. However, the gross ET, as a difference of rainfall and outflow, indicated an increase by only 26 mm, on average, following the Hurricane. This was due to 9% increase in rainfall for post-Hugo years compared to the pre-Hugo.

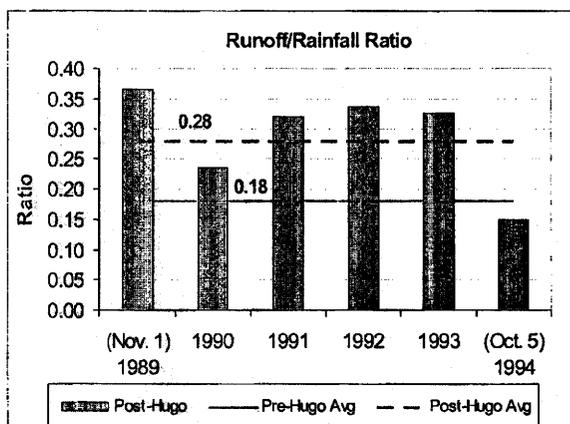


Figure 4-- Runoff-rainfall ratios for five years (1989-94) of post-Hugo compared to 12-year (1969-81) average of pre-Hugo data.

Event outflow results show a smaller mean peak flow rate before Hugo (11 mm/day), compared to after Hugo (20 mm/day) (Table 2). The pre-Hugo mean outflow/rainfall ratio was also smaller (38%), ranging from 9% to 74%, than post-Hugo (51%), ranging from 28% to 124%. The smaller mean event total rainfall for pre-hurricane events (72 mm) compared to the post-hurricane (95 mm) could have also contributed to this difference. Mean event peak flow, outflow, rain, and runoff ratio were all greater for the post-hurricane periods than for the pre-hurricane. However, based on a t-test for significance, none of these means for the post Hugo events was statistically different ($\alpha = 0.05$) from the corresponding means for the pre-Hugo events.

The pre-Hugo annual outflow versus rainfall relationship using 12-years (1969-80) of data ($\text{Flow} = 0.53 \times \text{Rain} - 406$) was strong ($R^2 = 0.90$) and significant ($p < 0.01$) as shown in Figure 5. Although the slope of the post-Hugo years (0.44) (excluding the last year

Table 2-- Event hydrograph characteristics for ten storm events prior to and after Hurricane Hugo

Period	Event Start	Event Duration	Begin Flow	Peak Flow	Time to Peak	Event Out-flow	Total Rain	Flow/Rain Ratio
		(days)	(mm)	(mm)	(days)	(mm)	(mm)	(%)
Pre-Hugo	11/25/1979	11	0.00	2.52	2	6.34	74.1	8.6
	12/05/1979	11	0.06	4.61	3	12.99	46.3	28.1
	12/15/1979	10	0.38	3.36	2	11.68	28.1	41.6
	01/08/1980	15	0.19	2.52	7	13.58	44.2	30.7
	01/21/1980	10	0.42	6.71	7	27.26	56.0	48.7
	03/11/1980	7	2.52	55.78	3	83.88	113.7	73.8
	03/26/1980	8	0.84	5.45	4	22.23	46.1	48.2
	04/07/1980	7	0.84	18.03	3	33.13	63.7	52.0
	06/23/1980	11	0.00	4.61	3	14.61	156.0	9.4
	07/02/1981	9	0.00	5.03	3	7.76	86.8	8.9
	Mean	9.9	0.53	10.86	3.7	23.35	71.5	35.0
	Std Dev	2.4	0.8	16.4	1.8	22.9	38.5	21.9
Post-Hugo	12/07/1989	12	0.35	4.19	3	23	64.5	35.7
	03/28/1990	17	0.00	5.03	5	28.83	104	27.7
	10/09/1990	6	0.00	36.07	6	72.16	197.4	36.6
	10/22/1990	22	0.39	18.03	2	32.76	70.6	46.4
	11/08/1990	20	0.04	41.52	3	62.97	50.7	124.2
	01/18/1991	6	0.84	15.94	3	29.36	66.5	44.2
	01/27/1991	11	2.10	18.45	4	46.56	79.1	58.9
	03/01/1991	12	0.32	11.74	3	34.94	66.0	52.9
	07/14/1991	5	0.02	12.58	3	25.18	81.2	31.0
	07/27/1991	16	0.01	24.33	7	49.51	136.5	36.3
	Mean	12.7	0.41	18.79	3.9	40.53	91.6	49.4

1994) was not much different than the pre-Hugo period (0.53), there was an increase in intercept by about 100 mm. The expected annual outflows calculated using this relationship with the post-Hugo annual rainfall are shown in Table 1. Clearly, the measured post-Hugo outflows were consistently higher than the expected for all years, except for the last year 1994, with an average increase of 89 mm. While the increases in first three years were within 44%, the highest increase (~ 203 mm; 163%) was observed for the year 1993 with the lowest rainfall. By 1994 the measured flow was already lower than the expected, indicating the possible return to pre-hurricane conditions five years after the natural regeneration of the forest stands. The average increase for first four years excluding 1994 was 136 mm (72%) consistent with the plots in Figure 5.

Nutrient Concentrations

Flow weighted concentrations for all nutrients for all years before and after the hurricane were lower than 1.0 mg l⁻¹, except for the TKN and total-N in 1980 for the pre-Hugo and 1994 for the post-Hugo (Table 3). This is consistent with data reported by Chescheir and others (2003) for a natural forested wetland in eastern North Carolina. Instantaneous nitrate and TKN concentrations were lower than 0.12 mg l⁻¹ and 3.0 mg l⁻¹, respectively, even immediately after the hurricane in late 1989. Total-P concentration was also lower than 0.13 mg l⁻¹. Data showed that most of the total-N and the TKN consisted of organic N, which ranged from 0.63 to 1.1 mg l⁻¹ with a mean of 0.84 mg l⁻¹ for the pre-Hugo years and 0.16 to 1.26 mg l⁻¹ with a mean of 1.01 mg l⁻¹ for the post-Hugo. The means of each of the nutrients showed higher concentrations after Hurricane Hugo compared to the pre-hurricane data, with the highest increase (39%) for the NO₃-N followed by 21% for the TKN, organic N, and total-N and 9% for total P (Table 3).

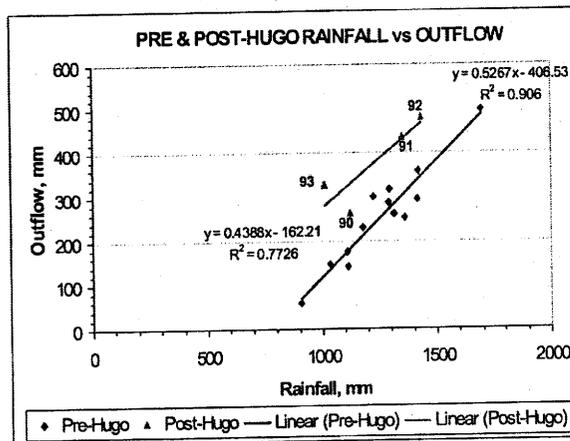


Figure 5-- Regression relationships between the annual rainfall and outflow for 12 years (1969-81) prior to Hugo and for four years (1990-93) after Hugo. Numbers 90, 91, 92, and 93 with triangular symbols represent post-hurricane data for the years 1990 to 1993.

Table 3—Average flow weighted nutrient concentrations in mg l⁻¹ for WS80 before and after Hurricane Hugo and percent change between the two periods. (Data for 1992 was not available).

Average Annual Flow Weighted Nutrient Concentrations in mg l ⁻¹							
Year	NO3	NH4	TKN	TN	DIN	ON	PO4
1976	0.013	0.030	0.000	0.013	0.044	-0.030	0.050
1977	0.012	0.025	0.885	0.897	0.037	0.861	0.000
1978	0.016	0.035	0.960	0.977	0.052	0.925	0.016
1979	0.013	0.031	0.656	0.670	0.044	0.626	0.018
1980	0.009	0.039	1.124	1.133	0.047	1.085	0.041
1981	0.027	0.036	0.707	0.734	0.063	0.672	0.018
Pre-Hugo Avg.	0.018	0.040	0.879	0.897	0.058	0.839	0.029
1989	0.006	0.025	0.761	0.767	0.031	0.736	0.023
1990	0.009	0.068	0.968	0.977	0.077	0.900	0.053
1991	0.014	0.035	0.982	0.983	0.048	0.935	0.035
1992							
1993	0.016	0.024	0.185	0.205	0.040	0.161	0.013
1994	0.055	0.012	1.270	1.326	0.067	1.255	0.002
Post-Hugo Avg.	0.026	0.042	1.060	1.083	0.067	1.014	0.032
% Change	39%	6%	21%	21%	16%	21%	9%

Nutrient Exports (Loadings)

Calculated annual loadings rates for NO₃ and NH₄ were found to be very low (Table 4) compared to other coastal forested watersheds (Chescheir and others 2003) due to their measured low concentrations (Table 3). On the other hand the TKN loadings 1.8 kg ha⁻¹ for pre-Hugo and 3.8 ha⁻¹ for post-Hugo) were higher than those found in Chescheir and others' study. TKN loadings were as high as 4.8 kg ha⁻¹ soon after the hurricane. The nutrient loadings increased after the hurricane for all nitrogen and phosphorus components (NO₃, NH₄, TKN, Total-N, DIN, ON, and PO₄), ranging from 108% for NH₄-N to 154% for NO₃-N. These increases in nutrient loadings were primarily due to the 72% average increase in outflow as the increase in average flow weighted nutrient concentrations for all nutrients were below 39% (Table 4). Although the increase in outflow in 1993 was higher, the much reduced concentrations resulted in lower loadings for TKN, organic N and total N.

Table 4-- Average annual nutrient loadings in kg ha⁻¹ for WS 80 before and after Hurricane Hugo and percent change between the two periods.

Annual Outflows and Nutrient Loads (kg ha ⁻¹)										
Year	# of days	Rainfall (mm)	Outflow (mm)	NO3	NH4	TKN	TN	DIN	ON	PO4
1976	58	211.70	80.40	0.01	0.02	0.00	0.01	0.04	-0.02	0.04
1977	365	1114.30	141.95	0.02	0.04	1.26	1.27	0.05	1.22	0.07
1978	365	1030.80	147.65	0.02	0.05	1.42	1.44	0.08	1.37	0.02
1979	365	1419.20	358.44	0.05	0.11	2.35	2.40	0.16	2.24	0.07
1980	366	1225.20	299.12	0.03	0.12	3.36	3.39	0.14	3.25	0.12
1981	281	906.20	58.87	0.02	0.02	0.42	0.43	0.04	0.40	0.01
Pre-Hugo Avg.		1198.26	220.37	0.03	0.07	1.79	1.81	0.10	1.71	0.06
1989	61	171.60	62.68	0.04	0.16	4.77	4.81	0.19	4.61	0.14
1990	365	1120.90	264.59	0.02	0.18	2.56	2.59	0.20	2.38	0.14
1991	365	1351.70	434.78	0.06	0.15	4.27	4.27	0.21	4.06	0.15
1992	366	1434.40	481.85							
1993*	365	1008.40	327.37	0.05	0.08	0.61	0.67	0.13	0.53	0.04
1994*	278	1348.60	202.32	0.11	0.02	2.57	2.69	0.14	2.55	0.00
Post-Hugo Avg.		1305.40	359.76	0.07	0.15	3.76	3.82	0.22	3.60	0.12
% Change		9%	63%	154%	108%	111%	111%	120%	110%	119%

DISCUSSION

Streamflow

This study focused on the effects of Hurricane Hugo on daily, event-based, and annual stream outflows and annual nutrient concentrations and loadings measured for near-five years on a first order-forested watershed in the lower coastal plain of South Carolina soon after the hurricane compared to the baseline data prior to the hurricane. Unfortunately, no measurements were available immediately prior to the Hurricane. Accordingly, the effects were evaluated using the baseline data at least eight years prior to the hurricane. By analyzing data in this way we assumed that the mature forest on the reference watershed was not affected by anything other than the climate from late 1981 until the hurricane hit in 1989. We are unaware of similar studies in the coastal plain of the USA, except one recently completed by Shelby and others (2006). That study compared hydrology and nutrient loadings results on two (forested and agricultural) coastal watersheds for periods during the large disturbances caused by the 1999 extreme weather conditions due to three hurricanes with those from the years immediately prior to the hurricanes. Those authors reported maximum daily flow rates as well as the loading rates during the largest Hurricane Floyd compared to all other years. Most of the nutrient losses in the year 1999 with the hurricanes occurred during two months (September-October) of the extreme weather. The four-year average nutrient loadings from the intensively managed forested watershed were much higher than the annual loadings for each of the years after the Hurricane Hugo on our study watershed (WS 80). Their study concluded that the majority of the outflows and nutrient exports from the watersheds can occur primarily through large storm events such as those caused by the hurricanes and tropical storms.

Although our study did not have data from the actual hurricane period as did the Shelby and others' (2006) study, ours' is unique in that we have examined how long the large disturbance such as hurricanes may affect the stream outflows in its increased magnitude since its occurrence. Extreme losses in vegetation due to large-scale disturbances, such as hurricanes or timber harvest, can result in higher runoff rates due to decrease in evapotranspiration (ET). Studies have shown that younger stands, such as those naturally regenerated or planted for regeneration after disturbance, will result in higher outflows due to lower ET than older and taller stands (Amatya and others 2006; Chescheir and others 2003). A few years after natural regeneration or planting the vegetation canopy is restored and the outflows and nutrient levels come back to base line levels. This study provided an opportunity to evaluate both the increase in outflows (runoff) as well as the duration of this increase. The average increase of 136 mm of outflow in our study watershed (WS 80) following the hurricane was consistent with the range of 111-164 mm increase reported by Lebo and Herrmann (1998) for their three-year study on effects of harvesting pine forests and also by Amatya and others (2006) who found an average increase of 158 mm for five years immediately after harvesting a drained pine forest both in coastal North Carolina. In addition, Sun and others (2000) showed that salvage logging after Hurricane Hugo greatly increased annual stream flow on WS77 (the treatment watershed located nearby WS 80; Figure 1). These findings tend to support a direct relationship between tree loss due to hurricane damage and a consequent loss in ET with increased outflows. Sun and others (2002), however, argued that for lowland wetland watersheds with very shallow water tables, such as those within the coastal plains, the reduction in transpiration may be counteracted to some degree by an increase of soil evaporation from exposed soil surfaces, resultant of canopy loss. The significant loss of tree canopy such as in our study due to the effects of hurricane can also cause increased surface temperatures, leading to an increase in ET, as well. Sun and others (2002) concluded that tree removal in these types of watersheds has an insignificant increase in total runoff, and thus has limited downstream impacts. But our data herein and studies by Amatya and others (2006), Lebo and Herrmann (1998) do not seem to support their findings.

Sun and others (2002) also note that vegetation in southern coastal, forested wetlands can recover quickly due to favorable soil moisture and heat conditions, thus allowing altered hydrology to return to pre-disturbed conditions relatively quickly. Our results showing at least a four year duration of increased annual outflows following the hurricane impact on vegetation is in between the Lebo and Herrmann (1998) and the Amatya and others (2006) studies. Lebo and Herrmann showed that outflow concentrations returned to baseline levels within two to three years, whereas Amatya and others (2006) showed that a pine forest in eastern North Carolina took about six years for the hydrology and three years for the nutrient concentrations to return to baseline levels since following a plantation for regeneration. In our study, herein for WS 80 by the fifth year (1994) the measured outflows were lower than the expected value, possibly indicating the return of outflows to base line levels. However, the conclusions from our results for 1993 and 1994 (Table 1; Figure 3) should be tempered with the recognition that the measured flow data for these years were not available and hence extrapolated using data from the adjacent treatment watershed (WS 77; Figure 1). For example, the very high increase (163%) in 1993 (Table 1) was somewhat unrealistic because of the constant 0.29 mm flow predicted by the regression equation with the intercept of 0.29 mm

for several days when the treatment watershed (WS 77) had zero flows during the dry summer-fall months yielding a total annual outflow of 327 mm compared to only 125 mm of expected outflow for the rainfall of 1008 mm.

The cumulative rainfall and outflow plots in (Figure 2) generally show no outflow in the summer months, unless large rain events occurred (> 75 mm). This is attributed to higher ET during the summer that lowers the water table allowing for greater soil water storage and infiltration during these months, unless large storms bring the water table to the surface. This is similar to findings in Amatya and others (2002; 2000) and other studies on coastal forests in the Southeast (Jackson and others 2004). The data analyzed for this study show that total outflow also increased after Hurricane Hugo with the increase of the amount of rainfall. Even though rainfall only showed a 9% increase for the period after the hurricane, a 72% average increase was found in the total outflow. However, the increase in outflow/rainfall ratio (a rainfall normalized outflow) ranged from 28% in 1990 soon after Hurricane Hugo to 83% in 1992 (Figure 3) with an average of 49% only. The near five-year pre-Hugo average ratio (0.18) is consistent with data reported by Chescheir and others (2003) for a natural forested wetland in coastal North Carolina.

The differences among hydrograph comparisons (Table 2) indicating higher means of peak flow rates, outflows, and runoff ratios for the post (1989-91) hurricane periods compared to the pre-(1979-81) hurricane period have not been statistically compared, and may not be statistically significant since the post-hurricane means were well within one standard deviation (not shown) of the pre-hurricane values. There are also other factors affecting our interpretations. First the pre-hurricane outflow characteristics come from a short period and most of those events were from the wet winter-early spring period when water tables are generally high and ET demands are low. The hydrologic effects of disturbances including harvesting are generally higher during summer-fall periods with higher ET demands (Amatya et al., 2006; Sun et al., 2002). It would be better to have events representing both dry summer and wet winter seasons from at least all five years (1976-81), and we intend to do that in subsequent analyses. Also, the analyses were based on daily data that may not accurately represent the peak flow rates, time to peak, as well as the outflow volumes including the rainfall amount. For example, event outflow/rainfall ratio of 124% (Table 2) indicates a potential error in the data; the potential contribution from the previous event was ruled out because of a begin flow of near zero (0.04 mm). Such analyses are best conducted using data on hourly or even finer time scale basis. Effects of the hurricane on storm event outflows can then be accurately evaluated by comparing the slope of the pre-hurricane peak flow rates (or storm outflows) and rainfall relationship with that of the post-hurricane one using statistical tests of significance as was done by Amatya and others (2000).

Increases in total outflow and peak outflows can have significant effects locally and downstream. These increases can lead to flooding, increases in nutrient export, erosion and sediment loading, and changes in ecological conditions affecting biota. Thus, it is important to interpret and to understand how disturbances, natural or anthropogenic, affect natural wetland functions and processes on these forested landscapes.

Nutrient Concentrations and Exports (Loadings)

Average flow weighted nutrient concentrations for all nutrients in this study even after the hurricane were lower than those reported for a natural wetland (Chescheir and others (2003)) and for the managed pine forests after their harvesting (Amatya et al., 2006; Lebo and Herrmann, 1998) all in coastal North Carolina. Nitrate showed the greatest increase in average concentration followed by the TKN. Our results of average annual total-N and PO_4 concentrations show increases (based on the pre-hurricane mean) of only about 0.1 mg l^{-1} and $0.01\text{-}0.03 \text{ mg l}^{-1}$, respectively, for the first two years after the hurricane. However, based on the calculated standard deviations, these increases may be insignificant. Binkley (2001), reporting data for WS80 (in longer periods), states that NO_3 , DON, and PO_4 concentrations in stream water, did not show any significant effects from Hurricane Hugo, and that ammonium showed little effect, if any. This may be explained, in part, by local characterizations of typical Santee forested watersheds. Low soil permeability, subdued topography, and large surface area and adsorption capacity throughout the soil profile, cause reduction of the transport of released dissolved solutes to streams and groundwater (Richter 1980). This could account for small changes in nutrient concentrations with increases in outflow as the major contributing factor to increased nutrient export.

Nutrient export in forest drainage is a combination of outflow volume and nutrient concentration in the drainage water (Chescheir and others 2003). Nutrient exports have been shown to vary seasonally, including effects from large tropical storms and hurricanes and associated excessive rainfall (Shelby and others 2006). This study does not consider actual increased nutrient loadings as a result of the hurricane as was done by Shelby and others' study. However, the results obtained herein for years immediately after the hurricane are important in understanding of

how long these coastal forested watersheds may continue to respond to the events after the large disturbance. In this study, the data shows increase in means of all post-Hugo nutrient loadings compared to the means of the pre-Hugo data (Table 4). Our results of annual total-N and PO₄ loadings show increases (based on the pre-hurricane mean) of 0.8 to 3 kg ha⁻¹ and 0.08-0.10 kg ha⁻¹, respectively, for the first two years after the hurricane compared to 2.1-2.2 kg ha⁻¹ and 0.12-0.36 kg ha⁻¹ reported by Lebo and Herrmann (1998) in their study following tree harvest in coastal watersheds. Amatya and others (2006) also reported a two-fold increase (7.0 kg ha⁻¹) in total-N soon after harvesting a drained pine forest in coastal North Carolina. But no increase in total-P was observed. The increases of nutrient loadings in our study lasted for only two years after the hurricane. However, variations within soil types, management practices, climate, and landscape, need to be considered in detail for such inter-site comparisons.

The average increase in nutrient loadings due to hurricane effects in our study varied from as much as 154% for NO₃-N to 108% for NH₄-N, while the nutrient concentrations showed an average increase of only 39 % for NO₃-N and 6% for NH₄-N. The fact that the concentration increases due to the hurricane were insignificant as reported by Binkley (2001) clearly indicates that the increase in nutrient exports were primarily due to increase in outflows. A general understanding is that less live vegetation will lead to less nutrient uptake for plant growth and storage, and also an increase in decomposing fallen trees could cause a larger amount of nutrient input to the stream. Binkley and others (2004) compared nitrogen and phosphorus concentrations for over 300 forested streams and looked at variations in stream water chemistry from forest disturbances. Overall, no significant increases were observed in nitrate, ammonium, or phosphate concentrations due to forest harvesting. Richter (1980) also found no evidence of increased concentrations due to prescribed burning of part of these watersheds. However, the difference of this study from the harvested sites is that fallen trees stayed on the site after the hurricane letting them decompose compared to their complete removal for a harvested case thereby removing large amounts of nutrients stored in the system.

It is important to consider the contribution of surface and subsurface flow to total outflows when trying to understand fluctuations in nutrient concentrations and loadings, as these compartments have varying processes and pathways that may affect the type and amounts of nutrients in the soils and outflows (Miwa and others 2003). Although this separation is not explored within this study, a detailed water budget could lend insight as to how the nutrient loadings were affected. It is possible that NH₄-N showed a higher increase than other nitrogen components because it is the primary form of mineralized nitrogen in most flooded wetland soils (Mitsch and Gosselink 2000). This was true for the year 1990 after the hurricane where the increase was greatest 73% for the NH₄-N. Both NO₃-N and NH₄-N are mobile in water, and increased outflow and flushing could cause higher loadings of these components. However, NO₃-N decreased in concentration (Table 3), leading to a lower total loading than NH₄ (Table 4), which increased in concentration. Increases in PO₄, the dissolved phase of phosphorus, may be attributed to increased outflows and flushing as well. As expected, increase in nitrate loadings on this naturally drained watershed were very small (<0.03 kg ha⁻¹) compared to the increases of as much as 3.0 kg ha⁻¹ observed soon after harvesting a drained pine forest in Coastal North Carolina (Amatya and others 2006). This was due to reduced NO₃-N concentrations on our site dominated by shallow subsurface and surface drainage compared to the North Carolina watershed where subsurface drainage is a dominant outflow process. The pH within the soils and outflow can also play a large part in controlling the type and amounts of nutrients, as well as the amount of organic matter present.

The amount of organic matter within the soil affects the hydraulic conductivity, and consequently the subsurface drainage rate and volume. It also increases the potential to affect the amount and forms of nutrients exported. As mentioned earlier, a large quantity of decomposing vegetation in the watershed should result in increased organic matter content within the soils and in surface runoff. It would be beneficial to examine this process, if possible, for potential relationships between the soil organic matter and peak flows and total outflows.

Nutrient loadings can also have great impacts locally and downstream. Amount and types of nutrients can largely affect biota in and around the water. Large inputs of N and P in coastal streams can eventually lead to algal blooms and hypoxic conditions downstream in estuaries and oceans. However, forested wetland systems have the potential to buffer these types of inputs and act as sinks. Thus continued research of these hydrological and biogeochemical processes including the in-stream transport and their interactions with natural (hurricanes) or manmade (harvesting) disturbances will lead to a better understanding of the various functions these natural wetland functions provide.

SUMMARY AND CONCLUSIONS

A study was conducted to examine the effects of Hurricane Hugo on the post-hurricane outflows and nutrient export from a first-order forested watershed located in coastal South Carolina. The hurricane that occurred in September 1989 destroyed almost 80% of the forest canopy on the otherwise undisturbed watershed composed of matured pine

hardwood forest. Five years (1976-81) of data eight years prior to and five years (1989-94) immediately after the hurricane were used as pre- and post-disturbance data, respectively, for the evaluation. It was concluded that the post-hurricane annual increase of as much as 133 mm outflow (or 44 %) at least up to three years after the hurricane was most likely due to the loss of healthy vegetation canopy and the subsequent decrease in evapotranspiration. Limited data on concentrations showed almost no effect on most of the post-hurricane nutrient concentrations, except for the $\text{NH}_4\text{-N}$ in the first year after the hurricane. It was also concluded that the large increases in $\text{NH}_4\text{-N}$ and TKN loadings up to two years after the hurricane were primarily due to increase in outflows. Limited two years of post-hurricane event hydrograph characteristics data did not show any substantial increases in outflow characteristics. Further research and analysis using statistical tests with longer-term data and event hydrograph characteristics in a finer time scale in the context of water table levels, and organic matter content, and pH could contribute to a better understanding of the hydrologic and nutrient cycling processes and their export in these systems and their overall implications.

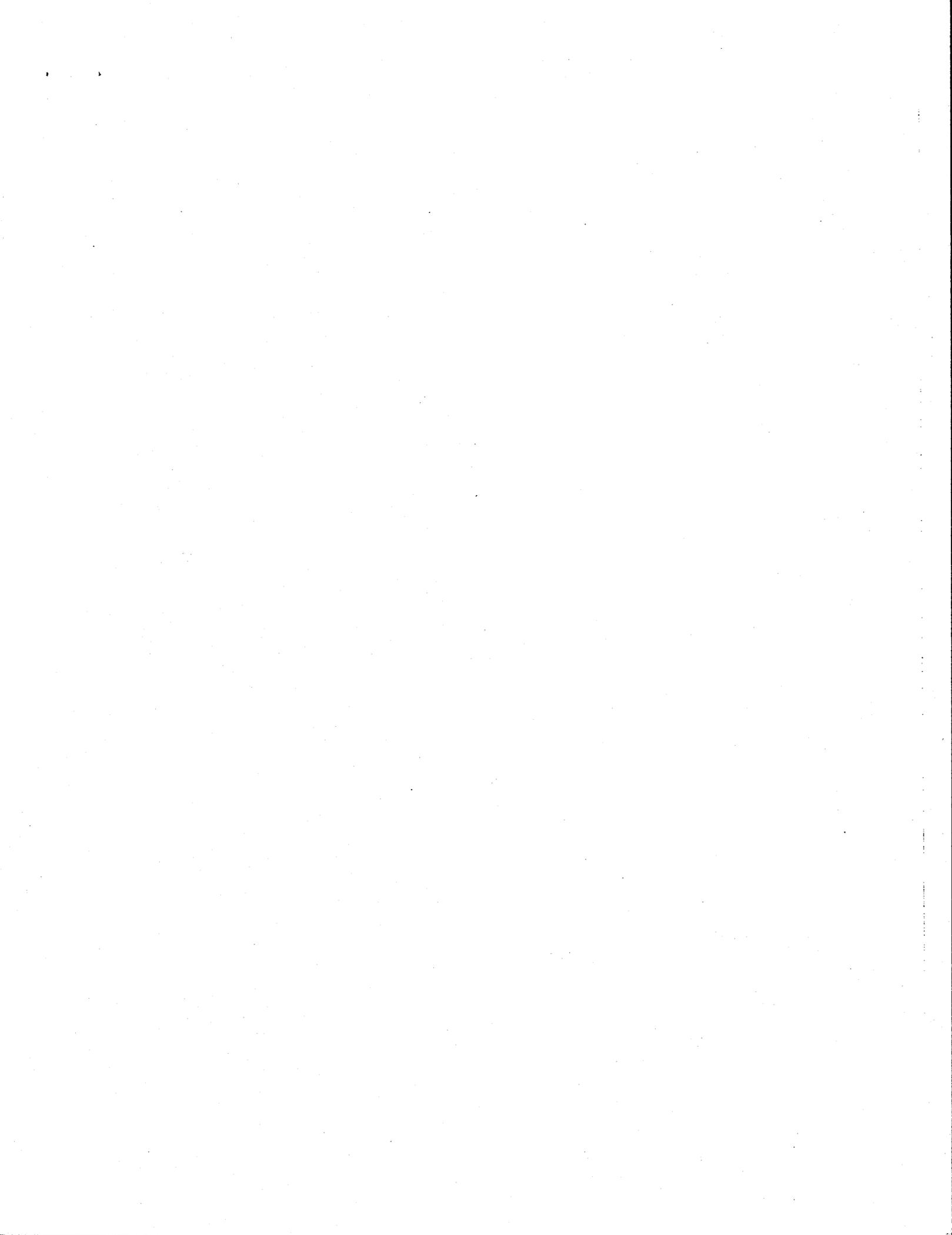
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EASTERN HEMLOCK TRANSPIRATION: PATTERNS, CONTROLS, AND IMPLICATIONS FOR ITS DECLINE IN SOUTHERN APPALACHIAN FORESTS

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Abstract--Eastern hemlock, a principal riparian and cove canopy species in the southern Appalachian Mountains, is facing potential widespread mortality due to the hemlock woolly adelgid (HWA). To estimate the impact that the loss of this species will have on forest transpiration (E_t) we quantified whole-tree (E_c) and leaf-level (E_l) transpiration over a range of tree sizes (9.5 - 67.5 cm or 3.7 - 26.6 in) during 2004 and 2005. Maximum rates of E_c varied by diameter, with large trees transpiring a maximum of 186 kg (or 49 gal) water tree⁻¹ day⁻¹. Large trees had higher maximum rates of instantaneous E_l compared to small trees (1.99 versus 1.54 mmol m⁻² s⁻¹). With increasing HWA infestation, regardless of leaf area, trees are expected to have declining transpiration rates. Hemlock mortality could reduce annual- and winter-spring E_t by 10 and 30 percent, respectively. The lack of an evergreen riparian canopy species will alter the dynamics of E_t and stream discharge.

INTRODUCTION

Individual tree species can exert enormous control on forest transpiration and interception rates, and on the intra-annual dynamics of these two processes (see Bosch and Hewlett 1982, Swank et al. 1988, Pataki and Oren 2003, Moore et al. 2004, Ewers et al. 2005). Differences in transpiration rates among species arise from both structural and physiological adaptations, such as: leaf habit and phenology (Oren and Pataki 2001), stomatal and leaf hydraulic conductance (Sack and Tyree 2005), stomatal sensitivity to vapor pressure deficit (Oren et al. 1999), and differences in sapwood area and leaf area (Wullschlegel et al. 1998, Meinzer et al. 2005). The spatial location of individual species can also influence the magnitude and dynamics of the hydrologic budget. For example, species that predominately grow in areas with stable access to water may potentially transpire longer or at greater rates compared to species without access to stable water sources (Dawson 1993). Thus, on short and long temporal scales, the loss of a single forest tree species from a catchment or landscape can impact the hydrologic budget. Furthermore, depending on the ecology and physiology of the extirpated species, the magnitude of impact on the hydrologic budget will vary.

Many eastern North American forest tree species have been eliminated, or reduced in dominance as a result of insect and pathogen outbreaks (Allison et al. 1986, Ellison et al. 2005). Although the loss of forest species has occurred several times in the past, the impact of their respective losses on the hydrologic cycle is unknown. At present, *Tsuga canadensis* L. (eastern hemlock) trees are declining and facing potential extirpation throughout their range from an introduced insect, the hemlock woolly adelgid (HWA, *Adelges tsugae* Annand). Although the present infestation ranges from Maine to Georgia and west to Tennessee, the rate of HWA dispersal and tree decline is most pronounced at the southern extent due to the putative non-lethal winter temperatures on HWA populations (Skinner et al. 2003). Thus, forests in the southern extent of eastern hemlock's range will likely experience the first hydrologic consequences resulting from its potential decline. In addition, eastern hemlock is one of the principal riparian and cove canopy species in the southern Appalachian Mountains, and commonly the only evergreen canopy species in mesic sites (K. Elliott unpublished data, Brown 2004). Thus, it is likely an important species in terms of direct and indirect effects on hydrologic processes. To our knowledge, estimates of eastern hemlock water use and transpiration do not exist for the southern Appalachians. In the northeast U.S., sap flow and stomatal conductance rates for eastern hemlock (which occur not only in riparian areas, but also as almost pure stands across the landscape) are less than 20% of co-occurring dominant hardwood species (Catovsky et al. 2002). Because eastern hemlock is typically concentrated in riparian areas in the southern Appalachians, these reported rates may underestimate the impact that eastern hemlock mortality could have on the hydrologic budget in southern Appalachian ecosystems.

We have a unique opportunity to document the ecological role of *Tsuga canadensis* on hydrologic processes prior to HWA induced mortality, and to use this information to predict the consequences its loss may have on future hydrologic cycling processes. In this study, we focused specifically on the transpiration component of the hydrologic cycle, as this component alone constitutes 30-40% of the water budget in southern Appalachian systems (Swift et al. 1975). Working in typical habitat at the southern limit of this species' range, our goals were 1) to evaluate whole-

tree transpiration over a range of tree sizes and environmental conditions, and describe the temporal dynamics; and 2) to use a model of transpiration to estimate the change in transpiration with the loss of hemlock. We conclude with a discussion of the potential impact of losing this species on the hydrology of the system.

METHODS

Study site

The study site was located in the riparian corridor (~700 m asl) along Shope Fork, a third order stream draining the Coweeta Basin in the Nantahala Mountain Range of western North Carolina, USA. Species composition in riparian corridors and mesic coves in this area is dominated by: eastern hemlock (50% of the basal area); rosebay rhododendron (*Rhododendron maximum* L., 2000 stems ha⁻¹ & 5% basal area), an ericaceous woody shrub; and sweet birch (*Betula lenta* L. & 5% basal area) (Brown 2004). The remaining 40% of basal area is composed of various hardwood species, including *Quercus* spp., *Carya* spp., *Nyssa sylvatica*, and *Liriodendron tulipifera*; however, their frequency and density are not consistent in these areas. Climate in the Coweeta Basin is classified as marine, humid temperate (Swift et al. 1988). Average annual precipitation on the valley floor of the basin is 1821 mm; and mean annual temperature is 12.6°C (Swift et al. 1988).

Sap flux measurements

During April 2004 and November 2005 we monitored sap flux density on 16 trees (Table 1) by installing thermal dissipation probes (Granier 1985) at breast height in the outer 2 or 3 cm of the functional xylem. Each tree monitored had two sets of sap flux density probes installed circumferentially. Before insertion into the xylem, probes were coated with thermally-conductive silicone grease. Areas around the probe insertion points were protected with reflective insulation to shield probes from solar radiation, thermal gradients, and rainfall. Dataloggers queried the sensors every 30 s, and logged 15 min means (Model CR10X, Campbell Scientific, Logan, UT, USA). Probe output was converted to sap flux density using the equation of Granier (1985). For all trees, readings for the two replicate sets of sensors were averaged. We routinely replaced sensors if null, out of range, or negative readings were recorded, or if probes were physically damaged.

Allometry & Scaling

We estimated sapwood area of monitored trees from DBH vs. sapwood area relationships developed on 12 hemlock trees ranging 10.0 to 65.5 cm (3.9 to 25.7 in) DBH growing in riparian forested areas at Coweeta. From these data, both heartwood, and heartwood + sapwood radii could be predicted as a function of over-bark DBH ($R^2 = 0.97$, $P < 0.01$). Over-bark DBH was recorded for all measured trees in the winter of 2004 and 2005.

To scale sap flux density measurements made in the outer 2 or 3 cm of sapwood to whole-tree sap flow, we characterized the radial distribution of sap flux density on two trees in a similar site (Table 1) (Ford and Vose, James et al. 2001). From these known distributions, we modeled the radial distribution of sap flux density in the 16 trees. Total sap flow or whole-tree water use ($\text{g H}_2\text{O s}^{-1}$) in the 16 trees was calculated as the sum of the products of sapwood area and sap flux density at each radial depth.

Table 1-- *Tsuga canadensis* tree characteristics estimated or measured during 2005

	Small (n=2)		Medium (n=9)	Large (n=5)
	Min.	Max.	Avg.	Avg.
DBH (cm)	9.5	10.6	32.1	56.9
Projected leaf area ^a (m ²)	18.51	21.92	126.60	294.06
Sapwood area ^a (cm ²)	95.6	110.0	496.0	1106.2
Biomass ^a (kg)	30.42	38.52	474.54	1460.42
Height ^a (m)	8.14	8.48	15.10	22.71

^a Denotes predicted parameters based on equations in Santee and Monk (1981).

Climate data

An open-field climate station, located approximately 1 km away from the site, recorded average hourly values of air temperature (T), relative humidity (RH; model HMP45C, Campbell Scientific, Inc.), and global radiation ($W m^{-2}$, model 8-48, Epply Lab Inc., Newport RI). From ambient T, we calculated saturation vapor pressure (e_s) (Lowe 1977); and from RH and e_s we calculated actual vapor pressure (e_a). Air vapor pressure deficit (VPD) was calculated as the difference between e_s and e_a .

Modeling

We used a time series model constructed and validated on these data station (see Ford et al. 2005 for details) to predict daily transpiration from the environmental variables in five riparian stands throughout the basin for 2004. These stands represented typical riparian and cove habitat, and in them hemlock frequency ranged 25-78% and density ranged 250-1038 stems ha^{-1} (K. Elliott and J. Vose, unpublished data). Mensurational data from these stands combined with climate data from co-located meteorological stations were used to generate transpiration model output for hemlock. We compared the magnitude of hemlock annual and seasonal modeled transpiration in these stands to known annual and seasonal transpiration rates in the basin.

RESULTS AND DISCUSSION

Impacts of Hemlock Mortality on Transpiration

Our results show important differences between eastern hemlock in the southern Appalachians and those in the northern extent of the range, both in the seasonal transpiration pattern and the magnitude of transpiration rates. Differences in seasonal patterns can be explained by environmental variables, while differences in transpiration rates can be explained by habitat.

First, we found that for all size classes of trees, the seasonal pattern of eastern hemlock transpiration had a unimodal distribution, with a high peak in the spring when the leaves of co-occurring hardwood species have not yet emerged (Figure 1) and low rates of transpiration in December and January. The largest trees had a maximum rate of 178 and 186 $kg\ water\ tree^{-1}\ day^{-1}$ in 2004 and 2005, respectively; while the smallest trees had a maximum rate of 16 and 7 $kg\ water\ tree^{-1}\ day^{-1}$ in 2004 and 2005, respectively. Year round water use by this species was a function of the evergreen leaf habit and relatively mild winter temperatures typical of the southern Appalachians. Eastern hemlock retains 3 to 6 needle age classes, so whole-tree seasonal transpiration patterns are less influenced by seasonal variation in leaf area relative to other conifers in the southern Appalachians. Hadley (2000) similarly reported peaks in gas exchange of understory hemlock in spring and fall due to greater light penetration when co-occurring deciduous trees had not yet leafed out (Figure 1). Although the transpiration rates that we measured for hemlock were within the range of those reported for other southeastern forest tree species (Oren and Pataki 2001), spring transpiration rates far exceeded the reported range, and fell within the range of rates reported for riparian tree species (e.g., *Salix spp.* and *Populus spp.*) (see table II in Lams and Muller 2002). This contrasts with rates of spring transpiration for hemlock in the northeast. Comparing similar sized trees, Catovsky et al. (2002) reported a mean daily sap flow rate during April of 7.4 $kg\ H_2O\ tree^{-1}$, while we found that the mean daily sap flow rate during April was 27.7 $kg\ H_2O\ tree^{-1}$. We also found that relatively high rates of transpiration are sustained throughout the winter months. Many studies do not measure winter transpiration; however, the few studies that have (Ellsworth 2000, Martin 2000) show that in temperate forests with an evergreen species component, winter transpiration can be significant. In contrast to our findings, Catovsky et al. (2002) could not detect winter transpiration for eastern hemlock in the northeast. The lack of transpiration could be explained in part by low temperatures. We found that low morning temperatures in the winter reduced transpiration rates and whole tree daily water use (data not presented). Furthermore, we found that when minimum morning temperatures fell below $-10^{\circ}C$, mid-day hemlock transpiration was significantly reduced regardless of daily PAR and VPD conditions (data not presented). This suggests that damage to the leaves or vascular system (i.e., freezing-induced cavitation) may have occurred and reduced the transport capacity of the trees during the day.

The distribution of hemlock primarily in riparian areas in southern Appalachian ecosystems has two important hydrologic implications. First, the loss of hemlock in the southern Appalachians may have a greater hydrologic impact than losses in other ecosystems where hemlock is distributed more uniformly across the landscape. Second, the riparian zone distribution may explain differences in reported rates of transpiration for hemlock located on drier sites. During this study, we analyzed periods in between rain events for evidence of mid-day depression in transpiration, and found none (data not presented). Furthermore we saw no evidence of a decline in transpiration

with increasing length between precipitation events (maximum length: 17 days). This suggests that these trees had stable access to water resources, either from rooting in the saturated zone of the soil (water table) or that soil moisture in the functional rooting zone never declined to critical water potentials.

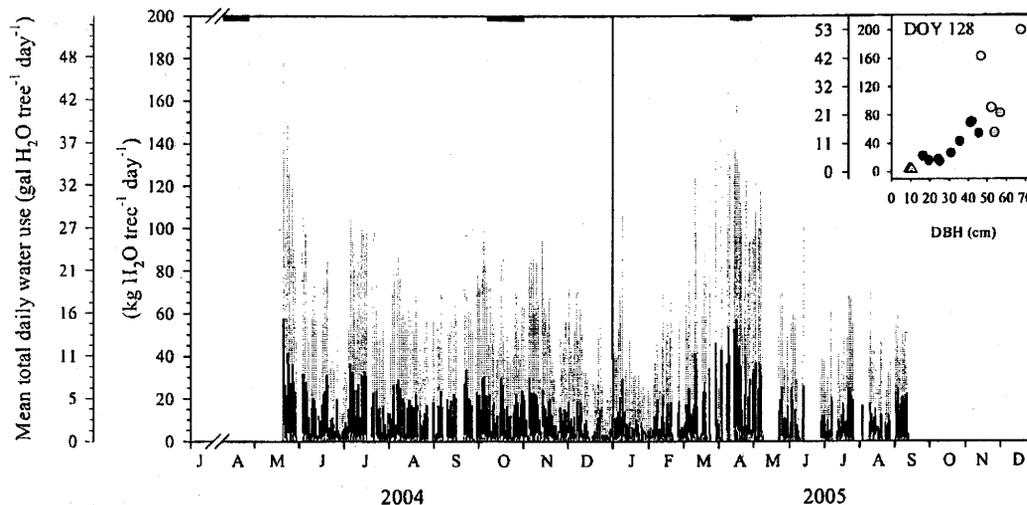


Figure 1-- Mean daily water use over the study period by large (grey bars in background), medium (vertical black bars) and small (grey bars in foreground) diameter size-class *T. canadensis* trees (see Table 1). Horizontal black bars show period of leaf expansion and fall of co-occurring deciduous trees. Gaps in record are from equipment failure. Inset shows daily water use for all trees by size class for day of year (DOY) 128 (April) in 2005.

Impacts on stand-level water budgets

In typical riparian and cove stands we predicted an average annual transpiration rate of 63.3 mm yr^{-1} for the hemlock component. Approximately 50% of this annual total was transpired in the winter and spring (9.1 mm yr^{-1} and 25.9 mm yr^{-1} , respectively). We do not have estimates of stand-level transpiration specifically for riparian areas; however, watershed-based estimates of transpiration for Coweeta hardwood stands that contain mixtures of deciduous hardwoods, pine, hemlock, and evergreen understories are on the order of 600 to 700 mm yr^{-1} (Swift et al. 1975, Vose and Swank 1994), with winter and spring transpiration approximately 15% of the annual total ($\sim 100 \text{ mm yr}^{-1}$). If we apply the watershed-level estimates to the riparian areas as a first approximation, hemlock mortality would reduce annual stand-level transpiration by roughly 10%, and reduce winter and spring stand-level transpiration by roughly 30%. We would expect evapotranspiration to decrease even more due to decreased interception capacity resulting from needle loss and eventual decay of standing dead hemlock snags. Combined, a reduction in transpiration and interception could, at least in the short term, result in 1) increased soil moisture, 2) increased discharge, 3) decreased diurnal amplitude of streamflow, and 4) increased width of the variable source area (see Dunford and Fletcher 1947, Bren 1997). Longer-term hydrologic responses will most likely be determined by post-mortality successional patterns (discussed below).

Post-Mortality Shifts in Species Composition and Possible Hydrologic Consequences

Combining historical and current species distributions, we predict two different scenarios to occur with the potential decline in eastern hemlock in the southern Appalachians. First, on sites with a dense *R. maximum* subcanopy, post-hemlock mortality seedling recruitment of any species into the canopy will likely be low (Clinton and Vose 1996, Beckage et al. 2000, Nilsen et al. 2001, Lei et al. 2002); however, *R. maximum* biomass increases will likely occur with increased resource availability. Despite a predicted increase in *R. maximum* biomass, watershed-scale experiments suggest that hemlock mortality will result in long-term decreases in riparian forest transpiration because low leaf conductivities to water vapor in *R. maximum* (Nilson 1985, Lipp and Nilsen 1997) limits responsiveness at the watershed scale. For example, in 1948-49 all of the *R. maximum* and *Kalmia latifolia* (mountain laurel) was cut

in a 28 ha watershed at Coweeta, representing 22% of the watershed basal area (Johnson and Kovner 1956). Most of the *R. maximum* was distributed in cove and riparian areas while the mountain laurel was distributed in drier ridge sites. After cutting, mean annual watershed evapotranspiration (ET) only decreased 6%, and the average increase in annual streamflow was 4%. Despite the amount of basal area cut, this small decrease in ET reflects conservative water use by *R. maximum*.

A second outcome may be expected on sites with little to no *R. maximum* subcanopy. Historical pollen evidence indicates that when the hemlock (5400 BP) and chestnut waned (early 1900's) in dominance, birch (an early successional species) increased first, followed by red maple and oaks (later successional canopy species) (Allison et al. 1986). Hence, we expect early successional species (i.e., *B. lenta*) to increase in dominance (Orwig and Foster 1998), then eventually be replaced with late-successional canopy species. Ecologically, sweet birch is relatively shade tolerant and takes advantage of gap openings and N patches for regeneration (Crabtree and Bazzaz 1993); however, sweet birch is a short-lived species and over longer time scales, other species also common to riparian corridors in the southern Appalachians may increase in dominance, such as *Nyssa sylvatica* L. (black gum) and *Liriodendron tulipifera* L. (yellow poplar) (Brown 2004). With these eventual species changes, the magnitude and timing of transpiration will also change. We predict that decreases in the amount of riparian forest leaf area, and the seasonal leaf area dynamics (deciduous vs. evergreen) will result in at least a short-term increase in transpiration per unit leaf area, an overall decrease in riparian forest transpiration, and profoundly decreased rates of winter and early spring transpiration. These changes are likely to have significant impacts on soil moisture dynamics within the riparian zone, with the potential subsequent impacts on nutrient and carbon cycling processes.

CONCLUSIONS

The potential for widespread losses of forest tree species due to nonnative invasive insects and diseases is a growing concern (Ellison et al. 2005). Understanding the functional implications, such as impacts on hydrologic processes, of losing individual species in complex ecosystems is a challenging task. For HWA, natural resource managers are now faced with the daunting task of trying to control the rate of spread and impacts of HWA at landscape scales. If control efforts are not successful, the next task will be to decide on appropriate restoration-based management actions. We contend that both control and restoration efforts should be guided by an understanding of both the structural and functional consequences. Our study shows that hemlock in the southern Appalachians has two distinct ecohydrological roles: one role is an evergreen tree that has relatively stable water fluxes throughout the year; the other is a riparian area tree with high rates of water flux in the spring. It is probable that no other native tree species will fill these ecohydrological roles if hemlock is lost from the southern Appalachian ecosystems.

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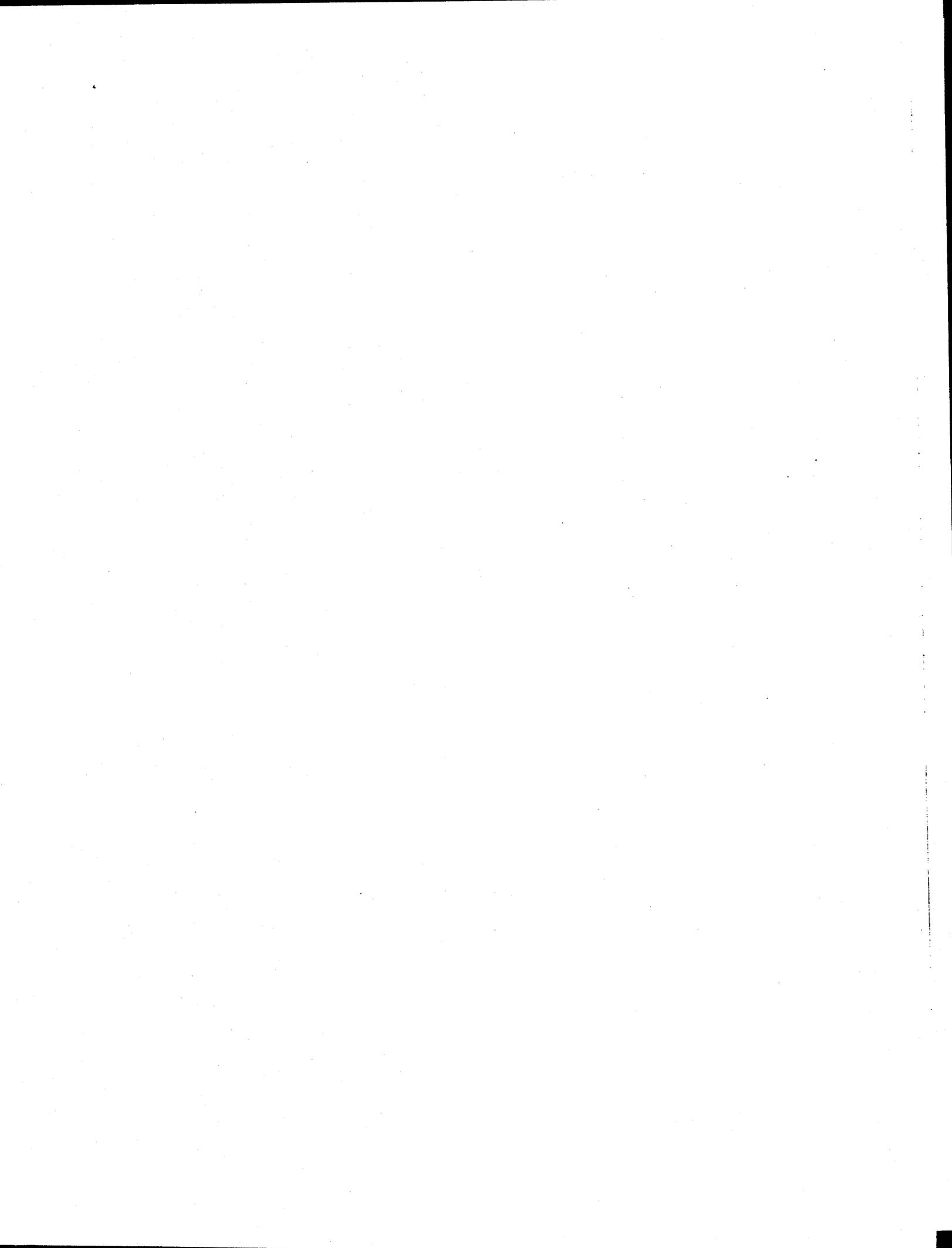
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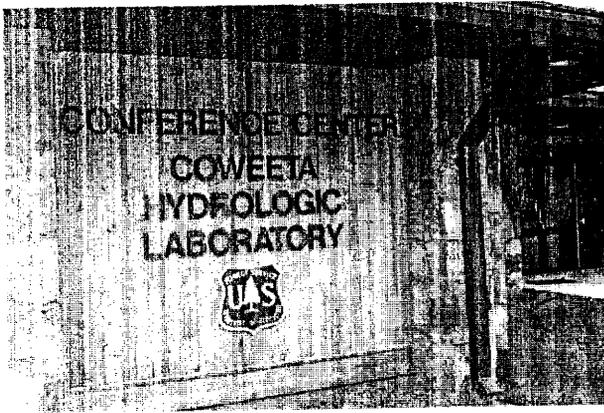
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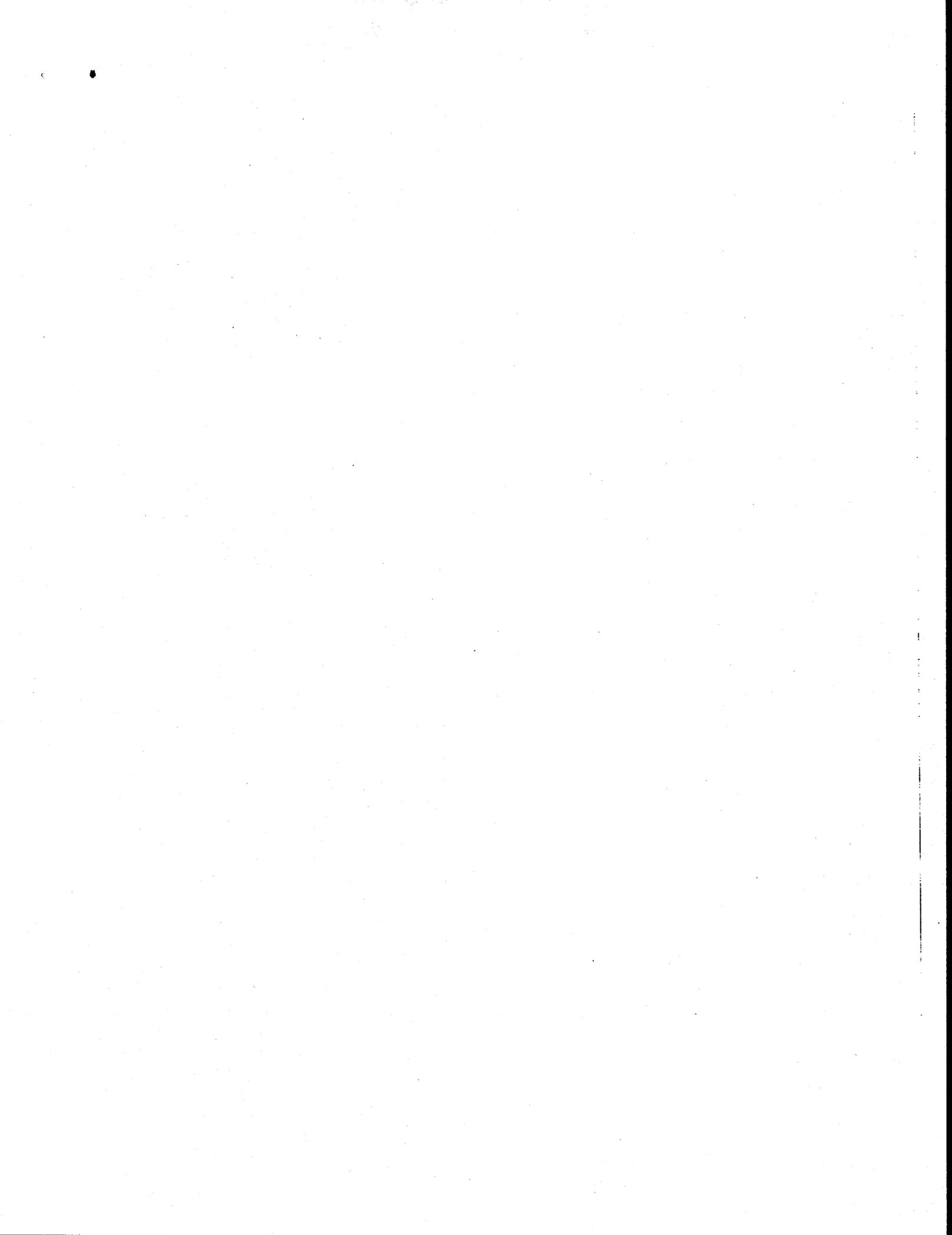
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2nd Interagency Conference on Research
in the Watersheds, May 16 – 18, 2006
Coweeta Hydrologic Laboratory, Otto, NC





**2nd Interagency Conference on Research in the Watersheds
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Presentations

Topic Title - Agency Updates

Agency Update – USDA Forest Service – Research and Development
Deb Hayes – National Program Leader for Watershed Research, Washington DC

Agency Update – USDA Agricultural Research Service
Mark Weltz - National Program Leader, Washington DC

Agency Update – USDA Forest Service – National Forest System
Ted Geier – Planning Hydrologist, Region 9, Milwaukee, WI

Agency Update - USDA Natural Resources Conservation Service
Tim Sweeney – Water Resource Planning Specialist, National Watershed
Management Center, Little Rock, AR

Agency Update - US Environmental Protection Agency, Region 4
Bill Cox – Chief of Watershed Management Office, Atlanta, GA

Topic Title - Challenges and Opportunities for Watershed Research

Regional Forested Watershed Response to Global Change
John Hom – Deputy Program Manager, Global Change Program
USFS - Northern Research Station

Hydrology and Forest in the West: A Brief Look at Current and Future Issues
Kelly Elder – Research Hydrologist
USFS – Rocky Mountain Research Station

Land Use and Future Water Supply
Jim Vose – Project Leader, Coweeta Hydrologic Laboratory
USFS - Southern Research Station

Topic Title – Introduction to the Coweeta Hydrologic Laboratory

The Coweeta Hydrologic Laboratory – A Brief History
Jim Vose – Project Leader, Coweeta Hydrologic Laboratory
USFS - Southern Research Station

Topical Area - Hydrologic Data Collection and Modeling

An ARS Data System for Assessing Conservation Practices in Watershed Study
Jin-Song Chen – Hydrologist
USDA ARS – Grazinglands Research Laboratory, El Reno, OK

Hydrologic Model Uncertainty Associated with Simulating Future Land-Cover / Use Scenarios
Darius Semmens – Research Scientist
US EPA – Office of Research and Development, Las Vegas, NV

Watershed Soil Information Derived from High-temporal-frequency Soil Moisture and Temperature Measurements
Susan Moran, Hydrologist
USDA ARS - Southwest Watershed Research Center, Tucson, AZ

A New Sensitivity Analysis Framework for Model Evaluation and Improvement, Using a Case Study of Model RHEM

Haiyan Wei, Research Technician
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Forest Management Effects on Extreme Flooding Events: Evaluating Hydrologic Modeling Approaches

Mary B. Adams, Project Leader
USFS NERS – Timber and Watershed Laboratory, Parsons, WV

Emergent Properties and Dominant Process with Changing Scales: An Application of Nonlinear System Identification Method for Rainfall-runoff Modeling

Daehyok Shin, Postdoctoral Researcher
UNC-Chapel Hill, Chapel Hill, NC

Modeling Hydrologic Processes and Responses to Disturbances in Contrasting Forested Watersheds in the Southeastern U.S. with the MIKE SHE Model

Ge Sun, Research Hydrologist
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Evaluation and Application of the MIKE SHE Model for a Cypress-Pine Flatwoods Watershed in North Central Florida

Jianbiao (Jeff) Lu, Hydrologist
Ecological Solutions, Inc., Roswell, GA

Scaling the Effects of Nitrogen Deposition from the Fernow Watershed to the Region

John Horn, Deputy Program Manager, Global Change Program
USFS NERS - Global Change Program, Newtown Square, PA

Topical Area - Fire Effects on Watersheds

Fire Effects on Forested Watersheds

Katherine Elliott, Research Ecologist
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Evaluations of the Impacts of the Okanagan Mountain Park Fire on In-stream Wood and Other Watershed Processes

Adam Wei, Associate Professor / Endowed Watershed Management Chair
University of British Columbia – Okanagan, Kelowna, BC

Influence of Fire Intensity on Streamwater Conditions in Rocky Mountain Montane Forest Watersheds

Chuck Rhoades, Research Biogeochemist
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Topical Area - Watershed Management, Planning and Regulation

Research and the National Forest Plan Revisions and Challenges

Dick Rightmyer, Forest Soil Scientist
USFS – Chattahoochee - Oconee National Forest, Gainesville, GA

The Comparison of Two Watersheds Using a Pacific Northwest Watershed Loading (PNWL) Model

Walter Frick, Research Oceanographer
US EPA - Ecosystems Research Division, Athens, GA

Research Summary of Organic BMPs Used in Storm Water Management

Britt Faucette, Research Ecologist
Director of Research and Development
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Georgia's Total Maximum Daily Load Implementation Challenges

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The Forestry Exemption to the North Carolina Sedimentation Pollution Control Act

Sean Brogan, Water Quality and Wetlands Staff Forester
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Topical Area - Long Term Data and Watershed Response to Disturbance

Long Term Hydrologic Trends on the Little River Experimental Watershed

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Quantifying Trail Erosion and Stream Sedimentation with Marked-Recapture Tracers

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Incorporation of Variable-source-area Hydrology in the Phosphorus Index: A Paradigm for Improving Relevancy of Watershed Research

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Chemistry and Discharge Variability in Headwaters of the Sierra Nevada of California

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Long Term Hydrologic Database: Goodwater Creek, Missouri

John Sadler, Research Leader
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Atmospheric/Oceanic Influences on Climate in the Southern Appalachians

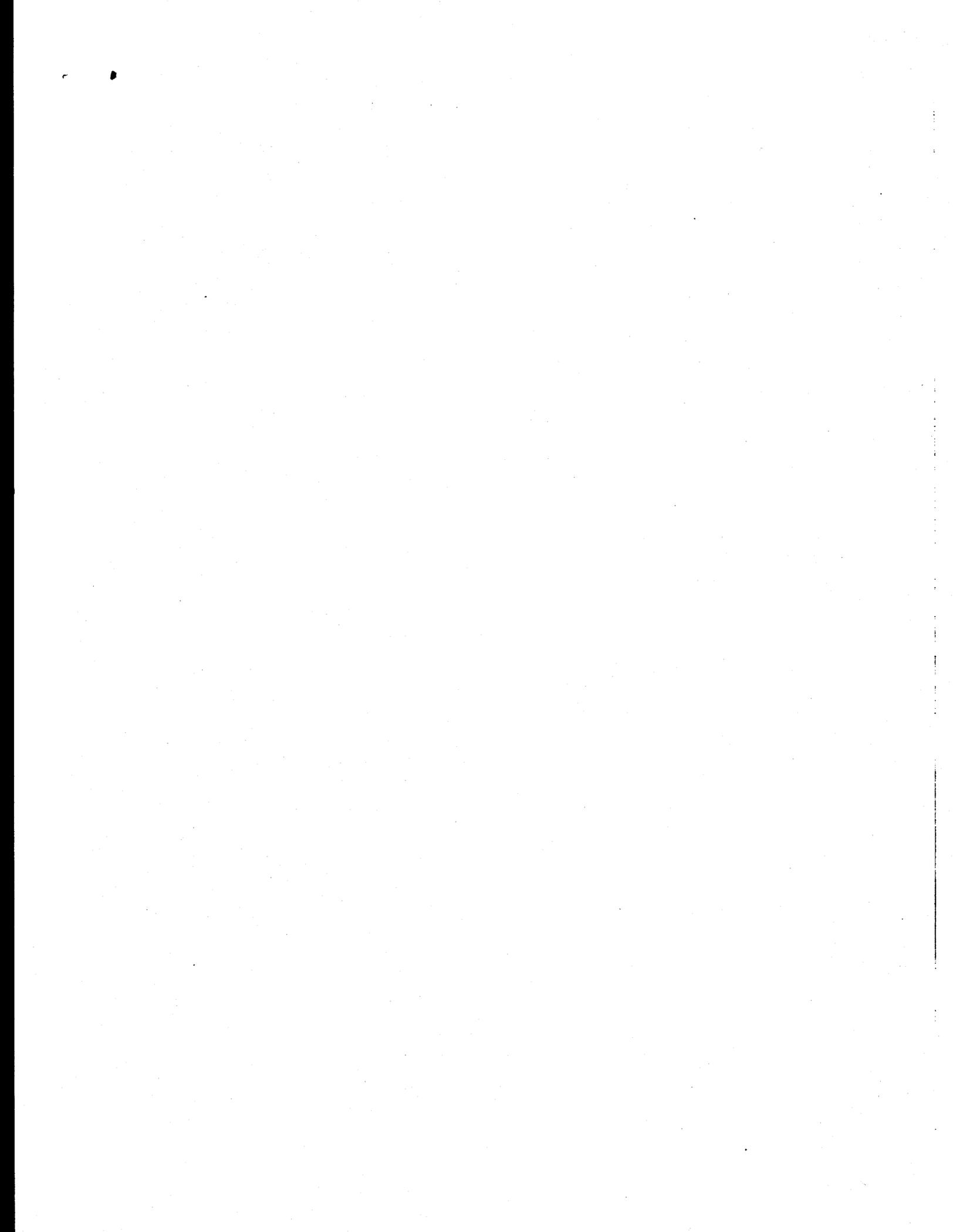
Mark Riedel, Research Hydrologist
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Hurricane Impact on Stream Flow and Nutrient Exports for a First-Order Forested Watershed of the Lower Coastal Plain, South Carolina

Devendra Amatya, Research Hydrologist
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Eastern Hemlock Transpiration: Patterns, Controls, and Implications for Its Decline in Southern Appalachian Forests

Chelcy R. Ford, Postdoctoral Ecologist
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USFS - SRS - Coweeta Hydrologic Laboratory May 16 - 18, 2006

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