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Forest ecohydrological research in the 21st century: what are the critical needs?

James M. Vose,[1]* Ge Sun,[2] Chelcy R. Ford,[1] Michael Bredemeier,[3] Kyoichi Ostsuki,[4]
Adam Wei,[5] Zhiqiang Zhang[6] and Lu Zhang[7]

[1] *USDA Forest Service Southern Research Station, Coweeta Hydrologic Laboratory, 3160 Coweeta Lab Road, Otto, NC 28763, USA*
[2] *USDA Forest Service Southern Research Station, Eastern Forest Environmental Threat Assessment Center, Raleigh, NC 27606, USA*
[3] *Forest Ecosystems Research Center, Universität Göttingen, Buesgenweg 2, Göttingen D-37077, Federal Republic of Germany*
[4] *Department Forest and Forest Products Sciences, Kyushu University, 394, Tsubakuro, Sasaguri, Fukuoka 811-2415, Japan*
[5] *Department of Earth and Environmental Sciences, University of British Columbia, 3333 University Way, Kelowna, BC V1V1V7, Canada*
[6] *College of Soil and Water Conservation, Beijing Forestry University, Qinghua East Road 35#, Haidian District, Beijing 100083, People's Republic of China*
[7] *CSIRO Land and Water, Clunies Ross Street, Canberra, ACT 2601, Australia*

ABSTRACT

Modern ecohydrologic science will be critical for providing the best information to policy makers and society to address water resource challenges in the 21st century. Implicitly, ecohydrology involves understanding both the functional interactions among vegetation, soils, and hydrologic processes at multiple scales and the linkages among upland, riparian, and aquatic components. In this paper, we review historical and contemporary ecohydrologic science, focusing on watershed structure and function and the threats to watershed structure and function. Climate change, land use change, and invasive species are among the most critical contemporary issues that affect water quantity and quality, and a mechanistic understanding of watershed ecosystem structure and function is required to understand their impacts on water quantity and quality. Economic and social values of ecosystem services such as water supply from forested watersheds must be quantified in future research, as land use decisions that impact ecohydrologic function are driven by the interplay among economic, social, political, and biological constraints. Future forest ecohydrological research should focus on: (1) understanding watershed responses to climate change and variability, (2) understanding watershed responses to losses of native species or additions of non-native species, (3) developing integrated models that capitalize on long-term data, (4) linking ecohydrologic processes across scales, and (5) managing forested watersheds to adapt to climate change. We stress that this new ecohydrology research must also be integrated with socio-economic disciplines. Published in 2011. This article is a US Government work and is in the public domain in the USA.

KEY WORDS contemporary issues; climate change; critical hydrologic functions; forest ecohydrology; research needs

INTRODUCTION

Forest hydrological science has traditionally focused on the effects of forest management on the hydrological cycle at a small watershed scale (Ice and Stednick, 2004). In most cases, the initial research conducted in the early and mid 1900's began as 'classic' paired watershed studies (Bosch and Hewlett, 1982) where streamflow amount and timing from treatment watersheds (typically involving treatments manipulating of vegetation) were compared to that of reference watersheds. These studies resulted in powerful empirical tools (i.e. regression models, numerical models, graphical analyses, etc.) that could be used to predict the impacts of forest vegetation changes on water yield and water quality. The value of these studies cannot be overstated (Post and Jones, 2001) and they represent among the earliest endeavors to link physical science (i.e. hydrology) to vegetation dynamics (i.e. forest ecology) (Rodriguez-Iturbe, 2000). However, the watershed ecosystem was often treated as a black box with little attention paid to the structural components and biological processes that regulate hydrologic and biological responses within the watershed. The emergence of the concept of ecosystem science in the 1950's and 1960's led to increased interest in water quality and the biogeochemical cycling processes that determine how ecosystems cycle carbon and nutrients, and ultimately influence water quality. Small watersheds provided convenient study units for defining ecosystems and testing ecosystem concepts developed by Odum and others in the 1950's and 1960's (Odum, 1959; Bormann and Likens, 1967; Odum, 1969). The ecosystem concept recognized that water, nutrient, and carbon cycles were tightly linked and interdisciplinary approaches that examined the roles of soil, vegetation, and associated biota, as well as the atmospheric environment were needed to understand these linkages. The complexity of the issue required nontraditional research approaches; and indeed, some of the earliest and best examples of ecohydrological research

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*Correspondence to: James M. Vose, USDA Forest Service Southern Research Station, Coweeta Hydrologic Laboratory, 3160 Coweeta Lab Road, Otto, NC 28763, USA. E-mail: jvose@fs.fed.us

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come from these early watershed-based ecosystem studies. Implicitly, ecohydrology involves understanding both the functional interactions among vegetation, soils, and hydrologic processes at multiple scales, and of the linkages among upland, riparian, and aquatic components (Figure 1).

Renewed interest in understanding forest ecohydrological processes of forest ecosystems and the role of forests in affecting water supply and ecosystem services has been triggered by recent worldwide water crises and the ongoing climate change debate. The solution to water resource issues will not be a general one, as different parts of the world are facing different emerging water resource issues (Table I). For example, until recently, water has been plentiful in the humid temperate and subtropical regions of the world, and thus social, economic, and population growth have not been limited by potential water supply. Instead, development activities have focused more on excess water removal and management (e.g. flood control and agricultural drainage) and watercourse alteration to achieve development and transportation goals. In contrast, in many other parts of the world, scarce and variable supplies of freshwater have severely limited social and economic development (Vörösmarty *et al*., 2000; Brown and Lall, 2006; Falkenmark *et al*., 2007). Future projections of the world's supply of freshwater suggest substantial and growing areas where water limitations will be extreme (Jackson *et al*., 2001; Oki and Kanae, 2006) impacting both humans and aquatic ecosystems (Vörösmarty *et al*., 2010).

The science of forest ecohydrology will be critical for providing the best information to policy makers and society to address water resource challenges in the 21st century. Decades of research on hydrologic function in managed and unmanaged forest watersheds has provided a solid foundation for understanding how watersheds respond to observed natural disturbances and management activities (National Research Council of the National Academies, 2008). However, our current observed empirical, conceptual, and predictive understanding may not be sufficient to address contemporary and future issues (Wagener *et al*. 2010). Contemporary and future issues that affect water quantity and quality are primarily a result of the intensification of human activities across the globe (e.g. climate change, land use change, and invasive species) that have created conditions that are outside the range of many of our historical observations and understanding derived from those observations A complete and predictive understanding of watershed structure function required to manage and maintain healthy watersheds in the 21st century necessitates a coordinated interdisciplinary approach that combines expertise from socio-economic science, soil science, forestry, ecology, hydrology, biology, and climatology (Figure 2) (de la Cretaz and Barten, 2009). Most importantly, the *impact of humans* on forest hydrologic systems coupled with the *reliance of humans* on the water-based ecosystem services provided by forests requires forest ecohydrologists to work at large spatial scales and understand the linkages among forest and human-dominated landscape components.

Our objectives in this paper are to describe our current understanding of: (1) the driving forces that regulate the quality and quantity of water from forested watersheds, (2) the critical ecohydrological functions of forested watersheds at risk and their role in providing water-based ecosystem services, and (3) future research needs to address 21st century water concerns.

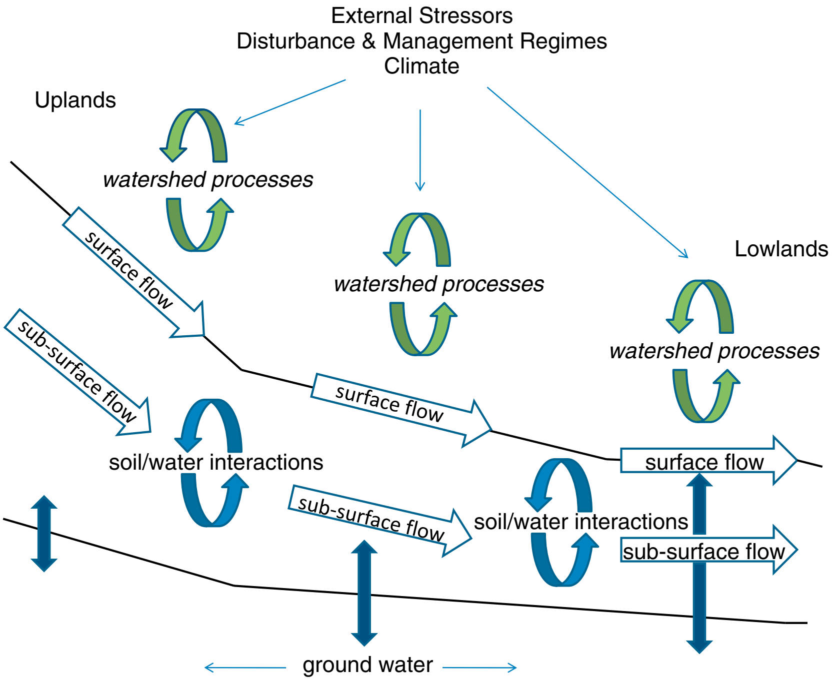

Figure 1. New approaches to ecohydrology will require linking soils, vegetation, and climate at landscape scales thorough surface and subsurface flow paths in the context of external stessors, disturbances,and management regimes.

Table I. Examples of forest water resource issues from countries across the globe.

Country	Forest water issues	References
United States	• Impacts of climate change in snow-dominated ecosystems in western United States	Brown, 2000
	• Impacts of wildfires on soil erosion and sediment loading	Ice and Stednik, 2004
	• Ecohydrological consequences of bioenergy production and intensive forest management	Sun *et al*., 2008
	• Increase demand for freshwater due to population growth	Hooper *et al*., 2005
	• Impacts of invasive species	Jackson *et al*., 2001
	• Impacts of climate change on water resources, including changes in the frequency and severity of extreme events (drought and flooding)	
	• Developing forest management strategies to mitigate or adapt to climate change	
Canada	• Interaction of climate change and forest management	Kurz *et al*., 2008
	• Impacts of natural disturbances on watershed processes	Buttle and Metcalfe, 2000
	• Salvage logging effects on water quality and quantity	Schindler, 2001
	• Impacts of climate change and land use change on aquatic habitat	
China	• Groundwater withdrawal and water shortage in northern China	Xia *et al*., 2007
	• Reforestation/Afforestation impacts on water yield in northern China	Sun *et al*., 2006a,b
	• Water pollution from small industries and increasing urbanization	Wei *et al*., 2008 McVicar *et al*., 2007
Japan	• Managing mature plantation forests	Onda *et al*. 2010
Europe	• Optimizing forest management for water quality and flood control in wet regions	
	• Optimizing forest and water tradeoffs in dry regions	
Australia	• Impact of brushfires on water quality and quantity	Lane *et al*., 2006
	• Afforestation impacts on water yield and salinity	Zhang *et al*., 2007
	• Forest management and groundwater use	Silberstein *et al*., 2007 Benyon *et al*., 2008

THREATS TO FOREST WATER RESOURCES

Population growth and increasing demand for fresh water

A major driving force threatening the supply of clean water is increased demand from a growing human population (Cech, 2005). Rapid population growth and associated environmental degradation and water pollution (Vörösmarty *et al*., 2000, 2010) in the past decades have caused serious water stresses around the world, even in the historically 'water-rich' regions of the globe (Jackson *et al*., 2001). The United Nations projects that the world population will increase from 6 billion in 1999 to as high as 9·2 billion, resulting in a doubling of water demand (Cech, 2005). Higher water demand for direct human consumption and agricultural and industrial uses are expected to increase vulnerability of ecological and social systems to severe water scarcity in many parts of the world (Oki and Kanae, 2006).

Land use change

Land use activities are likely to pose among the greatest threats to water resources in the 21st century (Scanlon *et al*., 2007). Nearly, a century of research in watersheds across the world has shown that forested watersheds, whether managed or unmanaged, provide the cleanest and most stable supplies of water of all land uses (Ice and Stednick, 2004). Land use change is driven by increasing demands for food and fiber to support human population growth (Foley *et al*., 2005), and perhaps to provide non-fossil fuel-based energy sources in the future (Jackson *et al*., 2005). Forest to urban land use conversion, including new roads and commercial development required to support expanding populations, will potentially degrade water quality and reduce reliable supplies of surface water. As forests shrink and become more fragmented, their ability to produce high quality water, moderate and dampen the effects of extreme rainfall events, and sustain healthy aquatic ecosystems will also diminish.

In contrast, in many parts of the globe, reforestation and afforestation are occurring at a rapid pace. For example, approximately 62 million ha have been planted in China over the past 60 years (The State Forestry Administration, 2009) and an additional 40 million ha are expected to be planted and regenerated by 2020 (Sun *et al*., 2006a,b; Yin *et al*., 2010). In addition, emerging biofuels markets and carbon sequestration policies have the potential to increase the extent and intensity of forest plantations (Jackson *et al*., 2005). Recent reviews of the potential impacts of this expansion of intensively managed forest plantations could have negative impacts on runoff. For example, Farley *et al*. (2005) reported that afforestation of grasslands resulted in a 75% reduction in streamflow when planted with eucalypts and a 40% decrease when planted with pines. The magnitude of the impacts varies depending on a variety of factors. For example, streamflow in drier regions may be more vulnerable than that in wetter regions (Farley *et al*., 2005; Sun *et al*., 2006a,b).

Climate change

Although population growth and land use change are the most obvious sources of stress on the world's water

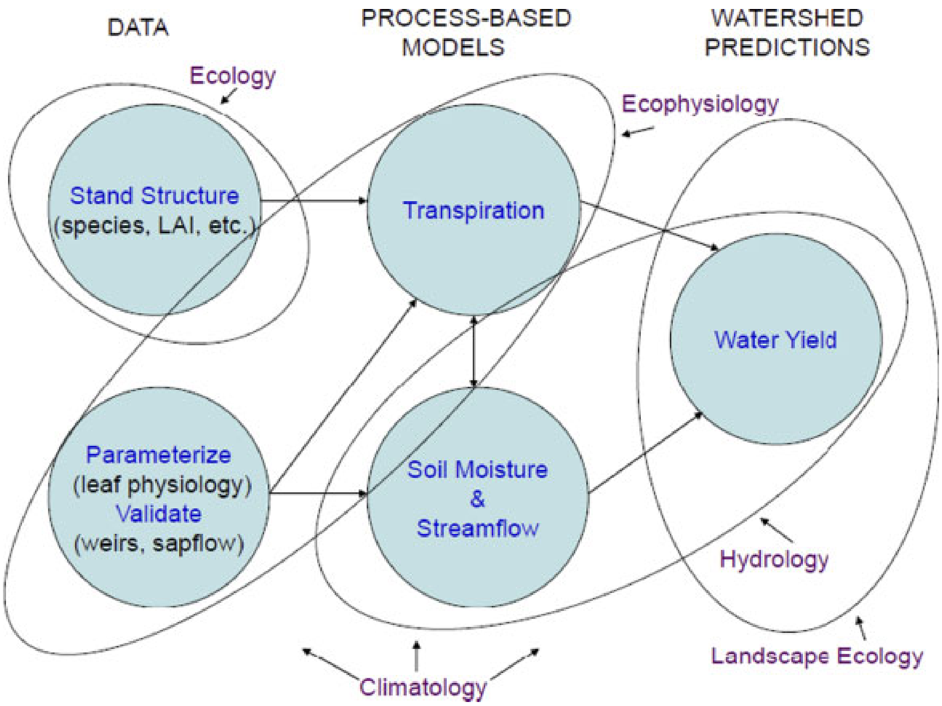

Figure 2. Traditional hydrology focused principally on small experimental watersheds to characterize water yield in disturbed and reference forests. New approaches require interdisciplinary research that combines and integrates multiple disciplines (e.g. socio-economic sciences, soils, forestry, ecology, hydrology, biology, and climatology) and spatial and temporal scales to predict water yield responses to factors such as climate change, invasive species, and land use change.

resources, other factors, including climate change and increasing climatic variability will also contribute (Gleick, 2003). Some models project more frequent El Niño-like conditions (Thompson *et al*., 2003), with increased extreme rainfall event frequency. Climate change and increased climate variability will both directly and indirectly influence water quantity and quality. The direct effect of climate change is via changes in rainfall and temperature, whereas the indirect effect is associated with changes in plant physiology. For example, increased temperature could decrease streamflow by increasing evapotranspiration (ET), although this may be buffered by increased annual precipitation in some cases (Oki and Kanae, 2006; Sun *et al*., 2008). The physiological effect of elevated atmospheric CO_2 has been hypothesized to be the cause of increased runoff for some large river basins around the world (Gedney *et al*., 2006). However, most studies of climate change impacts on water resources have only considered the direct effect, mainly because the interactions and feedbacks between the atmosphere and vegetation under increased CO_2 are not well understood.

Climate change in many parts of the world will likely lower streamflow, which in turn will decrease water supply, degrade aquatic communities, and diminish water quality. Extreme rainfall events will likely increase flood severity and frequency, negatively affecting human safety, human welfare, and aquatic community functioning. Changes in hydroperiod (Ford and Brooks, 2002; Ernst and Brooks, 2003) and sea level rise (Ross *et al*., 1994) will have significant direct impacts on hydrological processes in forested wetlands (Amatya *et al*., 2006). Most importantly, climate change and variability have the potential to interact with land use change and alter disturbance regimes to exacerbate the direct impacts on water quality and quantity (Wilcox, 2010). For example, warming and drought in North America has resulted in widespread mountain pine beetle infestations throughout the western United States and Canada (Kurz *et al*., 2008), and warming has modified snow regimes (Barnett *et al*., 2008) and has increased the frequency and severity of large wildfires (Westerling *et al*., 2006). The effects of these types disturbances on water quantity and quality under historic climatic conditions and disturbance regimes are well understood (National Research Council of the National Academies, 2008); however, it is less certain whether similar responses (and management actions to mitigate responses) can be expected under climate change and new disturbance regimes. Accordingly, understanding and managing the ecohydrological structural and functional attributes that characterize forested watersheds (e.g. flood control, nutrient and carbon sequestration, sediment trapping, and salinity control) will become more important for protecting water resources in the face of climate change and climate variability.

Invasive species

A fourth major stressor to forested watersheds in many parts of the world is the invasion of non-native species including disease organisms, insects, and a host of other invertebrate and vertebrate pests. It is likely that the frequency and impact of non-natives will increase significantly over the next several decades (Ellison *et al*., 2005; Lodge *et al*., 2006). Non-native species have the potential to significantly alter ecohydrological function (Gordon, 1998) and reduce the capacity of forested watersheds to provide clean and abundant water, as well as support diverse, healthy aquatic ecosystems, and native faunas. Although some information exists on the direct impacts of aquatic non-native species on aquatic ecosystem structure and function, we know considerably less about the impacts of non-native species on water quantity and quality (Hooper *et al.*, 2005). For example, invasion of non-native species into riparian areas of the western United States significantly alters groundwater depth and ecophysiological function of native species (Pataki *et al.*, 2005; Scott *et al.* 2006).

Fire

In Australia, bushfires are considered threats to water resources. Impact of bushfires on streamflow or water yield is variable depending on degree of disturbance, ecological response to disturbance, and time since disturbance. Early post-fire responses include significant increases in annual flow (Lane *et al.*, 2010), peak flows (Mackay and Cornish, 1982), and changes in baseflow (O'Loughlin *et al.*, 1982). These responses may persist several years after fires. Langford (1976) and Kuczera (1987) investigated long-term water yield responses from *Eucalyptus regnans* forests in Victoria, Australia and showed that water yield gradually declined to a minimum after about 27 years as stands regenerate following fire and increased again as the stands age. Models predicting changes in water yield with forest age for mountain ash, mixed species, and snow gum forest types estimate that the difference in water yield between fire and no-fire scenarios is +13% of the mean annual flow for the River Murray (Watson *et al.*, 1999; Murray-Darling Basin Commission, 2007).

ECOHYDROLOGIC FUNCTIONS AT RISK

Soil protection and sediment trapping and filtering

Healthy forested watersheds efficiently trap and filter suspended sediments. While exposed soil surfaces on the forest floor are susceptible to splash displacement, surface runoff, and erosion (Nanko *et al.*, 2006, 2008, 2010), the forest understory and litter layer protects soils from rainsplash erosion. The forest litter layer also is highly porous and rainfall intensity rarely exceeds infiltration rates in forested watersheds. As a result, except in areas where the forest floor has been removed by fire or other disturbances or soils have been severely compacted, both overland flow and soil erosion have very low observed rates in forested watersheds (Ice and Stednick, 2004). Where soil erosion rates are higher, such as from recently constructed roads or skid trails, the forest litter layer and other forest floor components such as coarse wood and understory vegetation play important roles in filtering and trapping suspended sediments (Swift and Burns, 1999; Ide *et al.*, 2009, Fukuyama *et al.*, 2010). Thus, management practices that ensure that areas (such as riparian buffers) that receive suspended sediment are of sufficient size and capacity to effectively trap, filter, and retain sediment are key in influencing overland flow, soil erosion, and sediment loading in streams. Long-term forest watershed research has demonstrated that land uses or disturbances (such as extreme wildfire) that expose bare soil, alter infiltration capacity, or reduce the functional capacity to filter and trap suspended sediments will result in reduced water quality. Fortunately, long-term forest watershed research also shows that many of these impacts are often rapidly reversible (or greatly reduced) with reforestation and afforestation (Ice and Stednick, 2004).

Biogeochemical cycling and nutrient sequestration

Forested watersheds cycle and sequester elements in a complex process that begins with the contact of atmospheric constituents with the forest canopy and ends with in-stream biological and physical processes (Figure 3). Biogeochemical cycles involve the interplay among autotrophic and heterotrophic organisms, organic matter, mineral soil, and underlying geomorphic deposits. In most cases, forested watersheds are highly conservative and have the capacity to sequester large amounts of externally (e.g. atmospheric deposition) and internally generated (e.g. weathering and recycling) chemical compounds (Chiwa *et al.*, 2010). The ability to accumulate chemicals in above- and below-ground wood (both live and dead) (Spears and Lajtha, 2004) provides a distinct advantage of forest ecosystems compared to other land uses. Forests accumulate and cycle not only chemicals required for growth but also non-essential chemicals that may be toxic or harmful (e.g. heavy metals) to aquatic and human health. Because the rate of chemical accumulation is tied to the rate of wood accumulation, the greatest accumulation rates occur in rapidly growing, healthy, forested watersheds. Physical and biological processes in forest soils also play an important role in processing and sequestering chemicals. Understanding how those processes interact with the hydro-geomorphic setting is fundamental to designing functional riparian management zones to protect water quality (Price and Leigh, 2006). For example, nitrate-nitrogen in subsurface runoff can be ameliorated in a forested riparian buffer if the flow-path is directed through the biologically active zone (Chescheir *et al.*, 2004). This knowledge also provides the basis for engineering natural systems for treating wastewater or runoff. In general, forestry-based land uses that are implemented with BMPs have the greatest capacity for sequestering nutrients and chemicals. However, watershed research also shows that this capacity is not unlimited and can be significantly impacted by chronic or acute disturbances that alter ecosystem structure and function.

Regulating streamflow

Forested watersheds play critical roles in regulating streamflow, although their capacity to mitigate extreme precipitation events and reduce flooding is limited (Burt and Swank, 2002; Eisenbies *et al.*, 2007). Streamflow is comprised of baseflow (i.e. water released gradually from

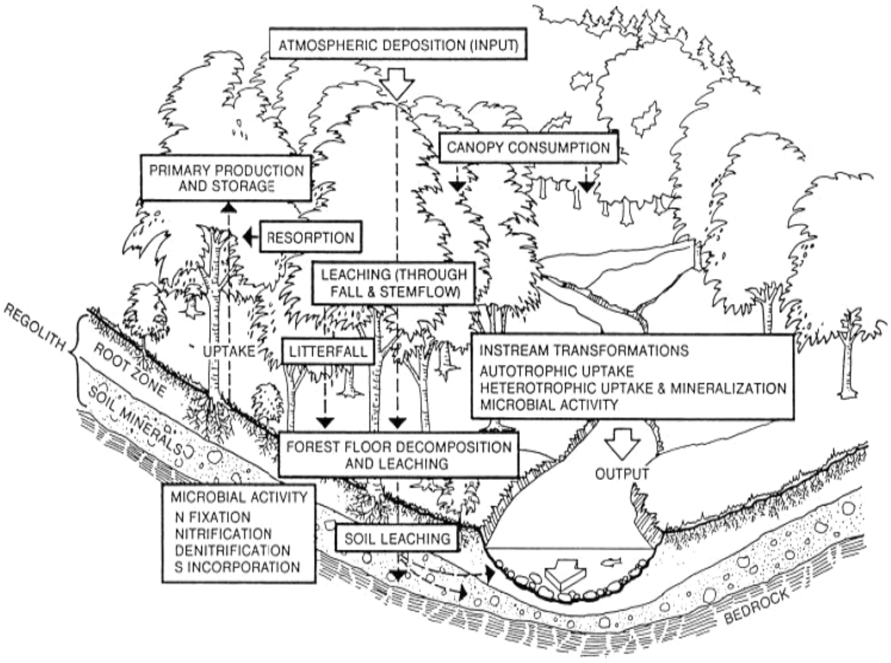

Figure 3. Understanding biogeochemical cycling in forest ecosystems provides one of the best examples of the complexity of watershed processes. In addition to quantifying inputs and outputs, all of the terrestrial and aquatic processes that regulate accumulation and transformations need to be quantified.

groundwater and soil water storage) and stormflow (i.e. baseflow + flow generated during storm events) and is the integrated product of climate, geology, vegetation, and soils. In many non-forest land uses, most stormflow (or quickflow) results from increased surface runoff (i.e. overland flow) (Price and Leigh, 2006). In unsaturated soils, infiltration rates associated with forests typically exceed rainfall intensity and overland flow are rare, except on areas where disturbance (such as roads and skid trails) removes or disrupts the forest floor or compacts the soil. Instead, most stormflow in forested watersheds is generated from subsurface flow processes (Hewlett *et al*., 1984; Sun *et al*., 2006a,b) and direct channel inputs. Subsurface flow is considerably slower than overland flow and helps buffer the 'flashiness' of streamflow response to large storms in forested watersheds. Overland flow can also occur when soils are saturated or in areas where soil strata limit infiltration, resulting in perched water tables. Such perched water tables are common in lowland forest watersheds in coastal areas. The frequency, timing, and magnitude of saturated conditions are influenced by forest evapotranspiration, which removes water from the soil and increases the soil storage capacity. For example, the highest stormflows are most likely to occur in the dormant season when evapotranspiration rates are lowest.

Across the globe, forest evapotranspiration rates range from 400 to 3000 mm per year (Table II). This extremely wide variation in ET reflects differences in climatic, ecosystem types and species composition, land use, and management regimes that affect canopy interception and plant transpiration (Sun *et al*., 2010). Forest evapotranspiration reduces the amount of baseflow relative to land uses with lower amounts of evapotranspiration (e.g. agriculture, pasture, and developed land). Forest watershed research shows that forest land uses play an important role in regulating streamflow responses to extreme rainfall events; i.e. flooding. However, the capacity to buffer extreme events is not unlimited. Very large storms can exceed the storage capacity of even the healthiest forested watersheds; nevertheless, forest land affords greater protection than most if not all other alternative land uses (Hewlett and Helvy, 1970; Bolstad and Swank, 1997; Price and Leigh, 2006).

Forest management activities that change structure, species composition, or forest age can also influence streamflow regulation. The most dramatic effects have been shown when forest management changes functional groups; for example, from broadleaf to coniferous species. Komatsu (2005) presented evaporation rates from 67 forest sites across the world using the Priestley–Taylor α coefficient for daytime (α represents the ratio between forest actual ET and ET from a free water surface; i.e. equilibrium ET) and found that there are (1) clear differences in α values of Priestley–Taylor parameter between broad-leaved and coniferous forests, (2) a greater variation in α values among individual coniferous forests than among individual broad-leaved forests, and (3) a clear relationship between canopy height and α values for coniferous forests. Based on the interception data from 16 forest sites, a relationship between stem density and interception ratio was found (Komatsu *et al*., 2007). Thus, converting forest type or controlling the canopy height and stem density of coniferous forest could be the tools for regulating streamflow. Converting temperate coniferous forests into broadleaved forests has also been shown to increase water yield because winter interception loss is larger for evergreen compared to broadleaf deciduous forest (Swank

Table II. Examples of evapotranspiration (ET) and precipitation (P) rates from forest ecosystems around the globe.

Biomes	Dominant forest types	ET (mm/year)	P (mm/year)	ET/P	References
Tropical rainforests (Sarawak, Malaysia)	Dipterocarp (*Dipterocarpaceae*) and tropical heath forest	1545	2151	0.72	Kumagai *et al*., 2005
Subtropical forests (Huitong, Hunan, China)	Chinese fir (*Cuninghamia lancelata*)	890	1158	0.77	Wei *et al*., 2005
Temperate forests (Ohio, USA)	Oak (*Quercus* spp.), Maple (*Acer* spp.)	673	802	0.84	Sun *et al*., 2010
Boreal forests (Saskatchewan, Canada)	Jack pine (*Pinus banksiana*)	438	559	0.78	Amiro, 2009
Dry, warm forests (NSW, Australia)	Evergreen broadleaf (*Eucalyptus crebra; Callitris glaucophylla*)	685[a]	529[a]	>1.0	Zeppel *et al*., 2008
Temperate broadleaf forest (Sarukawa, Miyazaki, Japan)	*Castanopsis* spp., *Quercus* spp.	3073	1095	0.36	Komatsu *et al*., 2007
Temperate conifer forest (Sarukawa, Miyazaki, Japan)	Japanese cedar	3000	1154, 1255	0.38,0.42	Komatsu *et al*., 2007
Temperate conifer forest (Tatsunokuchi, Okayama, Japan)	*Pinus thunbergii*	1229	855	0.70	Komatsu *et al*., 2007
Temperate broadleaf forest (Tatsunokuchi, Okayama, Japan)	Mixed broadleaf forest	1142	915	0.80	Komatsu *et al*., 2007
Temperate broadleaf forest (Hitachi-Ohta, Ibaraki, Japan)	Mixed deciduous broadleaf forest	1568	697.4	0.44	Komatsu *et al*., 2007
Temperate conifer forest (Hitachi-Ohta, Ibaraki, Japan)	*Cryptomeria japonica, Chamaecyparis obtusa*	1343	545.8	0.41	Komatsu *et al*., 2007

[a] One year measurement data.

and Douglass, 1974). This management practice could be applied to the uniform precipitation monsoon regions such as northern Japan to increase water yield (Komatsu *et al*., 2008a). However, it could not be applied to summer precipitation monsoon regions such as western Japan because winter precipitation is low and thus winter interception loss is small regardless of forest type (Komatsu *et al*., 2007). Controlling interception loss by stem density management such as thinning could be effective for low precipitation regions; however, the greater the annual precipitation, the lower the interception ratio (Komatsu *et al*., 2008b). Thus, the limitation of the controlling interception loss should be recognized.

FUTURE RESEARCH NEEDS

Demands for water-based ecological services from forest ecosystems will likely dramatically increase in the 21st century. At the same time, forest ecosystems are likely to become increasingly stressed due to climate change, atmospheric deposition, urbanization, and demographic changes. While much is known about watershed structure and function (Swank *et al*., 2001; Ice and Stednick, 2004; Jackson *et al*., 2004) there is still a considerable amount of uncertainty about key aspects of ecohydrological functions. Specifically, how will ecohydrologic functions respond to global environmental changes in the future? Will forested watersheds be able to continue providing ecosystem services required by society? What management options are available to mitigate or adapt to global environmental changes? Forest hydrology research is facing new challenges to answer these increasingly complex questions. To address these challenges, we have identified several specific areas that may guide future research efforts.

Understanding watershed responses to climate change and variability

We need to know and predict how forest watersheds will respond to climate change. For example, how will carbon, nutrient, and water cycling processes change in response to higher temperatures and altered moisture regimes? How will these changes impact water quality and quantity? To answer these questions, we need to develop a predictive understanding (through a combination of monitoring, experiments, and modeling) of processes that regulate biogeochemistry and water cycling in forested watersheds.

There is a long history of research on the impacts of disturbance on water resources using paired catchment approaches. Indeed, small-scale watershed studies have proven extremely valuable for understanding the interactions among vegetation, soil, and climate throughout the world. Despite their value, we contend that empirical relationships derived from paired catchment studies (and models built from them) will have limited utility for predicting responses to future climate because conditions may be very different than observed in the measurement record. In many parts of the world, recent changes in air temperature and increased frequencies of extreme events have produced climatic conditions comparable to what would be expected under climate change (Easterling *et al.*, 2000; Huntington 2006). A first step to understanding watershed responses to climate change is to exploit this recent variation in climate and conduct retrospective analyses of the relationships among long-term streamflow and climate. As a conceptual framework, we can describe the relationship among forests, climate (e.g. precipitation and temperature), and streamflow (Figure 4) response thresholds that result in undesirable impacts on social and/or ecological systems and design mechanistic studies around these thresholds. In this example, precipitation is described by a normal distribution centered on the mean annual amount with a lower frequencies of extreme amounts at the tails (i.e. extreme wet or extreme dry). For most years, interactions between rainfall and vegetation result in streamflow levels within the desirable or acceptable range; however, at the extremes, excess rainfall or low rainfall results in undesirable streamflow levels (e.g. extreme low flows or extreme floods; Figure 4). Applying this framework to long-term climate data at the Coweeta Hydrologic Laboratory in western North Carolina, USA, indicates an increase in extreme precipitation events (Laseter *et al.*, 2010) in the past 25 years. Coupled with long-term streamflow records, we can assess watershed hydrologic sensitivity to climate variability (in this case to increased frequency of extreme precipitation events) and gain insight into to how forested watersheds might respond to increases in the frequency of extreme events. For example, we compared differences (% change relative to average annual flows during non-extreme years) in streamflow during extreme dry years and extreme wet years among three deciduous hardwood reference watersheds that varied in species composition, elevation, slope, and aspect (Table III). Streamflow responses to extreme events were greatest (+56% and −47·9% for wet and dry years, respectively) on the low elevation south facing watershed and lowest (+35·8% and −31·8% for wet and dry years, respectively) on the east facing, high elevation watershed. These differences in response patterns are likely due to a combination biological (e.g. species) and physical (e.g. soil depth, slope) factors and suggest substantial variation in response patterns across forest types and landscape positions.

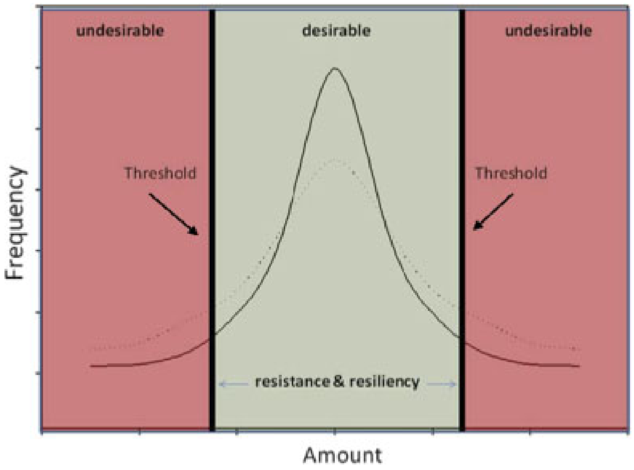

Figure 4. Conceptual model of ecosystem responses to climate change related extreme events. The solid line represents a hypothetical annual precipitation regime where most years are centered around the mean. The dashed line represents a hypothetical increase in extreme events due to climate change; i.e. the frequency of extreme wet and dry years increase. Ecosystem possess and inherent capacity to adapt (by either resisting the disturbance or recovering quickly from the disturbance) to most climate conditions (the green area); however, during extreme precipitation events, the ability of the ecosystem to resist or recover quickly is exceeded and undesirable outcomes occur such as flooding during extreme rainfall, or a lack minimum flow occurs during extreme drought. The response threshold varies by ecosystem type and condition. For example, degraded ecosystems may have a lower response threshold than non-degraded ecosystems.

Understanding watershed responses to losses of native species or additions of non-native species

What are the impacts of losing or adding individual native and non-native species or functional groups on ecohydrologic processes at the watershed scale? To answer this question, not only is a detailed understanding of the impacts of species on hydrological processes needed, but also a robust, scaling approach that accurately translates these responses to the watershed scale is key. The difficulty in measuring or modeling transpiration at the tree scale has posed a significant challenge to ecohydrologists and limited the ability to assess the impacts of changes in species composition on hydrologic processes. Sapflow technology and scaling approaches have greatly improved our ability to understand variation in transpiration rates among species and community types, however.

Large variation in transpirations rates has been observed among species (Figure 5 in Ford *et al.*, 2010) and community types (Table IV). This species and community level understanding is critical for evaluating the implications of changes in species composition due to management, natural succession, invasive species, or disturbance. For example, pine plantations consume nearly twice the water consumed by longleaf pine savannas, but only marginally more than mature upland hardwood forests (Table IV). The potential for large increases in fast growing forest plantations for bioenergy may have significant implications on water resources across the globe (Farley *et al.*, 2005; Jackson *et al.*, 2005)

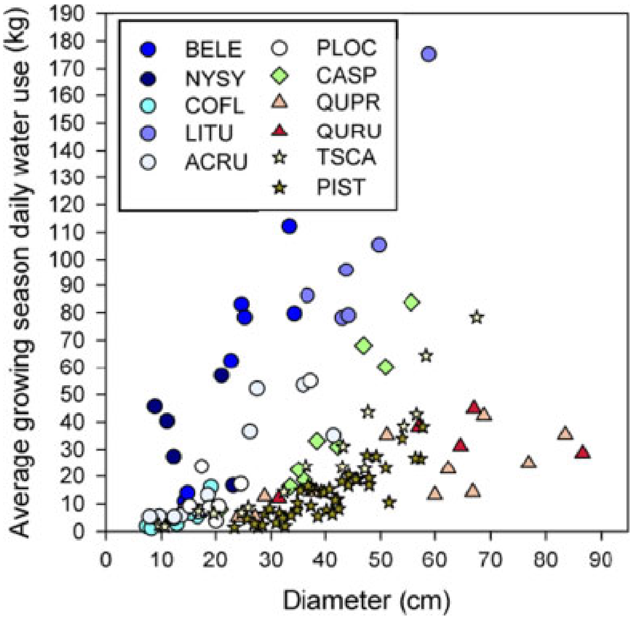

Figure 5. Observed daily water use (DWU) estimated from sap flux density in trees of varying species (legend text denotes first two letters of Latin binomial: BELE *Betula lenta*, NYSY *Nyssa sylvatica*, COFL *Cornus florida*, LITU *Liriodendron tulipifera*, ACRU *Acer rubrum*, PLOC *Platnus occidentalis*, CASP *Carya* spp., QUPR *Quercus prinus*, QURU *Q. rubra*, TSCA *Tsuga canadensis*, PIST *Pinus strobus*) in reference watersheds at Coweeta (except PIST). Symbols represent the mean DWU of replicate trees in each species during the growing season for deciduous species, days of year 128–280 in 2006. Mean DWU during the entire annual period is shown for coniferous species (TSCA is during 2004, PIST is during 2006). LITU, QURU, QUPR, CASP, and PIST data are from Ford *et al*. (2010). TSCA data are from Ford and Vose (2007). BELE, NYSY, COFL, ACRU, and PLOC are from Ford and Vose (unpublished data) but follow the methods in Ford *et al*. (2010). Symbols: circles are species with diffuse porous xylem anatomy, diamonds are species with semi-ring-porous xylem anatomy, triangles are species with ring-porous xylem anatomy, and stars are for species with tracheid xylem anatomy.

Developing integrated models that capitalize on long-term data

Simulation models are commonly used as a tool for synthesis and prediction in forest hydrological research (Sun *et al*., 2006a,b; Lu *et al*., 2009). The complexity of ecohydrologic processes, the large number of factors that drive these processes, and multiple scales necessitate a ecosystem-based modeling approach (Hanson *et al*., 2004; Sun *et al*., 2008; Tian *et al*., 2010). However, model development and applications are often challenged by our understanding of the processes and data availability. Both simple and complex models are often guilty of ‘getting the right answer for the wrong reasons’ due to limitations in simulating internal state variables (e.g. soil moisture and groundwater table depth) or simulating processes that are not easily measured or understood at large spatial scales (e.g. evapotranspiration and internal drainage) (Sun *et al*., 2008). Indeed, the reliability of predictions from multi-scale models depends upon process-level hydrologic research. Experimental watersheds in many countries have accumulated valuable hydrologic and ecosystem process data, that when combined, provide the data required for developing models that can predict the impacts of changes in climate and land use, and other disturbances on water quality and quantity. Multi-tiered watershed data also provide the basis for validating with-in basin hydrologic conditions and the interaction between water and other biogeochemical processes such as carbon cycles. Remote sensing and GIS technology, when integrated with simulation models, can be a powerful tool to quantify large-scale ecosystem processes including ET (Mu *et al*., 2007), carbon balances (Zhao *et al*., 2005; Xiao *et al*., 2008), and water use efficiency studies (Tian *et al*., 2010). Thanks to the advances of remote sensing technology and large number of *in situ* environmental monitoring networks such as the FLUXNET, geospatial data are increasingly available for ecohydrological model testing and validation at a regional scale.

Linking ecohydrologic processes across scales

Future environmental changes and forest management options affect the ecohydrological processes at multiple scales, from changes in tree ecophysiology (Ford *et al*., 2010) to water flow pattern of large river basins (Wei and Zhang, 2010) and to regional climate (Liu *et al*., 2008). Understanding how the multi-scale changes in ecohydrology impact water resources requires expanding spatial scales. The impacts of forest management and land use changes at large spatial scales are cumulative over space and time. Although studies examining the impacts of forest practices (i.e. cutting, roads, drainage, etc.) on water will continue to have value, the specific effects will need to be considered cumulatively, in the context of large spatial scales and a rapidly urbanizing landscape. To address these needs, hydrologic research quantifying cumulative effects of multiple land uses and their disturbances, landscape design, and watershed restoration will need to be accelerated. However, hydrologic connectivity of landscape components requires that we understand how headwater activities influence downstream ecosystems and their services, and carefully consider the benefits of restoring connectivity versus maintaining highly developed disconnected landscapes to prevent the movement of exotic species, high sediment, nutrients, and toxins (Jackson and Pringle, 2010) Conducting hydrologic research at different spatial scales will allow better understanding of (1) watershed ecosystem behavior across scales and (2) the ability to generalize responses patterns across scales. In addition, there is a great need to answer fundamental questions to better understand how, where, and at what rate population pressures and consequent land use change and development will impact water supplies and quality at a scale beyond the scale at which the experimental data are collected (Sun *et al*., 2008). This necessitates an interdisciplinary approach that integrates both human and ecological systems along a gradient of forest to human-dominated landscape components.

Traditional forest hydrological research has been conducted using a paired watershed approach at the small watershed scale (<100 km^2), a scale that vegetation can

Table III. Streamflow response to extreme annual precipitation at Coweeta Hydrologic Lab.

Watershed	Description		Mean annual streamflow (cm/year)	Streamflow deviation in extreme wet years (%)	Streamflow deviation in extreme dry years (%)
2	Aspect	SSE	79.9	56.5	−47.9
	Elevation (max)	1004 m			
	Species[a]	Oak			
18	Aspect	NW	100.5	46.4	−46.1
	Elevation (max)	993 m			
	Species[a]	Oak, Cove hardwood			
36	Aspect	ESE	166.7	35.8	−31.8
	Elevation (max)	1542 m			
	Species[a]	Cove hardwood, Northern hardwood			

[a] Day *et al*. 1988.

Table IV. Sapflow-based estimates of transpiration among community types in the southern United States.

Vegetation type	Mean annual transpiration (mm/year)	Reference
Longleaf pine savanna	244	Ford *et al*., 2008
Old field	250	Stoy *et al*., 2006
Oak-pine-hickory forest	278	Oren and Pataki, 2001
Upland oak forest	313	Wullschleger *et al*., 2001
Mixed pine hardwood	355	Phillips and Oren, 2001
Mixed pine hardwood	442	Stoy *et al*., 2006
Planted loblolly pine	490	Stoy *et al*., 2006
Mixed pine hardwood	523	Schafer *et al*., 2002
Slash pine flatwoods	563	Powell *et al*., 2005

be easily manipulated and climatic influences can be singled out. However, it is difficult to apply traditional small paired watershed approaches to larger landscapes or watersheds. Thus, innovative methods are needed to quantify cumulative effects of land use change, forest disturbance, and climate change on watershed processes at large spatial and temporal scales (Zhang *et al*., 2008; Wei and Zhang, 2010). When the scales of interest become larger, the dominant factors that control water cycles change. For example, at the small watershed scale, topography and soil depth may be important for streamflow generation (Hewlett and Hibbert, 1966); however, at the regional scale climate is likely to mask the influences of topography and soils in water balances. Humans have altered the landscape and water resource allocation globally. The great challenge for large-scale watershed studies is that various variables (e.g. forest disturbance and climatic variability) interactively affect watershed processes and there relative effects must be understood to understand the role of individual variables. There is a wealth of historical monitoring river basin monitoring and accumulated streamflow data around the world that remain to be explored to determine climate–landuse–hydology relations at a large scale.

Managing forest watersheds to adapt to climate change

Forests are unique among other land uses because they are long-lived and relatively stable, yet their structure and function can be altered by management and/or natural disturbances. These structural and functional changes can be either transient or long term, depending on the intensity of the management action or disturbance. Understanding how climate change will impact forested watersheds, and developing management strategies to mitigate or offset those impacts, is critical to maintaining water supplies for human uses and aquatic species and habitats. Although it is well recognized that increasing water yield through forest management would be insufficient to meet future water needs (National Research Council of the National Academies 2008), forest management has the potential to alter the hydrological responses to climate change by influencing biological factors that determine ET. Much of the ecophysiological information used to understand controls on hydrologic processes under current climate conditions will be useful for decision making about management activities in anticipation of future climates. For example, management activities that favor or replace one species (or several species) over another can alter ET through changes in transpiration (E_t) or interception (E_i), and alter sensitivity to climatic variation because (1) tree species vary considerably in transpiration per unit leaf area, and overall whole-tree water use due to differences in rooting depth, tree height, leaf boundary layer resistance, leaf chemistry, and stomatal sensitivity to vapor pressure and (2) species can vary in sensitivity to year-to-year climatic variation (Stoy *et al*., 2006; Ford *et al*. 2010). In addition, stand density can be managed to influence the amount of water evaporated from canopy and soil surface through changes in live and dead leaf, branch, stem area, and litter coverage. In the face of climate change, natural resource managers may be facing a new paradigm where water is the primary ecosystem service derived from forested landscapes and management decisions will be based on established (or new) management regimes and best management practices that optimize water resources.

CONCLUSIONS

Abundant and clean water is fundamental to the viability of aquatic ecosystems, human welfare, and economic growth and development throughout the world (Cech, 2005). The combination of increased demand for freshwater, changes in land use and cover, and climate change will place even greater demands on forest watersheds across the globe to meet the water resource needs of humans and aquatic ecosystems (Sun *et al*., 2008; Vörösmarty *et al*., 2010). The forest hydrology community has a long history of studying the climate–vegetation–hydrology and has helped define the basic ecohydrology of forested watersheds. The science of ecohydrology will be at the forefront of many emerging issues and be relied upon to provide policy and decision makers with the information required to ensure that water and other natural resources are protected or enhanced. Policymakers, natural resource managers, and researchers must start now to develop collaborative, science-based strategies to protect water resources in the face of these co-occurring threats (Lodge *et al*., 2006). We have identified several new or high priority research areas needed to fully develop our understanding of the effects of the accelerated pace of land use change, climate change, and invasive species expansion on water resources. We stress that this new ecohydrology research must also be integrated with socio-economic disciplines. Economic and social values of ecosystem services such as water supply from forested watersheds must be quantified in future research. We are living in an increasingly human-dominated landscape in most parts of the world and land use decisions that impact ecohydrologic function are driven by the interplay among economic, social, political, and biological constraints.

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