Chapter 14
Early Successional Forest Habitats and Water Resources

James M. Vose and Chelcy R. Ford

Abstract Tree harvests that create early successional habitats have direct and indirect impacts on water resources in forests of the Central Hardwood Region. Streamflow increases substantially immediately after timber harvest, but increases decline as leaf area recovers and biomass aggrades. Post-harvest increases in stormflow of 10–20%, generally do not contribute to downstream flooding. Sediment from roads and skid trails can compromise water quality after cutting. With implementation of Best Management Practices (BMPs), timber harvests are unlikely to have detrimental impacts on water resources, but forest conversion from hardwood to pines, or poorly designed road networks may have long lasting impacts. Changing climate suggests the need for close monitoring of BMP effectiveness and the development of new BMPs applicable to more extreme climatic conditions.

14.1 Introduction

Watershed management requires understanding the tight linkages among vegetation, soils, and water quantity and quality. Because of these linkages, forest management activities that alter vegetation, such as creation of early successional habitats, have the potential to impact water resources. From a hydrologic standpoint, we define early successional habitats by the structural and functional attributes that are created by disturbance and influence hydrologic processes. Early successional habitats can be created by either natural disturbances (e.g., hurricanes, tornados, severe wildfires), or human-mediated intentional (e.g., forest cutting)

J.M. Vose (✉) • C.R. Ford
USDA Forest Service, Southern Research Station, Coweeta Hydrologic Laboratory, Otto, NC 28763, USA
e-mail: jvose@fs.fed.us; crford@fs.fed.us

C.H. Greenberg et al. (eds.), *Sustaining Young Forest Communities*, Managing Forest Ecosystems 21, DOI 10.1007/978-94-007-1620-9_14,

and unintentional (e.g., invasive insects and disease introductions) disturbances (White et al., Chap. 3). Defining structural attributes of early successional forests include low leaf, stemwood and sapwood areas, high forest floor mass and coarse woody debris, and a high proportion of fast-growing, shade intolerant species (Keyser, Chap. 15). Defining functional attributes include high leaf-level C gain and low water use efficiencies, rapid organic matter decomposition, and accelerated nutrient cycling and accumulation (Keyser, Chap. 15). Although early successional forest attributes can be maintained with repeated disturbances, these attributes more often are transitional and recovery to pre-disturbance conditions occurs quickly (e.g., leaf area) or over several decades (e.g., species composition). Where disturbances are particularly severe, such as road building or loss of a dominant overstory species, structural and functional attributes may never recover to pre-disturbance conditions (Ellison et al. 2005). Combined, these changes in structural and functional attributes can impact water resources, and land managers need to consider those impacts when managing forests for multiple benefits. In particular, forest harvesting (with and without species conversion) and associated forest operations have the potential to substantially alter both water quantity and quality; in some cases, these changes persist long-term. In short, good land management is good watershed management.

Our understanding of the changes in water resources associated with creating early successional habitats is largely derived from a long history of paired watershed studies that have examined long-term streamflow and water quality responses to forest cutting (Calder 1993; Stednick 1996; Jones and Post 2004; Brown et al. 2005). Paired catchment studies have been critical to understanding how land management and other disturbances affect streamflow and quality. Accurate measurement of streamflow is at the core of paired watershed studies and this typically requires installation of a weir at the watershed outlet (Reinhart and Pierce 1964). Streamwater quality can be measured directly for some parameters (e.g., turbidity, pH, temperature, conductivity) using automated sensors, or water samples can be analyzed in a laboratory for these and other parameters such as nutrient concentration. The primary goal of the paired catchment method is to isolate streamwater response to cutting by accounting for the influences of climate or other factors. Using a paired untreated watershed that serves as a reference, streamflow response to cutting can be determined by examining the difference between expected streamflow (e.g., what would be expected if the watershed had not been treated) from observed streamflow. When measured streamflow differs from expected, the inference is that the treatment alone resulted in the streamflow response. Catchment scale manipulations at experimental watersheds such as the Coweeta Hydrologic Laboratory in the Southern Appalachians of North Carolina, the Fernow Experimental Forest in the Central Appalachians of West Virginia, and Hubbard Brook Experimental Forest in New Hampshire involve various intensities and types of management activities, as well as variation in watershed characteristics such as aspect, elevation, and size (Adams et al. 2008). These long-term watershed studies provide a powerful database from which we can examine the effects of managing for early successional habitats on streamflow amount, timing, and quality.

Annual streamflow generally increases for the first few years after forest canopy removal,, but the magnitude, timing, and duration of the response varies considerably among ecosystems. Using data from water yield studies across the globe, a general model suggests that for each percent of the forest removed streamflow increases 2.5–3.3 mm (Calder 1993; Stednick 1996); however, general models typically explain less than 50% of the variation of the streamflow increase (Stednick 1996) due to high variability in stand structure, pre- and post-harvest species composition, and the interaction between vegetation and climate. In some cases, streamflow returns to pre-harvest levels within 10–20 years. In others, streamflow remains higher, or can even be lower than pre-harvest flow, for several decades after cutting. This wide variation in temporal response patterns is attributable to the complex interactions between climate and vegetation, which can vary considerably from dry to wet to snow-dominated climatic regimes, and with differences in vegetation structure and phenology (coniferous vs. deciduous forest) (McNab, Chap. 2).

While gauged watershed studies provide the foundation for quantifying streamflow responses to forest disturbances, process-level studies are required to fully understand the structural and functional attributes that regulate the magnitude and duration of responses. For example, timber harvest simultaneously alters forest structure by reducing leaf area index, interception surface area, and vegetation height. Harvesting also alters forest function by changing the relative abundance of plant species (Loftis et al., Chap. 5; Elliott, Chap. 7), and the physical environment by changing the energy balance, wind environment, hydrologic flowpaths, and soil temperature and moisture. The topographic/edaphic complexity and high vegetation diversity of forest ecosystems in the Central Hardwood Region is likely to result in a wide range of streamflow response patterns. A more in depth understanding of the factors regulating these response patterns can help managers create and maintain early successional habitats and protect or enhance water resources.

Water quality can also be substantially affected by management activities that create early successional habitats and can have detrimental impacts on aquatic habitats and organisms (Moorman et al., Chap. 11). Research indicates that the harvest of forest biomass in itself has little or no measureable impact on sediment yield. Instead, the primary factors that determine sediment yield are the forest operations required to remove logs, such as roads and skid trails, and the implementation and effectiveness of Best Management Practices (BMPs) that either minimize erosion or prevent sediment from reaching the stream. Stream nutrients can also be impacted by creating and maintaining early successional habitats; however, response magnitude and duration vary considerably among chemical constituents, post-disturbance successional dynamics, and other silvicultural practices such as the use of herbicides or fertilizers.

In this chapter we focus on the first several years after harvesting to assess potential impacts of using forest harvests to create early successional habitats on water resources. To provide examples and illustrate concepts, we use data primarily from long-term studies in the Southern Appalachians, but also include and integrate results of studies from watershed experiments in other areas of the Central Hardwood Region. In addition, we include a discussion of the potential implications of climate change and how associated changes in precipitation regimes might interact with early successional habitats.

14.2 The Hydrologic Budget of Forested Watersheds

The three main components of the hydrologic budget of forested watersheds are **inputs** in the form of rain, snow, and ice (P); **outputs** in the form of transpiration, canopy interception, and soil and forest floor evaporation (evapotranspiration, ET), and groundwater recharge and streamflow (RO or runoff); and change in **soil water storage** (S). Thus, the hydrologic budget can be expressed in terms of a simple mass balance equation: $RO = P – ET \pm S$. Over the long-term, changes in soil water storage (S) are assumed to be negligible so that the storage component of the budget is usually ignored.

Understanding components of the water budget is useful for interpreting and predicting potential impacts of creating and maintaining early successional habitats. ET is the primary component influenced by forest cutting. However, significant alterations to hydrologic flowpaths due to compaction, roads, and other physical changes can influence runoff processes as well, especially stormflow. Timber harvesting alters ET by changing forest structure and function, and the micrometeorological factors that drive transpiration and evaporation. Structural changes include less leaf and stem surface area, and change in the distribution and arrangement of branch surface area. A major functional change that ensues when shifting from mature trees to seedlings, sprouts, and herbaceous vegetation is a decrease in abundance of plant species with conservative water use, resulting in increased transpiration per unit leaf area (Wallace 1988). The vegetation layer can also be more coupled to the atmosphere after forest harvest, thus changing energy balances and wind profiles (Swift 1976; Swank and Vose 1988). For example, Sun et al. (2010) found that net radiation of an 18-year old loblolly pine plantation was 20% higher than a younger stand (4–6 year old) in on the Coastal Plain of North Carolina, resulting in a 25% higher ET in the former.

14.3 Streamflow Responses to Forest Removal

14.3.1 Amount and Timing

Forest harvesting increases annual streamflow in almost all cases in the Central Hardwood Region (Jackson et al. 2004). For example, average increases (% increase relative to that expected based on flow in a reference watershed) in water yield for the first 2 years after cutting ranges from 9.1% at Hubbard Brook in New Hampshire, 14.3% at the Fernow in West Virginia, and 23.0% at the Coweeta Hydrologic

Table 14.1 Post-treatment streamflow response expressed as a percentage increase relative to expected streamflow (adapted from Vose et al. 2010)

Experimental forest	Average annual response (first 2 years post-cut)	Minimum	Maximum
Coweeta, NC ($n = 6$)	23.0	10.3	44.1
Fernow, WV ($n = 3$)	14.3	10.8	18.2
Hubbard Brook, NH ($n = 3$)	9.1	1.7	18.9

Laboratory in North Carolina (Table 14.1). Comparing clearcut harvests with and without BMPs in hardwood forest in eastern Kentucky, Arthur et al. (1998) found a 138% (without BMPs) and a 123% (with BMPs) increase in streamflow during the initial 17 month post-cutting period. Water yield was still 15 to 12% greater 8 years after cutting for the BMP and without BMP watersheds, respectively (Arthur et al. 1998). Differences among regions are likely the result of a complex array of factors, but syntheses of worldwide data from watershed experiments suggest that absolute increases after cutting are greatest in high rainfall areas (Bosch and Hewlett 1982; Swank and Johnson 1994). Other factors include soil depth, the proportion of the annual water budget accounted for by ET, and annual snow fall. The amount of steamflow response is greatest during the first few years following treatment and can be estimated for upland hardwood forests using a model (Douglass and Swank 1975) where first year streamflow increase (water yield) is predicted as a function of the amount of basal area removed and an index of solar radiation inputs:

$$\text{Yield} = 0.00224 * (\text{BA} / \text{PI})^{1.4462},$$

where

Yield = first year increase in streamflow (cm),
BA = amount of basal area removed (%), and
PI = solar insolation index.

Highest yields are observed when 100% of the forest is harvested on north facing slopes. On south or west facing slopes where solar radiation inputs are greater, first year responses are lower because ET on harvested south facing slopes is not as responsive to the increased energy load as ET on harvested north facing slopes. The model also includes an equation to predict the exponential decline in streamflow response as the forest re-grows and LAI recovers (Swank and Douglass 1975). Applications of the model indicate good performance in the Southern Appalachians (Swank and Johnson 1994; Swank et al. 2001) and other eastern deciduous and coniferous forests (Douglass and Swank 1975; Douglass 1983).

Forest cutting can also impact streamflow timing throughout the year and alter storm hydrographs. For example, in areas with high snowfall and shallow soils, cutting increases the proportion of annual streamflow in the spring and summer months due to faster snowmelt and reduced transpiration. In areas with deeper soils and higher precipitation, typical of the Southern Appalachians, flow increases are greatest in the late summer and fall, and may extend into the winter months (Swank and Johnson 1994). For example, on a south facing clearcut watershed in the Southern Appalachians, streamflow increased by approximately 48% during August through October, a time when flows from mature forests are typically lowest (Swank et al. 2001). Storm hydrographs (i.e., a graphical analysis of stream flow vs. time during and after storm events) can also be impacted by cutting and the effects of timber harvesting on flooding have been a focus of intense debate and research for the past several decades (Lull and Reinhart 1972; Andreassian 2004; Eisenbies et al. 2007). Flooding is defined by hydrologic events that exceed bankfull. The linkage between timber harvesting, storm hydrographs, and flooding is complex, and can be better understood by examining the components of stormflow, and then dissecting how forest harvesting influences these components. Streamflow is comprised of baseflow and stormflow, with the latter being described by both the magnitude (peakflow) and duration (stormflow volume). Flooding occurrence and severity is determined largely by peakflow (essentially analogous to stage or the height of the stream) and stormflow volume (the amount of flow contributed by the storm). In forests of the Central Hardwood Region, peakflow and stormflow volume are primarily affected by forest operations that create soil disturbances that alter stormflow pathways; chief among these operations is the road network. For example, in the Southern Appalachians, stormflow volume was nearly double on a watershed logged with a high road density (Douglass and Swank 1976) compared to a watershed logged with a low road density (Swank et al. 2001). However, increases were still relatively minor (10% increase for the low road density watershed versus 17% increase for the high road density watershed). Peak discharges increased on the low road density watershed by up to 15% (Swank et al. 2001). In other sites where trees were felled, but no material removed and no roads were built, peakflow rates increased very little over all (<7%) although stormflow volume increased by 11% (Hewlett and Helvey 1970). In West Virginia, peak discharges after logging were up to four times greater during the growing season (Patric and Reinhart 1971) and they were up to 30% greater after cutting in New Hampshire (Hornbeck 1973).

If BMPs are implemented, most of the physical impacts related to harvest soil disturbances (e.g., skid trails, landing decks, etc.) are short-lived and have little impact on flood risk over the long-term. In contrast, construction of roads and associated engineering related to road surfacing, drainage, culvert design and location are much longer lasting. Depending on the design and surface area impacted, these can permanently alter hydrologic flow paths and storm hydrographs. In short, road design needs to focus on "disconnecting" the surface water draining from the road network to the stream network. Analyses of the impacts of cutting on downstream flooding suggests that many extreme flood events are unrelated to forest cutting and associated road networks and skid trails. Instead, they are primarily determined by storm size and intensity (Perry and Combs 1998; Kochendorfer et al. 2007) and occur regardless of forest management activities.

14.3.2 Duration of Streamflow Response

Among the biological and physical process changes that occur with timber harvest, the duration of streamflow response primarily depends on how quickly leaf and sapwood area recover, and the physiological and structural characteristics of the tree species that occupy the site after the cutting. Long-term streamflow responses for six watersheds in the Southern Appalachians illustrate the temporally variable nature of the response. The response depends on both the forest management objective (e.g., thinning, species conversion, clear cut, etc.,) and how the resulting vegetation responds to climate (Fig. 14.1). Few watershed treatments show no effect (e.g., zero line represents no difference between observed and expected flow based on flow from the reference watershed); and more importantly, few of the watersheds have returned to expected levels after 20 years. For example, where timber harvesting was followed by a species conversion (in this case, from deciduous hardwood to conifer, Fig. 14.1a–b), annual streamflow returned to reference levels after approximately 10 years, marking the point in time when canopy closure was complete. Thereafter, streamflow has been about 25% lower on the conifer dominated watershed (relative to the hardwood reference watershed) due to higher interception and year round transpiration by conifers (Swank and Douglass 1974; Ford et al. 2011).

Variation in sapwood area and species composition among hardwood species during succession can also play an important role in determining the magnitude and timing of streamflow responses after cutting (Ford et al. 2011). For example, transpiration rates for a given diameter yellow-poplar (*Liriodendron tulipifera*) are nearly twofold greater than hickory (*Carya* spp.) and fourfold greater than oaks (*Quercus* spp.). Yellow-poplar transpiration and stomatal conductance rates are also much more responsive to climatic variation compared to oaks and hickories (Ford et al. 2011) (Table 14.2). Xylem anatomy and resulting sapwood area are important determinants of stand transpiration (Wullschleger et al. 2001). For example, transpiration of trees with diffuse-porous, ring-porous, semi-ring-porous, and tracheid xylem anatomies vary more among these three xylem types than they do within a type by species (Fig. 14.2). Diffuse ring porous species have greater sapwood area than ring- or semi-ring porous species and as sapwood area increases, potential water transport increases (Enquist et al. 1998; Meinzer et al. 2005). Hence, if the early successional stand is dominated by diffuse porous species such as yellow-poplar, black birch (*Betula lenta*), or red maple (*Acer rubrum*), we would expect that growing season transpiration in an average year to be much greater (and hence, lower streamflow) than stands dominated by ring-porous species such as oaks or hickories, and likewise be more responsive to climatic variation. In most cases, post-harvest or post-disturbance vegetation succession in the Appalachians is a complex mix of species in both space and time (Elliott and Vose 2011) which makes simple extrapolations difficult. For example, as eastern hemlock (*Tsuga canadensis*) declines and its basal area is reduced by attack from an invasive exotic insect, black birch, a diffuse-porous sapwood species, is dominating early successional trajectories of leaf and sapwood area response (Orwig et al. 2002). This shift in species composition has the potential to increase transpiration by 30% (and thus correspondingly decrease streamflow) (Daley et al. 2007).

To fully understand and predict how post-harvest shifts in the relative abundance of tree species regulate streamflow response (e.g., to explain the variation shown in

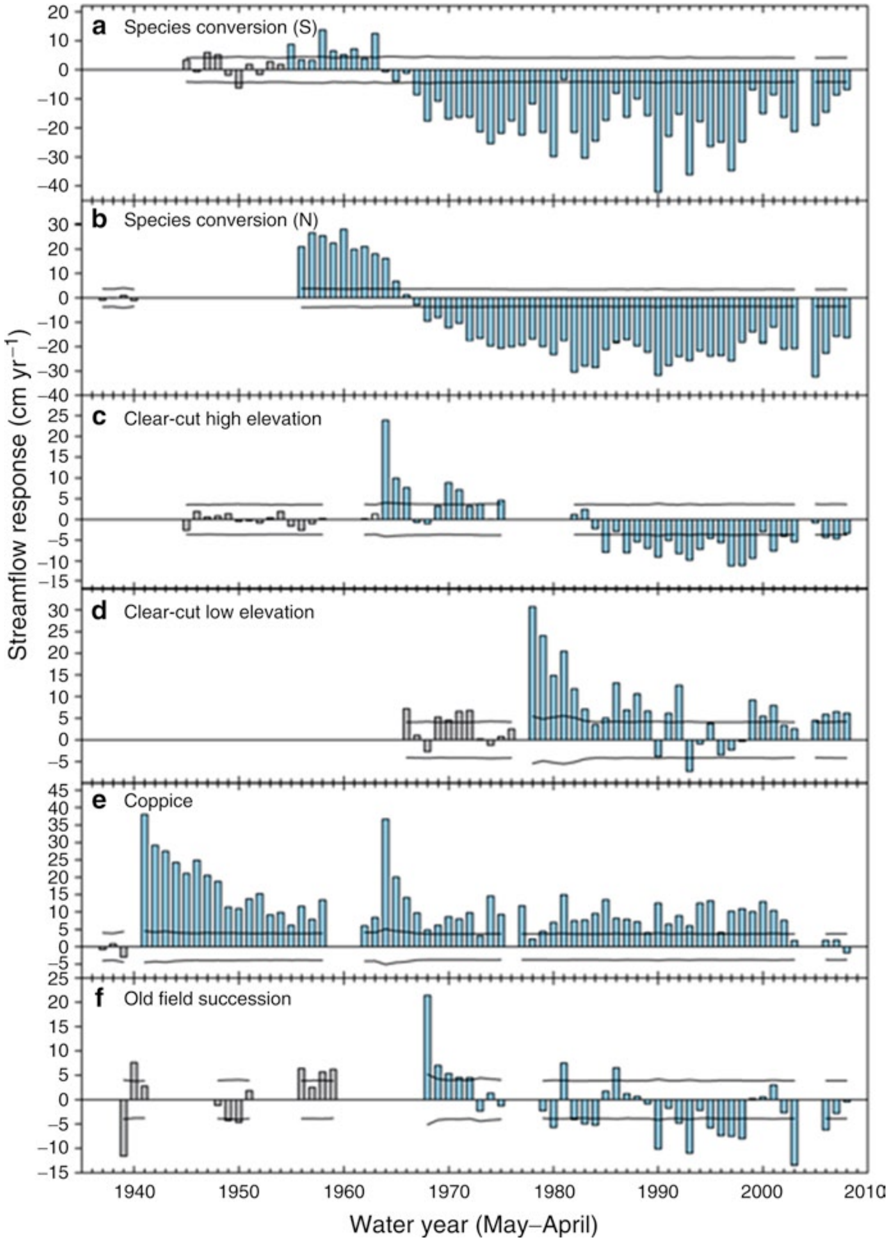

Fig. 14.1 Streamflow response (D, cm yr^{-1}) to forest cutting in the Southern Appalachians (see Swank and Crossley (1988) for site and treatment descriptions). Grey bars depict the calibration period and cyan bars depict streamflow response after treatments. Solid lines on either side of the zero line are 95% confidence intervals; data within the confidence intervals do not differ from zero. Species conversion treatments involved cutting hardwood species and planting *Pinus strobus* on north (N) and (S) facing watersheds (from Ford et al. 2011)

Table 14.2 Mean (standard error) growing season daily transpiration per unit leaf area (E_L, mm) for four hardwood species (Adapted from Ford et al. 2011). Within columns, species not sharing the same lowercase letters denote significant differences among species for that year. Within rows, years not sharing the same uppercase letters denote significant differences among years for that species

	Year		
Species	2004	2005	2006
Carya spp.	0.20 (0.03) b, A	0.19 (0.02) b, A	0.18 (0.02) c, A
Liriodendron tulipifera L.	0.45 (0.05) a, AB	0.39 (0.07) a, B	0.46 (0.03) a, A
Quercus prinus L.	0.21 (0.03) b, A	0.07 (0.01) b, B	0.10 (0.02) cd, AB
Quercus rubra L.	0.10 (0.02) b, A	0.07 (0.02) b, A	0.07 (0.01) c, A

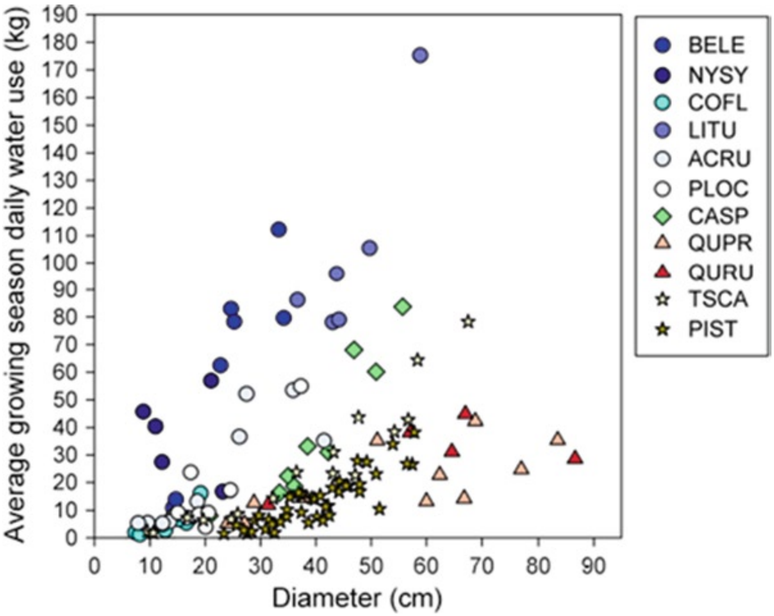

Fig. 14.2 Observed daily water use (DWU) estimated from sap flux density in trees of varying species (legend text denotes first two letters of Latin binomial: BELE *Betula lenta*, NYSY *Nyssa sylvatica*, COFL *Cornus florida*, LITU *Liriodendron tulipifera*, ACRU *Acer rubrum*, PLOC *Platanus occidentalis*, 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. 2011). TSCA data are from Ford and Vose (2007). BELE, NYSY, COFL, ACRU, and PLOC are from (C. Ford and J. Vose, unpublished) but follow the methods in (Ford et al. 2011). 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, stars are for species with tracheid xylem anatomy (from Vose et al. 2011)

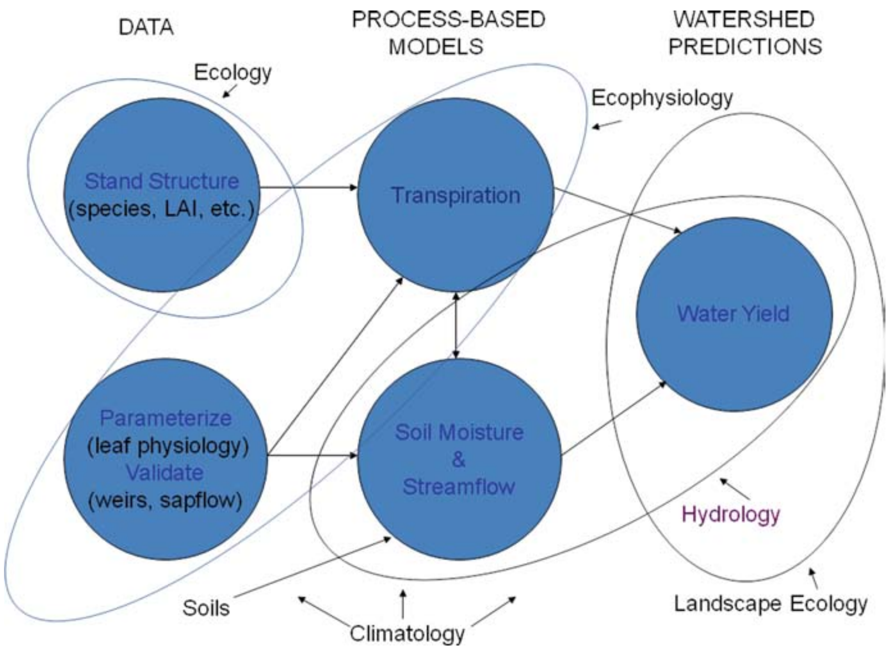

Fig. 14.3 Interdisciplinary approaches to understanding impacts of forest management and other disturbances on water yield requires linking species dynamics and physiology, soil moisture dynamics, and climate across scales ranging from leaves to landscapes (from Vose et al. 2011)

the empirical data shown in Fig. 14.1), we need to be able to link spatially explicit (i.e., cove, midslope, ridge, etc.) predictions of species composition and structure with: (1) species-specific physiology, (2) soil moisture and subsurface flow dynamics, and (3) microclimate. This is a significant departure from traditional hydrologic sciences and requires a multidisciplinary, multi-scale approach (Fig. 14.3).

14.4 Water Quality Responses

Considerable research has been conducted on the effects of forest harvesting on water quality in upland hardwood forests, as well as the development of BMPs to minimize impacts (Kochendorfer and Hornbeck 1999; Jackson et al. 2004; Sun et al. 2004). The most impacted water quality parameter is sediment load, although water temperature and dissolved nutrient concentrations can also be affected. The impact of all of these parameters can be reduced or eliminated with proper planning and BMP implementation. Thus, water quality from streams draining early successional forests can be as high from streams draining undisturbed forested catchments.

Sediment delivery to streams occurs primarily as a result of erosion from roads and skid trails associated with logging (Anderson et al. 1976; Swift 1988; Swank et al. 2001). For example, logging without BMPs resulted in annual sediment losses on the order of 3.1 MT ha^{-1} in the Central Appalachians compared to 0.04 MT ha^{-1} in uncut reference watersheds (Jackson et al. 2004). Careful layout and construction of roads and skid trails minimizes impacts (Swift 1988). However, roads and skid trails are particularly vulnerable to erosion during and shortly after construction, and stream crossings are the most likely locations for sediment delivery to streams. In a study in the Southern Appalachians examining the effectiveness of road construction BMPs, the majority of sediment was generated in two large storms that occurred shortly after new road construction and declined to pre-cut levels after road stabilization and reduced use after logging (Swank et al. 2001). Thus, it is critical to implement BMPs to ensure that newly constructed roads are quickly stabilized and that water and sediment moving from the forest roads and associated components such as ditches and cut banks is dispersed into areas that are disconnected from the streams to ensure infiltration and sediment trapping (Swift and Burns 1999). For example, in eastern Kentucky, BMPs such as streamside buffer strips and proper road construction and rehabilitation reduced suspended sediment considerably compared to a watershed clearcut without BMPs (Arthur et al. 1998). By contrast, other management activities that can be used to create early successional habitats without roads and skid trails (e.g., high intensity prescribed burning) are much less likely to cause a decline in water quality. For example, felling and burning low quality pine-hardwood stands in the Southern Appalachians resulted in no off-site movement of sediment (Swift et al. 1993).

Stream temperature, which affects dissolved oxygen concentration, may also be impacted by timber harvesting and the creation of early successional habitat. However, the magnitude and duration of the increase depends on the width of riparian buffers and the size of the harvested area. In the Central Hardwood Region, removal of forest canopy adjacent to forest streams increases maximum summer stream water temperatures by as much as 6°C (Swift and Messer 1971; Hornbeck and Federer 1975; Swift 1983; Clinton et al. 2010; Clinton 2011). However, maintaining a riparian forest buffer reduces or eliminates this effect (Hornbeck et al. 1986; Moore et al. 2005; Clinton 2011). For example, Clinton (2011) found that a buffer width as narrow as 10 m was adequate to prevent an increase in stream temperature after cutting. In addition, when only small areas of riparian forest canopy are removed, stream temperature responses are often dampened or eliminated within relatively short distances (e.g., 150 m) downstream (Clinton et al. 2010).

Disruption of terrestrial nutrient cycling processes through both alteration of soil abiotic conditions and reduced vegetation nutrient uptake can lead to nutrient transport into streams. Forest ecosystems are characterized by conservative nutrient cycling; most chemical constituents are limiting and tightly cycled by biogeochemical processes. Creating early successional habitats results in a considerable disruption to nutrient cycling processes and alters the environmental characteristics that regulate them. Opening the forest canopy increases soil temperature, and reduced transpiration rates increase soil moisture (Swank and Vose 1988). Both soil temperature and moisture influence nutrient cycling. For example, warmer and wetter soils result in increased nitrogen (N) mineralization and nitrification (Knoepp and Swank 2002; Knoepp and Vose 2007). Hence, these systems can transform N held tightly in organic matter to more mobile inorganic forms such as nitrate-N (NO_3^-).

In undisturbed forests, N typically limits productivity; most available N is used by the vegetation or immobilized by microbes. When nutrient uptake is disrupted by forest harvesting, combined with accelerated mineralization and nitrification, excess nutrients can be transported to streams. Studies examining changes in streamwater chemistry after timber harvesting have found that increases in nutrient concentrations can occur (especially for NO_3^-), losses are generally small relative to overall site nutrient pools and have little or no impact on water quality (Arthur et al. 1998; Martin et al. 2000; Swank et al. 2001). Nutrient responses tend to be greater in higher latitudes where nutrient cycling processes are more limited by temperature compared to responses at lower latitudes and elevations (Hornbeck et al. 1986). However rapid re-establishment of vegetation (both woody and herbaceous) plays a major in sequestering nutrients and re-establishing nutrient cycling processes. Indeed, major losses of nutrients (especially N, but also calcium and potassium) have been observed when vegetation regrowth is precluded by herbicides (Likens et al. 1970). Hence, one of the key BMPs to keep nutrients on site is to ensure rapid re-establishment of vegetation.

14.5 Potential Interactions with Climate Change

Because of the combination of biological and physical controls on hydrologic processes, climate change will both directly and indirectly impact the nation's water resources (Brian et al. 2004; Sun et al. 2008). The direct impacts of climate change on water resources will depend on how climate change alters the amount, type (e.g., snow vs. rain), and timing of precipitation; how this influences baseflow, stormflow, groundwater recharge, and flooding; and how these new hydrologic regimes interact with land use types (see Wear, Chap. 16). Long-term USGS streamflow data suggest that average annual streamflow has increased and this increase has been linked to greater precipitation in the eastern continental USA over the past 100 years (Lins and Slack 1999; Karl et al. 1995; IPCC 2007). However, fewer than 66% of all Global Circulation Models (GCMs) can agree on the predicted change in direction of future precipitation, e.g., wetter vs. drier (IPCC 2007). Inter- and intra-annual precipitation variability in the continental USA is a natural phenomenon related to large-scale global climate teleconnections (e.g., El Niño Southern Oscillation, Pacific Decadal Oscillation, North Atlantic Oscillation). Many regions of the USA have experienced an increased frequency of precipitation extremes over the last 50 years (Easterling et al. 2000a; Huntington 2006; IPCC 2007). As the climate warms in most GCMs, the frequency of extreme precipitation events increases across the globe (O'Gorman and Schneider 2009). However, the timing and spatial distribution of extreme events are among the most uncertain aspects of future climate scenarios (Karl and Knight 1998; Allen and Ingram 2002). Despite this uncertainty, recent experience with droughts and low flows in many areas of the USA indicate that even small changes in drought severity and frequency will have a major impact on society, including drinking water supplies (Easterling et al. 2000b; Luce and Holden 2009).

Most of the world's knowledge of the interactions among management, climate, vegetation, soils, and streamflow has been derived from long-term experiments on paired catchments. A key question is whether this knowledge, built primarily on empirical relationships under historical climate regimes, will allow robust predictions of responses under future climatic regimes. Creating early successional habitats has the potential to alter the hydrological responses to climate change again by influencing biological factors that determine evapotranspiration and physical factors that create soil disturbances or alter hydrologic flow paths. Management activities that favor or replace one species (or several species) over another can alter ET through direct and indirect changes in transpiration or interception (Ford et al. 2011, Stoy et al. 2006). For example, land management practices that favor high transpiration and interception may create conditions that mitigate the impacts of higher rainfall, but worsen the impacts of drought. As a result, streamflow responses (amount and timing) and recovery rates may be different under future climates. In general, hydrologic responses to climate change are larger in the humid Central Hardwood Region (McNab, Chap. 2). than in drier regions, and most climate models suggest the eastern USA will become more water-stressed (Sun et al. 2008). Thus, understanding the role of vegetation in hydrologic processes becomes increasingly important in the Central Hardwood Region as the climate gets warmer and more variable.

14.6 Summary

Because of the tight linkage between vegetation, soils, and water quantity and quality, creating early successional habitats has both direct and indirect impacts on water resources in the Central Hardwood Region. Decades of research using paired catchments in upland hardwood forests has shown:

1. Streamflow increases substantially in the first few years after cutting, but increases decline as sites revegetate and leaf area recovers. Streamflow increases are greater where precipitation is highest and where evapotranspiration represents a large portion of the overall site water budget.
2. The magnitude and rate of recovery to pre-disturbance streamflow depends on species composition and how species vary in transpiration and leaf and sapwood areas. Diffuse-porous species such as blackgum (*Nyssa sylvatica*), red maple, black birch, and yellow-poplar have the highest transpiration rates, while species with ring- or semi-ring porous sapwood, such as oaks and hickories, generally have the lowest transpiration rates for a given diameter. As such, watersheds dominated by the former would be expected to return to pre-cut streamflow levels faster than watersheds dominated by the latter; but depending on how the post-treatment vegetation differs from the pre-treatment vegetation, streamflow responses may be permanently higher or lower than reference conditions.

3. Stormflow increases by 10–20% following cutting and is directly proportional to the density and design of forest roads. However, these increases have not been shown to contribute to downstream flooding.
4. Sediment is the primary concern in terms of water quality responses to cutting and the primary sediment sources are roads and skid trails. BMPs have proven to be effective in reducing sediment.
5. Land managers will need to consider the potential interactions among future climate, changing vegetation structure and function, and physical impacts of forest operations on water resources.

As long as BMPs are properly implemented and maintained, creating early successional habitats in upland hardwood forests by harvesting trees is not likely to have a significant negative impact on either water quantity or water quality. However, it is also clear that forest operations associated with forest cutting (such as roads, stream crossings, culverts, etc.) can create permanent changes to hydrologic flow paths and serve as long-term sources of concern for water quantity and quality. In short, ensuring that BMPs are properly implesmented and functional requires a long-term commitment by land managers. Finally, much of what we know about the effects of disturbances on water resources (and the BMPs required to minimize those effects) has been developed from empirical data under historical climate regimes. Climatic conditions predicted for the eastern USA under climate change scenarios suggests the need for close monitoring of BMP effectiveness and the development of new BMPs applicable to more extreme climatic conditions in the future.

Literature Cited

Adams MB, Loughry L, Plaugher L (2008) Experimental forests and ranges of the USDA Forest Service. Gen Tech Rep NE-321, USDA Forest Service Northeastern Research Station, Newtown Square

Allen MR, Ingram WJ (2002) Constraints on future changes in climate and the hydrologic cycle. Nature 419:224–232. doi:10.1038/nature01092

Anderson H, Hoover M, Reinhart K (1976) Forest and water: effects of forest management on floods, sedimentation, and water supply. Gen Tech Rep PSW-18, USDA Forest Service Pacific Southwest Forest and Range Experiment Station, Berkeley

Andreassian V (2004) Waters and forests: from historical controversy to scientific debate. J Hydrol 291:1–27. doi:10.1016/j.jhydrol.2003.12.015

Arthur MA, Coltharp GB, Brown DL (1998) Effects of best management practices on forest streamwater quality in eastern Kentucky. J Am Water Resour Assoc 34:481–495. doi:10.1111/j.1752-1688.1998.tb00948.x

Bosch JM, Hewlett JD (1982) A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J Hydrol 55:3–23. doi:10.1016/0022-1694(82)90117-2

Brian HH, Callaway M, Smith J, Karshen P (2004) Climate change and U.S. water resources: from modeled watershed impacts to national estimates. J Am Water Resour Assoc 40:129–148. doi:10.1111/j.1752-1688.2004.tb01015.x

Brown AE, Zhang L, McMahon TA, Western AW, Vertessy RA (2005) A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. J Hydrol 310:28–61. doi:10.1016/j.jhydrol.2004.12.010

Calder IR (1993) Hydrologic effects of land use change. In: Maidment D (ed) Handbook of hydrology. McGraw-Hill, New York

Clinton BD (2011) Stream water responses to timber harvest: riparian buffer width effectiveness. For Ecol Manage 261:979–988. doi:10.1016/j.foreco.2010.12.012

Clinton BD, Vose JM, Fowler DL (2010) Flat Branch monitoring project: stream water temperature and sediment responses to forest cutting in the riparian zone. Res Pap SRS-51, USDA Forest Service Southern Research Station, Asheville

Daley MJ, Phillips NG, Pettijohn C, Hadley JL (2007) Water use by eastern hemlock (*Tsuga canadensis*) and black birch (*Betula lenta*): implications of effects of the hemlock woolly adelgid. Can J For Res 37:2031–2040. doi:10.1139/X07-045

Douglass JE (1983) The potential for water yield augmentation from forest management in the eastern United States. J Am Water Resour Assoc 19:351–358. doi:10.1111/j.1752-1688.1983.tb04592.x

Douglass JE, Swank WT (1975) Effects of management practices on water quality and quantity: Coweeta Hydrologic Laboratory, North Carolina. Municipal watershed management symposium proceedings. Gen Tech Rep NE-13, USDA Forest Service Northeastern Forest Experiment Station, Broomall

Douglass JE, Swank WT (1976) Multiple use in southern Appalachian hardwoods – a ten-year case history. In: Proceedings of the 16th international union of forestry research organization world congress, IUFRO, Oslo

Easterling DR, Evans JL, Groisman PY, Karl TR, Kunkel KE, Ambenje P (2000a) Observed variability and trends in extreme climate events: a brief review. Bull Am Meteorol Soc 81:417–425

Easterling DR, Meehl GA, Parmesan C, Changnon SA, Karl TR, Mearns LO (2000b) Climate extremes: observations, modeling, and impacts. Science 289:2068–2074. doi:10.1126/science.289.5487.2068

Eisenbies MH, Aust WM, Burger JA, Adams MB (2007) Forest operations, extreme flooding events, and considerations for hydrologic modeling in the Appalachians – a review. For Ecol Manage 242:77–98. doi:10.1016/j.foreco.2007.01.051

Elliott KJ, Vose JM (2011) The contribution of the Coweeta Hydrologic Laboratory to developing an understanding of long-term (1934–2008) changes in managed and unmanaged forests. For Ecol Manage 261:900–910. doi:10.1016/j.foreco.2010.03.010

Ellison AM, Bank MS, Clinton BD, Colburn EA, Elliott K, Ford CR, Foster DR, Kloeppel BD, Knoepp JD, Lovett GM, Mohan J, Orwig DA, Rodenhouse NL, Sobczak WV, Stinson KA, Stone JK, Swan CM, Thompson J, Holle BV, Webster JR (2005) Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Front Ecol Environ 9:479–486. doi:10.1890/1540-9295(2005) 003[0479:LOFSCF]2.0.CO;2

Enquist BJ, Brown JH, West GB (1998) Allometric scaling of plant energetics and population density. Nature 395:163–165. doi:10.1038/25977

Ford CR, Vose JM (2007) *Tsuga canadensis* (L.) Carr. mortality will impact hydrologic processes in southern Appalachian forest ecosystems. Ecol Appl 17:1156–1167. doi:10.1890/06-0027

Ford CR, Hubbard RM, Vose JM (2011) Quantifying structural and physiological controls on canopy transpiration of planted pine and hardwood stand species in the southern Appalachians. Ecohydrology 4:183–195. doi:10.1002/eco.136

Hewlett JD, Helvey JD (1970) Effects of forest clear-felling on the storm hydrograph. Water Resour Res 6:768–782

Hornbeck JW (1973) Storm flow from hardwood-forested and cleared watersheds in New Hampshire. Water Resour Res 9:346–354

Hornbeck JW, Federer CA (1975) Effects of management practices on water quality and quantity: Hubbard brook experimental forest, New Hampshire. Municipal watershed management symposium proceedings. Gen Tech Rep NE-13, USDA Forest Service Northeastern Forest Experimental Station, Broomall

Hornbeck JW, Martin CW, Pierce RS, Bormann FH, Likens GE, Eaton JS (1986) Clearcutting northern hardwoods: effects on hydrologic and nutrient ion budgets. For Sci 32:667–686

Huntington TG (2006) Evidence for intensification of the global water cycle: review and synthesis. J Hydrol 319:83–95. doi:10.1016/j.jhydrol.2005.07.003

IPCC (2007) Contribution of working Groups I, II and III to the fourth assessment report of the intergovernmental panel on climate change. In: Core writing team, Pachauri RK, Reisinger A (eds) Climate change 2007: synthesis report. Geneva

Jackson CR, Sun G, Amatya DM, Swank WT, Riedel MS, Patric J, Williams T, Vose JM, Trettin CC, Aust WM, Beasley RS, Williston H, Ice GG (2004) Fifty years of forest hydrology in the southeast. In: Ice GG, Stednick JD (eds) A century of forest and wildland watershed lessons. Society of American Foresters, Bethesda

Jones JA, Post DA (2004) Seasonal and successional streamflow response to forest cutting and regrowth in the northwest and eastern United States. Water Resour Res 40:W05203. doi:10.1029/2003WR002952

Karl TR, Knight RW (1998) Secular trends of precipitation amount, frequency, and intensity in the USA. Bull Am Meteorol Soc 79:231–241

Karl TR, Knight RW, Plummer N (1995) Trends in high-frequency climate variability in the twentieth century. Nature 377:217–220. doi:10.1038/377217a0

Knoepp JD, Swank WT (2002) Using soil temperature and moisture to predict forest soil nitrogen mineralization. Biol Fertil Soils 36:177–182. doi:10.1007/s00374-002-0536-7

Knoepp JD, Vose JM (2007) Regulation of nitrogen mineralization and nitrification in Southern Appalachian ecosystems: separating the relative importance of biotic vs. abiotic controls. Pedobiologia 51:89–97. doi:10.1016/j.pedobi.2007.02.002

Kochendorfer JN, Hornbeck JW (1999) Contrasting timber harvesting practices illustrate the value of BMPs. In: Proceedings of the 12th central hardwood forest conference. Gen Tech Rep SRS-24, USDA Forest Service Southern Research Station, Asheville

Kochendorfer JN, Adams MB, Miller GW, Helvey JD (2007) Factors affecting large peakflows on Appalachian watersheds: lessons from the Fernow Experimental Forest. Res Pap NRS-3, USDA Forest Service Northern Research Station, Newtown Square

Likens GE, Bormann FH, Johnson NM, Fisher DW, Pierce RS (1970) Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed-ecosystem. Ecol Monogr 40:23–47

Lins H, Slack JR (1999) Streamflow trends in the United States. Geophys Res Lett 26:227–230. doi:10.1029/1998GL900291

Luce CH, Holden ZA (2009) Declining annual streamflow distributions in the Pacific Northwest United States, 1948–2006. Geophys Res Lett 36:L16401. doi:10.1029/2009GL039407

Lull HW, Reinhart KG (1972) Forests and floods in the eastern United States. Res Pap NE-226, USDA Forest Service Northeastern Forest Experimental Station, Broomall

Martin CW, Hornbeck JW, Likens GE, Buso DC (2000) Impacts of intensive harvesting on hydrology and nutrient dynamics of northern hardwood forests. Can J Fish Aquat Sci 57:19–29. doi:10.1139/cjfas-57-S2-19

Meinzer FC, Bond BJ, Warren JM, Woodruff DR (2005) Does water transport scale universally with tree size? Funct Ecol 19:558–565. doi:10.1111/j.1365-2435.2005.01017.x

Moore RD, Spittlehouse DL, Story A (2005) Riparian microclimate and stream temperature response to forest harvesting: a review. J Am Water Resour Assoc 41:813–834. doi:10.1111/j.1752-1688.2005.tb03772.x

O'Gorman PA, Schneider T (2009) The physical basis for increases in precipitation extremes in simulations of 21st-century climate change. Proc Natl Acad Sci 106:14773–14777. doi:10.1073/pnas.0907610106

Orwig DA, Foster DR, Mausel DL (2002) Landscape patterns of hemlock decline in New England due to the introduced hemlock woolly adelgid. J Biogeogr 29:1475–1487. doi:10.1046/j.1365-2699.2002.00765.x

Patric JH, Reinhart KG (1971) Hydrologic effects of deforesting two mountain watersheds in West Virginia. Water Resour Res 7:1182–1188. doi:10.1029/WR007i005p01182

Perry CA, Combs LJ (1998) Summary of floods in the in the United States, January 1992 through September 1993. Water supply paper 2499, USGS, Desoto

Reinhart KG, Pierce RS (1964) Stream-gaging stations for research on small watersheds. Agri Handbook 268. USDA

Stednick JD (1996) Monitoring the effects of timber harvest on annual water yield. J Hydrol 176:79–95. doi:10.1016/0022-1694(95)02780-7

Stoy P, Katul G, Siqueira M, Juang J, Novick K, McCarthy HR, Oishi AC, Umbelherr J, Kim H, Oren R (2006) Separating the effects of climate and vegetation on evapotranspiration along a successional chronosequence in the southeastern US. Glob Change Biol 12:2115–2135. doi:10.1111/j.1365-2486.2006.01244.x

Sun G, Riedel M, Jackson R, Kolka R, Amatya D, Shepard J (2004) Influences of management of Southern forests on water quantity and quality. In: Rauscher HM, Johnsen K (eds) Southern forest sciences: past, current, and future. Gen Tech Rep SRS-75, USDA Forest Service Southern Research Station, Asheville

Sun G, McNulty SG, Myers JAM, Cohen EC (2008) Impacts of multiple stresses on water demand and supply across the southeastern United States. J Am Water Resour Assoc 44:1441–1457. doi:10.1111/j.1752-1688.2008.00250.x

Sun G, Noormets A, Gavazzi MJ, McNulty SG, Chen J, Domec JC, King JS, Amatya DM, Skaggs RW (2010) Energy and water balance of two contrasting loblolly pine plantations on the lower coastal plain of North Carolina, USA. For Ecol Manage 259:1299–1310. doi:10.1016/j.foreco.2009.09.016

Swank WT, Crossley DA Jr (1988) Introduction and site description. In: Swank WT, Crossley DA Jr (eds) Ecological studies, vol. 66: forest hydrology and ecology at Coweeta. Springer, New York

Swank WT, Douglass JE (1974) Streamflow greatly reduced by converting deciduous hardwood stands to pine. Science 185:857–859. doi:10.1126/science.185.4154.857

Swank WT, Johnson CE (1994) Small catchment research in the evaluation and development of forest management practices. In: Moldan B, Cerny J (eds) Biogeochemistry of small catchments: a tool for environmental research. Wiley, Chichester

Swank WT, Vose JM (1988) Effects of cutting practices on microenvironment in relation to hardwood regeneration. In: Guidelines for regenerating Appalachian hardwood stands: proceedings of a workshop. SAF Publication 88-034. West Virginia University Books, Morgantown

Swank WT, Vose JM, Elliott KJ (2001) Long-term hydrologic and water quality responses following commercial clearcutting of mixed hardwoods on a southern Appalachian catchment. For Ecol Manage 143:163–178. doi:10.1016/S0378-1127(00)00515-6

Swift LW Jr (1976) Algorithm for solar radiation on mountain slopes. Water Resour Res 12:108–112. doi:10.1029/WR012i001p00108

Swift LW Jr (1983) Duration of stream temperature increases following cutting in the southern Appalachian mountains. In: Proceedings of the international symposium on hydrometeorology. American Water Resource Association, Bethesda

Swift LW Jr (1988) Forest access roads: design, maintenance, and soil loss. In: Swank WT, Crossley DA Jr (eds) Ecological studies, vol. 66: forest hydrology and ecology at Coweeta. Springer, New York

Swift LW, Burns RG (1999) The three Rs of roads. J Forest 97:40–45

Swift LW Jr, Messer JB (1971) Forest cuttings raise temperatures in small streams in the southern Appalachians. J Soil Water Conserv 26:111–116

Swift LW Jr, Elliott KJ, Ottmar RD, Vihnanek RE (1993) Site preparation burning to improve southern Appalachian pine-hardwood stands: fire characteristics and soil erosion, moisture, and temperature. Can J For Res 23:2242–2254. doi:10.1139/x93-278

Vose JM, Ford CR, Laseter S, Sun G, Adams MB, Dymond S, Sebestyen S, Campbell J, Luce C (2010) Can forest watershed management mitigate climate change impacts on water resources? Unpubl Rep, USDA Forest Service, Southern Research Station, Asheville

Vose JM, Sun G, Ford CR, Bredemeier M, Ostsuki K, Wei A, Zhang Z, Zhang L (2011) Forest ecohydrological research in the 21st century: what are the critical needs? Ecohydrology 4:146–158. doi:10.1002/eco.193

Wallace LL (1988) Comparative physiology of successional forest trees. In: Swank WT, Crossley DA Jr (eds) Ecological studies, vol. 66: forest hydrology and ecology at Coweeta. Springer, New York

Wullschleger SD, Hanson JJ, Todd DE (2001) Transpiration from a multi-species deciduous forest as estimated by xylem sap flow techniques. For Ecol Manage 143:205–213