Terrestrial and Longitudinal Linkages of Headwater Streams

By

J. Bruce Wallace [1,2]

and

S. L. Eggert [2]

Departments of Entomology [1] and Ecology [2]

University of Georgia

Athens, GA 30602

Recommended citation

Wallace, J.B. and S.L. Eggert. 2009. Terrestrial and longitudinal linkages of headwater streams. *In* Canaan Valley and its Environs: A Landscape Heritage Celebration. Canaan Valley Institute, Davis, West Virginia. http://www.canaanvi.org/canaanvi_web/uploadedFiles/Events/Past_Events/Wallace%20&%20Eggert%20Paper.pdf

Abstract

Headwater streams are important habitats for aquatic invertebrates, amphibians and fish. Within forested regions, headwater streams combined with adjacent forests and riparian zones are sites of organic matter deposition, storage, processing, and subsequent transport. In most streams draining deciduous forest, this organic matter from the surrounding forest provides the major fuel for the ecosystem. In addition to serving as habitat, these streams perform many valuable ecosystem services such as nutrient retention and transformation, hydraulic retention, sediment retention, thermal refuges, moderation of thermal regimes, and are important sites of secondary production for higher animals, including fish and birds. Unfortunately, headwater streams are being subjected to a diverse array of insults, which includes dams, urbanization (including residential, commercial and industrial use), agriculture, forestry, and mining interests, from man without fully considering the long-term consequences of our actions. Ecologists need to promote the importance of headwater streams as well devote more research to examining entire stream networks rather than individual, or simple longitudinal linkages along one system.

Extensive forest, rugged relief, abundant rainfall, and thousands of small streams characterize the central and southern Appalachians. These small streams are the headwaters, or portions of headwaters, of many large rivers including the Alabama, Chattahoochee-Apalachicola, Delaware, James, Ohio, Potomac, Roanoke, Santee, Savannah, Susquehanna, Tennessee, and Yadkin-Pee Dee. These rivers represent an important water resource for many major metropolitan areas (Wallace et al. 1992). Thus, proper management of the headwaters of these waterways is vital to maintaining downstream water quality. In many respects, the Canaan Valley, as a high elevation wetland with many tributary streams, is an excellent example of some of the past misuses of our headwaters. However, these streams suffer from numerous insults since early colonization of the area. These insults include extensive logging, especially around 1900 to the 1920s, serious impairments from mining and acid mine drainage, agriculture, as well as increasing urbanization and road construction (Meyer and Wallace 2001).

Our objectives here are to emphasize linkages between headwater streams and their terrestrial ecosystems, as well as, linkages between the headwater reaches and downstream areas. These linkages are critical for understanding lotic systems and how impairment in one reach may impact downstream segments, and headwater streams have contributed to our overall understanding of ecological systems. Headwater streams have advanced our knowledge of several aspects of ecosystems including the following: 1) detrital food webs; 2) biogeochemistry and nutrient dynamics; 3) linkages between ecosystems; 4) ecological consequences of exotic species; and 5) as harbingers of ecosystem change. First, we will explore how headwater streams have contributed to our ecological understanding of these phenomena.

Most headwater streams in the Appalachians drain, or did drain in the past, watersheds consisting primarily dense forests. These streams are often heavily forested even during winter, by a dense riparian understory of evergreen rhododendron. In-stream primary production tends to be a small fraction of total organic matter available for heterotrophic organisms such as bacteria, fungi, and invertebrates (Webster et al. 1995, Webster and Meyer 1997). Most of the energy base of these streams comes from the surrounding forest as coarse particulate organic matter (CPOM) consisting of leaves and woody debris with significant amounts of dissolved organic carbon (DOC) entering by groundwater (Webster and Meyer 1997). The invertebrates in these streams rely primarily on detritus and with its associated microbial assemblage for most of their secondary production (Hall et al. 2000, Rosi-Marshall and Wallace 2002). In the process of gleaning their nutrition from this detritus, invertebrates also play an important role in detrital processing and upstream to downstream linkages.

Evidence for the Role of Invertebrates in Detrital Processing in Headwater Streams

It has long been known that many invertebrates, primarily insects, readily consume autumn shed leaves that fall into streams (i.e., Hynes 1941, Petersen and Cummins 1974, Wallace et al. 1999). How important are these invertebrates in the processing of organic matter at the ecosystem level? On two different occasions two headwater streams at the Coweeta Hydrologic Laboratory in western North Carolina were treated with the insecticide methoxychlor. These treatment effects included: 1) massive invertebrate drift (primarily insects) resulting in altered community structure (Wallace et al. 1989, 1991b, Lugthart and Wallace 1992). 2) Abundance, biomass, and secondary production of invertebrates were reduced, especially of aquatic insect taxa (Lugthart et al. 1990, and Lugthart and Wallace 1992) and increased abundance on non-insect taxa (primarily oligochaetes, copepods, and turbellarians) occurred. 3) There are reduced rates of leaf litter processing without reduction in microbial respiration or taxa of fungi (Cuffney et al. 1990, Suberkropp and Wallace 1992). 4) The leaf litter standing crop increased 2.5 to 3X in the treatment stream compared with the two reference streams by the end of the 3rd –summer of treatment (Wallace et al. 1991a). Leaf litter processing rates remained low throughout the pesticide treatment and subsequently increased during recovery (Chung et al. 1993), as recolonization by aerial adults (Wallace et al. 1991b) of several taxa was fairly rapid (Whiles and Wallace 1992).

Assimilation efficiencies of most leaf-shredding insects is low, in the range of 10% (Wallace and Hutchens 2000), which means that approximately 90% of material ingested is egested as fine organic particles. These small particles are much more amenable to downstream transport than larger coarse particulate organic matter. Reductions in organic matter processing rates in the treatment stream were followed by a 5-fold reduction in export of fine particulate organic matter (FPOM) from the treatment stream compared to reference streams (Cuffney et al. 1990). During treatment, FPOM export from this small headwater stream was reduced by about 170 to 200 kg AFDM (Wallace et al. 1991c). Invertebrate manipulation also reduced export per unit maximum discharge during storms (Cuffney and Wallace 1989), as well as changed seasonal export response to storms (Wallace et al. 1991a). FPOM concentrations and export subsequently increased in the treatment stream with invertebrate recovery.

The impact of biotic manipulation of the invertebrate assemblage on the pesticide treated stream was as great as that produced by a range of discharges encompassing a 57-yr record (Wallace et al. 1991a). If one applies the FPOM export per 100 m of wetted 1st-order channel measure during the pesticide manipulation to the 33.3 km of 1st-order channels in the 1600 ha Coweeta Basin, these data indicate that macroinvertebrate activities in the headwaters contribute about 6 to 7 metric tons of FPOM annually to downstream reaches in the basin. This study demonstrated the importance of stream biota, particularly invertebrates, on the processing of CPOM and subsequent export of FPOM. Collectively these studies emphasize the importance of maintaining biodiversity in the headwaters as an important consideration in downstream management (Wallace et al. 1991a, Lugthart and Wallace 1992).

Importance of Longitudinal Linkages

There are other large-scale physical changes along stream gradients in addition to local physical gradients. These longitudinal changes are incorporated into a general framework of riverine ecosystems as the river continuum concept (RCC) (Vannote et al. 1980, Minshall et al. 1983), although results from a number of areas around the world indicate that changes in energy supply and biological communities as proposed in the RCC are not always applicable to all river systems (e.g., Statzner and Higler 1985, Meyer 1990, Cushing et al. 1995). In the Little Tennessee River Basin of the southern Appalachian Mountains, many biological attributes of streams are similar to those proposed in the original RCC, although some exceptions exist (Grubaugh et al. 1997). Striking differences in production of invertebrate assemblages occurred between the headwaters and downstream reaches of the river (Grubaugh et al. 1997). Shredders, gatherers, and predators dominated the invertebrate assemblage production in the headwaters, whereas in the larger Little Tennessee River, 80% of the secondary production was attributable to filter-feeding taxa, which are adapted to remove particles from suspension. At the large river sites, production per m^2 of substrate exceeded that of the shaded, headwater stream by 20 X (Grubaugh et al. 1997).

The dissimilarities in production and community structure between upstream and downstream sites of the L. Tennessee River continuum resulted from resources being unequally distributed along the river gradient (Wallace and Hutchens 2000). Hall et al. (2000) and Rosi-Marshall and Wallace (2002) used the trophic basis of production method (Benke and Wallace 1980) to estimate annual food consumption by invertebrates at upstream and downstream locations. Invertebrates in the headwaters consumed primarily leaf and amorphous detritus stored in the stream, which is easily obstructed by woody debris and highly retentive of CPOM. In contrast, primarily amorphous detritus and animal tissue are consumed in downstream areas (Rosi-Marshall and Wallace 2002). In the headwater stream draining a deciduous forest, there was an abundant supply of stored benthic organic matter, which was 8X greater than that found downstream. In contrast, annual transport of organic matter per linear m of stream of the larger river site exceeded that of the headwater stream by >260X. Thus, there were large differences in the form of organic matter, i.e., stored versus transported, available to benthic animal assemblages and these were reflected in the functional structure of these assemblages, i.e., shredders and collectors, versus filter-feeding organisms. Filter-feeding organisms in the Little Tennessee River were supported by the most available resource, FPOM in transport, which was delivered from upstream areas.

The invertebrate assemblage is dominated by shredders, gatherers, and invertebrate predators in small headwater streams, where they exploit the physical environment of stored organic particles (Wallace and Hutchens 2000). Their feeding activities tend to decrease particle size of organic resources and favor downstream export of FPOM, which is more amenable to transport than larger CPOM. By contrast, downstream reaches have much higher discharge, greater stream power, and less retention, which promote entrainment of organic matter. Again, the invertebrate assemblage exploits the physical characteristics of the system by a dominance of filterers (Grubaugh et al. 1997), which promote retention of entrained organic matter. Thus, in both the upstream and downstream areas, the invertebrate feeding assemblages have evolved to effectively utilize the physical characteristics of the system. In doing so, the headwater stream assemblage promotes breakdown and transport of organic matter and forms an important linkage to downstream areas.

Export of invertebrates in the form of downstream drift from headwater streams is also another important source of food for downstream animals such as fish. Invertebrate export from fishless headwater streams in southern Alaska was estimated to support 100-2000 young-of-the-year salmonids in downstream habitats (Wipfli and Gregovich 2002).

Terrestrial Aquatic Linkages

Forested Headwater streams are intimately connected with the terrestrial environment (Richardson 2000, Hutchens and Wallace 2002, Gomi et al. 2002). Terrestrially derived inputs of organic matter, leaves and woody debris, are the fuel that drive aquatic productivity within small streams (Fisher and Likens 1973, Webster and Meyer 1997, Richardson 1991). An ecosystem level manipulation of organic matter inputs over an 8-year period in the southern Appalachian Mountains of western North Carolina provided compelling evidence of just how tight the coupling is between headwater streams and riparian habitats. A gill-net canopy constructed over a 170 m stream reach, starting at the headwater spring seep, kept out allochthonous inputs from the surrounding forest (Wallace et al. 1997, 1999). A nearby reference stream served to distinguish litter exclusion effects from natural variation. Most leaf litter disappeared from the exclusion stream within 6 months. The loss of leaf litter in the stream bottom resulted in a pulse of organic and inorganic particles as they were flushed from the stream (Wallace et al. unpublished data). After three years of litter exclusion, all small wood was manually removed. Two years later, all large woody debris was removed by hand from the stream channel. Benthic storage of organic material further declined with the removal of wood from the stream bottom.

Reductions in benthic organic matter over the first 4 years of exclusion resulted in a 78% decline in invertebrate secondary production in mixed substrates of the exclusion stream compared to pretreatment years (Wallace 1999). By the end of the study, secondary production in the exclusion stream was the second lowest ever measured for a north temperate stream based on data provided by Benke (1993). The shredder and gatherer functional feeding groups, which depend entirely on organic matter resources from the terrestrial environment, were impacted most severely. Some detritivores with flexible feeding habits, such as *Tipula* spp. and *Tallaperla* spp. switched diets from leaf material to wood prior to wood removal, and then switched to amorphous detritus after wood removal (Hall et al. 2000, Eggert et al. unpublished data). Other taxa such as the caddisfly, *Pycnopsyche gentilis* did not shift diets and did not survive in the litterdepleted stream (Eggert and Wallace in review). Negative effects of reduced detrital inputs propagated up to invertebrate predators and salamanders, the top predator in the small streams. Populations of the two-lined salamander, *Eurycea wilderae* were significantly lower and grew at slower rates than those in the reference stream (Johnson 2002).

The flow of organic materials between terrestrial and aquatic habitats is not unidirectional. Organic matter and immature aquatic insects from the stream bottom can be deposited within the riparian zone during large storms (Wallace et al. 1995, Hutchens and Wallace 2002). Emerging aquatic insects also travel up into the riparian habitats and serve as food for terrestrial organisms (Nakano and Murakami 2001, Sanzone 2001, Sabo and Power 2002).

Biogeochemistry and nutrient dynamics

Anthropogenic inputs of nutrients through fossil fuel burning, wastewater effluent, fertilizer application, and urban runoff have altered nutrient cycles in the Appalachians and globally (Carpenter et al. 1998). Headwater streams are important sites of nutrient uptake and help reduce downstream nutrient loading. In the late 1990s a team of scientists from a number of institutions around the country started a Lotic Intersite Nitrogen eXperiment (LINX) where they studied the transformation and uptake of nitrogen in various sized streams at 12 sites, primarily Long-Term Ecological Research (LTER) sites representing a diverse range of biomes.

The LINX team used ^{15}N tracer released into streams to study the uptake and transformation of nitrogen. Headwater streams retained over 50% of inorganic nitrogen inputs from their watershed (Peterson et al. 2001). Nitrogen was transformed or removed quickly from small streams, often within minutes to hours of input. The shortest uptake distances of nitrogen were in the smallest streams. There was much less uptake, as indicated by longer distances traveled by nitrogen, in larger streams with higher discharge. Their data suggested that it is the small streams in basins that are probably most important in regulating nitrogen dynamics.

Small streams are often filled with organic detritus from the riparian habitat and have small cross sectional areas that allow maximum interface between substrates and the flowing water. In conjunction with the litter exclusion study in the southern Appalachians, Meyer et al. (1998) found that the biogeochemistry of dissolved organic carbon (DOC) was altered following the elimination of leaf litter inputs. DOC generation from leaf litter deposited in small stream contributed approximately 30% of the daily DOC export. Since DOC is an important source of organic matter in stream food webs, its absence in streams decoupled from their terrestrial habitat would negatively impact higher tropic levels (Meyer 1994). The average uptake distance of phosphorus and ammonium increased following leaf litter exclusion and increased even more following woody debris removal (Webster et al. 2002). In these small streams, microbial organisms colonize surfaces of leaves and small woody debris and immobilize dissolved nutrients (Tank and Webster 1998, Tank et al. 1998). Leaves and wood in small streams also serve to slow the flow of water. The presence of leave, wood, and associated microbes, prevents the movement of nutrients to downstream reaches. In essence, small streams serve a similar function as the human kidney by cleansing the system of wastes (Meyer 1990).

As with the flow of organic matter from streams to the terrestrial environment, nutrients likewise can move from stream to land. One well-known example of this phenomenon is the transfer of marine-derived nutrients by salmon in the Pacific Northwest (Naiman et al. 2002, Gende et al. 2002). Nutrients released from salmon carcasses following spawning, stimulates periphyton and invertebrate production in headwater streams (Wipfli et al. 1998, Chaloner and

Wipfli 2002). Bears, birds and other mammals transport nutrients in the form of salmon carcasses from streams to the terrestrial environment when they consume carcasses directly (Hilderbrand et al. 1999), or feed on the increased invertebrate biomass (Gende and Willson 2001). Salmon-derived nutrients may also increase the growth of riparian vegetation (Helfield and Naiman 2001).

Problems with assessing small streams

Small streams are critical to the functioning of the larger drainage network. Unfortunately, the importance of small streams is often under appreciated and underestimated (Meyer and Wallace 2001). Headwater streams are often inadequately mapped. First order streams make up 48% of the total river miles in the United States (Leopold 1964). However, maps of basin networks are usually drawn at a scale of 1:24,000 or larger, which excludes the smallest streams (Leopold 1994). Using the Coweeta Creek basin as an example, over 98% of the total stream length is unaccounted for on 1:500,000 scale maps (Table 1). Many of the smallest streams do not appear on 1:7,200 scale maps. It is ironic that over 190 papers have been published based on work completed in Coweeta headwater streams that do not exist according to U.S.G.S. maps (Meyer and Wallace 2001). For the Chattooga River watershed in the Blue Ridge Mountain of Georgia, North Carolina, and South Carolina, only 50% and 75% of perennial streams were shown on 1:100,000 and 1:24,000 scale maps, respectively (Hansen 2001). Almost none of the intermittent and ephemeral streams in the Chattooga basin were drawn on either map.

Table 1. Stream distances in the Coweeta Hydrologic Laboratory (16.3 km^2) in western North Carolina from maps of various scales.[1]

Map Scale	Kilometers of streams
1:500,000	0.5 km
1:24,000	24.4 km
1:7,200	56.0 km*

[1] Data courtesy of N. Gardiner, Institute of Ecology, University of Georgia

* There are many permanent streams that do appear on the 1:7,200 scale maps.

The problem exists because there are no hydrological criteria for the mapping of ephemeral (dashed blue lines) and perennial (solid blue lines) on U.S. Geological Survey maps. Most headwater streams are mapped according to the personal aesthetics of laboratory-bound technicians (Leopold 1994). Hansen (2001) defined perennial, intermittent and ephemeral streams based on channel presence, flow duration, bed water level, aquatic insect presence, material movement, and channel materials. The state of West Virginia defines intermittent streams as those streams having watersheds of at least one square mile. West Virginia also uses a biological criterion where streams are classified as intermittent if they do not support species that require a continuous aquatic period of at least 6 months. Many small drainages and spring seeps of < 50 acres can support animals with multiyear life cycles, but they appear as dashed blue lines on topographic maps. In order to properly protect all of the waters of the U.S. under the

Clean Water Act, hydrologic and biologically meaningful definitions of the smallest streams in river networks must be determined soon.

Small Streams Under Assault

Headwater streams in the Appalachians tend to have stenothermal temperatures compared to downstream areas (e.g., Vannote and Sweeney 1980). They offer unique thermal refuges to many unique species of invertebrates and amphibians. Very few taxonomic studies to the species level of identification have been made in the small intermittent and permanent streams of the central Appalachians. However, Morse et al. (1993, 1997) point out that much of the biodiversity and unique fauna of the Appalachians are found in these headwater streams. Unfortunately, as these latter authors point out these headwaters are under assault from many anthropogenic disturbances.

<u>Ecological consequences of exotic species</u>

Aquatic exotic species alter community dynamics in steams and complete for preferred habitat with native species. The headwaters of the Appalachian Mountains have not been affected tremendously by the introduction of exotic species with the exception of the rainbow trout (*Oncorhynchus mykiss*), which has replaced native brook trout (*Salvelinus fontinalis*) in second and third order streams of the Southern Appalachians (Larson and Moore 1985). Rainbow trout compete with brook trout for food (Ensign et al.1990) and may affect growth rates and habitat selection by brook trout (Whitworth and Strange 1983, Lohr and West 1992).

The close linkages between headwaters and terrestrial environments are evident because some of the terrestrial exotics have produced large changes in headwater streams. Outbreaks of terrestrial invaders such as the balsam and hemlock woolly adelgid and the gypsy moth may result in riparian canopy loss and pesticide runoff into nearby streams, affecting stream functioning (Griffith et al. 1996, Hutchens and Benfield 2000, Orwig 2002, Snyder et al. 2002). Nonnative scale and fungi causing diseases such as dogwood anthracnose, and beech bark disease have already invaded forests of the Appalachian Mountain region, and the fungus causing butternut canker is beginning to spread rapidly (Ward and Mistretta 2002). Another introduced fungus causing chestnut blight eliminated the American chestnut from eastern forests and has lasting impacts on Appalachian streams (Wallace et al. 2001). The elimination of American chestnut led to decreased leaf litter processing, decreased quality of litter inputs, and decreased invertebrate growth rates in headwater streams (Smock and MacGregor 1988). The input of large woody debris into streams between 1934 and the 1950s as a result of chestnut blight occurred prior to the second logging of Appalachian forests. This fortuitous event facilitated the retention of sediment and served to stabilize stream channels following subsequent logging (see Wallace et al. 2001). Clearly introduced species within streams and riparian habitats can have long lasting effects on headwater stream functioning.

Agriculture

The filling and tiling of former wetlands and headwater streams for agriculture has greatly reduced surface water area in the Appalachians and worldwide. The drainage density of the Kävlinge River catchment in Skåna, a southern province of Sweden has been severely altered for extensive agriculture (Wolf 1956). Between 1812 and 1953, 96.6% of the original surface water area in the catchment was lost due to channelization and drainage of streams for agriculture. Along with the loss of small streams, intensive agriculture in the region has resulted in nitrate levels in water that exceed safe drinking water levels. Over fertilization of agricultural land in low order sections of river networks often affects downstream river reaches. David and Gentry (2000) estimated that agricultural sources in Illinois contributed 10-15% of nitrogen and phosphorus loads to the Mississippi River. Dramatic shifts in the invertebrate community associated with increased sedimentation and temperature have been observed from headwater to downstream reaches of agriculturally impacted streams (Harding et al.1999). Steam fishes also are susceptible to sediment inputs from agricultural activities (Walser and Bart 1999, Waters 1995). Along with sedimentation effects, insecticide runoff from agricultural fields into headwater streams can have deleterious impacts on stream water quality (Liess et al. 1999).

Agricultural activity has the potential to modify and reduce the diversity of stream biota for many years after reforestation in the Appalachians (Harding et al. 1998). In 1940, a mountain farm experiment began at Coweeta Hydrologic Lab using standard Southern Appalachian farming techniques (mule and plow). Initially, no increase in storm runoff or soil loss was observed due to the presence of organic matter in the soil (Hursh 1951). The disappearance of organic matter from the soil three years after clearing resulted in an average of 768 pounds of sediment lost per day from May to September 1943. During one storm in 1949, 152,000 pounds of sediment was carried into the stream in a 65-minute period. The effects of cattle grazing on a headwater stream in a mountain watershed were also demonstrated at Coweeta. No visible effects on stream water quality were observed over the first 8 summers of grazing (Hush 1951). After the 9th summer of grazing, however, the 8 head of cattle had trampled an area large enough to cause increased storm runoff into stream channels, which flushed leaf packs from the small stream. Without leaf litter to trap sediment and slow the runoff of stormflow, the maximum effects were finally seen well after the experiment had begun, demonstrating the unique ecosystem services provided by organic matter accumulations in small streams.

Urbanization and Roads

Urban growth scenarios predict substantial (0.5->10%) increases in population and income growth for the central and southern Appalachian regions (Wear 2002). Losses of forested land are expected to occur in areas of increased urbanization. Largest forest losses are forecasted to occur in the Southern Appalachian Piedmont, the Blue Ridge Mountains, the Ridge and Valley, and the Southern Cumberland Plateau (Wear 2002). The Canaan Valley in West Virginia has also experienced rapid growth in the last 20 years due to increased recreation, tourism and residential development (Waldron and Wiley 1996).

The replacement of forested land and riparian habitats with impervious surfaces such as roads, rooftops, and lawns with urbanization alters the hydrology and geomorphology of streams (Finkenbine et al. 2000, Paul and Meyer 2001, Rose and Peters 2001). Increases in surface runoff associated with stormflow leads to declines in water quality, and increases in bacterial populations and turbidity (Bolstad and Swank 1997). Concomitant with increased urbanization is the increase of pesticide, herbicide, and fertilizer runoff into nearby streams during storms from residential and commercial property (Hoffman et al .2000, Winter and Duthie 2000). Increased sediment runoff from construction activities and erosion due to channel widening (Nelson and Booth 2002, Trimble 1997) results in habitat loss for aquatic life (Waters 1995). Fish species diversity and abundance declined significantly in Tuckahoe Creek, Virginia after 32 years of increased road construction, commercial and residential development and riparian losses (Weaver and Garman 1994). In southeastern Pennsylvania streams, only pollution tolerant species of fish and macroinvertebrates could survive in urbanized streams (Kemp and Spotila 1997). Urbanization has also been associated with less diverse, pollution tolerant macroinvertebrate assemblages in southeastern Wisconsin streams (Stepenuck et al.2002), south-central Maine streams (Huryn et al. 2002), and Georgia Piedmont streams (Roy et al, 2003).

Measurable aquatic degradation occurs at a level of about 10% impervious area (Booth and Jackson 1997, Wang et al. 2002, 2001). The construction of impervious surfaces such as roads has been long been associated with decreased water quality of nearby streams (Duncan et al. 1987, Forman and Alexander 1998, Jones et al, 2000, Wemple et al. 2001). Ruth Cooper Allman, a lifelong resident of Canaan Valley, also wrote of the disappearance of “millions of brook trout in the streams when pioneers came to the valley,” (Allman 1976). The construction of West Virginia Route 32 in 1932 resulted in so much sediment runoff into nearby streams that residents reported, “the water became so muddy in the spring that the fish either died or had to leave as they could not live in the muddy water,” (Allman 1976). Skid rails and logging roads are often major sources of sediment in streams located in logged watersheds and have significant effects on aquatic organisms (Tebo 1955). Soil erosion from logging roads has been studied extensively at the Coweeta Hydrologic Laboratory in western North Carolina (Swift 1988) and the Fernow Experimental Forest in West Virginia (Trimble 1977). Guidelines for building environmentally friendly and low cost forest roads have been pioneered in demonstration projects at both sites (Swift 1984a, 1984b, Kochenderfer and Helvey 1987). Long-term monitoring at one of these demonstration projects (Watershed 7 and Coweeta) showed that large sediment runoff occurred only during storms immediately after road construction (Swank et al. 2001).

Forestry Practices

Most Appalachian headwater streams were exposed to a major press, or chronic disturbance around the turn of the century with the widespread logging of Appalachian forest. An excellent and informative account of the early logging in West Virginia, including the Canaan Valley and surrounding areas can be found in Clarkson (1964). Early logging obviously represented major disturbances on streams as logs were often floated downstream with the aid of splash dams which scoured stream beds, and based on photographs (see Clarkson 1964) it is evident that many log slides were constructed in the channels, whether intermittent or permanent of small headwater streams. Logging also causes may other problems for streams and headwater biota. The include: changes in stream temperature regimes (Swift 1983); increased discharge and altered hydrology (Bormann and Likens 1979, Swank et al 2001); increased nutrients (Swank et al. 2001), solar radiation, and primary production (Webster et al. 1983, Duncan and Brusven 1985, Noel et al. 1986); increased sediment export (Gurtz et al. 1980, Webster and Golladay 1984, Swank et al. 2001); and changes in dissolved organic matter from the terrestrial ecosystem (Meyer and Tate 1983). These changes can be accompanied by big changes in the energy base of the stream with a shift from allochthonous detritus to autochthonous production (Webster et al. 1983). The physical and energy base changes can lead to large changes in macroinvertebrate community structure (Newbold et al. 1980, Gurtz and Wallace 1984, Noel et al. 1986, Stone and Wallace 1998). Those invertebratae taxa with short life cycles and the ability to exploit increases in primary production greatly increase in population abundance, biomass, and productivity (Wallace and Gurtz 1986). Studies in the central (Griffith and Perry 1991) and southern Appalachians (Benfield et al. 2001) show that long-term patterns of leaf litter breakdown can be altered for many years following logging. However, depending upon the extent of terrestrial succession, invertebrate assemblages can revert toward their pre-logged and forested reference stream condition in about 1.5 decades, although differences in reference and logged streams remain (Stone and Wallace 1998).

Most of the above studies cited are from southern Appalachian streams at the Coweeta Hydrologic Laboratory in western North Carolina. However, these studies clearly show that logging causes as array of physical and biotic disturbances to the streams draining logged catchments. Those that probably cause the most severe changes in benthic assemblages include: increased solar radiation, changed thermal regimes, increased sediments and physical disturbance of the substrate, increased water yield and storm flow.

Dams and Impoundments

Dams and impoundments alter the ecology, geomorphology, temperature, and hydrology of river networks (Stanford and Ward 2001, Chin et al. 2002, Nislow et al. 2002). Alterations of flow regimes and stream network fragmentation lead to direct habitat loss, water quality degradation, and decreased biodiversity of aquatic species (Dynesius and Nilsson 1994, Bunn and Arthington 2002). Small impoundments are common in higher elevations in the central and southern Appalachians (Manzel and Cooper 1992, Merrill 2001). The numbers of these small impoundments, mostly in the form of ponds <10 ha in area, are staggering. In one northern Georgia Piedmont watershed, 46% of 6,167 headwater streams have been impounded (Merrill 2001). Over 5,400 of these small ponds had inundated 8% of the total stream length. The large numbers of dams in the watershed served to severely fragment the river network. Approximately 31% of the stream length had a downstream impoundment within 5 km. Because 1:24,000 maps were used in the above analysis, these values are most certainly underestimates due to unmapped streams and impoundments. Headwater streams flowing into impoundments have lower biological integrity than free-flowing streams (Merrill 2001). Fortunately, many of the impacts of dams and impoundments may be reversible with dam removal, although responses will vary among upstream and downstream areas and will occur at different rates (Bednarek 2001), Hart et al. 2002). Following dam removal longitudinal linkages are re-established, and mobile organisms such as fish may respond relatively quickly (Bushaw-Newton et al. 2002). Changes in channel geomorphology, and sediment and nutrient transport may take longer to respond to dam removal (Doyle et al. 2002, Hart et al. 2002, Pizzuto 2002, Stanley and Doyle 2002).

Mining

Approximately 2017 km^2 of central and southern Appalachians were surface mined between the years 1930 and 1971. Some 32-48% of this mined area was not reclaimed and abandoned mines represent an on-going problem (Samuel et al. 1978). There were reduced invertebrates and fish populations for >20 years following cessation of mining operations, and Odonata, Ephemeroptera, Megaloptera, Diptera were severely affected (Roback and Richardson 1969). Some rivers of West Virginia have been so severely degraded by coal mining and industrialization, especially chemical plants, that only more tolerant species of benthic organisms can inhabit them (Tarter 1976). In recent years heated controversy has developed around the practice of mountaintop removal and valley fill mining (MTR/VF). As of 1998, some 1450 km of streams, primarily in WV, KY, TN and VA had been filled and permanently buried with overburden from mining operations (U.S. Fish and Wildlife Service 1998). Since these estimates of buried streams were made from a USGS 1:24,000 scale map, there is no doubt the value of 1450 km represents a significant underestimate (see above). This practice is extremely controversial and has been covered extensively in the WV Gazette in a series of articles by Ken Ward and others. For a detailed series of articles on this subject see the following website: http://www.wvgazette.com/news/Mining/. This is probably one of the most important environmental concerns facing headwater streams in WV as many questions remain about the practice of burying headwaters (see conclusions).

Conclusions

We are becoming quite knowledgeable about a number of functional aspects of headwater streams. In the last two decades we have begun to study stream segments along longitudinal reaches spanning multiple stream orders (Vannote et al. 1980, Grubaugh et al. 1997). We are making progress on how physical factors such as local geomorphology and discharge influence nutrient uptake, retention, and transformation, as well as, the influence of geomorphology and discharge on detritus retention, food webs and functional structure of biota, and biological processes. However, we need to apply what we know about individual streams and longitudinal linkages to entire stream networks (see Fisher 1997, Meyer and Wallace 2001). Current controversy in West Virginia and nearby states surrounding issues of mountaintop mining and the burial of small streams is an excellent example of such application.

Studies of streams as entire networks in basins are especially important because we are currently engaged in the burial of steams without contemplating the basin-wide consequences of these practices. Headwater streams are exceptionally valuable sites of nutrient uptake and retention. How much increase in downstream nutrients, as well as harmful chemicals in our water supplies we willing to accept from burial of headwater streams? Are we willing to accept the altered hydrology with the propensity to increase downstream flood peaks? These floods are associated with the flushing of organic matter, scouring of alga food resources, and enhanced drift of aquatic animals from streams. How much alteration of aquatic animal assemblages including invertebrates, amphibians, and fishes do we tolerate? The burial of headwater streams eliminates the many linkages between forests and headwaters and downstream segments, how much modification in food webs are we willing to accept? No complete biotic inventory is required for most, if not all, of the buried streams. How much of this loss (in perpetuity) of valuable habitat for invertebrate and amphibian production and potentially higher animals such as fish, birds, etc. that depend on the stream animals, is acceptable? It is obvious that some streams destined to be buried are currently are of excellent quality based on invertebrate assemblages. How much concern do we place on the loss of this biotic diversity and habitat? How much long-term, irrevocable damage to the immediate environment, as well as, potential long-term cumulative effects downstream is the public willing to accept? Finally, once the environment has been degraded and the resources are diminished, is it possible to maintain a reasonable and rewarding standard of living? These are all very important questions that directly affect a substantial area of the central Appalachians and unfortunately decisions are being made without considering the consequences of our actions.

Acknowledgments

Much of the information reported on here was supported by grants from the National Science Foundation (Ecosystems Studies Program and Long-term Ecological Research) and the United States Forest Service. We thank these agencies for their continued support of research on headwater streams.

Literature Cited

Allman, R. C. 1976. Canaan valley and the black bear. McClain Printing Co., Parsons, WV, 118 pp.

Bednarek, A. T. 2001. Undamming rivers: a review of the ecological impacts of dam removal. Environ. Manage. 27:803-814.

Benfield, E. F., J. R. Webster, J. L. Tank, and J. J. Hutchens. 2001. Long-term patterns in leaf breakdown in response to watershed logging. Int. Rev. Hydrobiol. 86:467-474.

Benke, A. C. 1993. Concepts and patterns of invertebrate production in running waters. Proc. Int. Assoc. Theor. Appl. Limnol. 25:15-38.

Benke, A. C., and J. B. Wallace. 1980. Trophic basis of production among net-spinning caddisflies in a southern Appalachian stream. Ecology 61:108-118.

Bolstad, P. V., and W. T. Swank. 1997. cumulative impacts of landuse on water quality in a southern Appalachian watershed. J. Am. Water Resour. Assoc. 33:519-533.

Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems:degradation thresholds, stormwater detection, and the limits of mitigation. J. Am. Water Resour. Assoc. 33:1077-1090.

Bormann, F. H., and G. E. Likens. 1979. Pattern and process in a forested ecosystem. Springer-Verlag, New York. 253 pp.

Bunn, S. E., and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environ. Manage. 30:492-507.

Bushaw-Newton, K. L., and 15 others. 2002. An integrative approach towards understanding ecological responses to dam removal: the Manatawny Creek study. J. Am. Water Resour. Assoc. 38:1581-1599.

Carpenter, S. R., N. F. Caraco, D. L. Correll, R. W. Howarth, A. N. Sharpley, and V. H. Smith. 1998. Nonpoint pollution of surface waters with phosphorus and nitrogen. Ecol. Appl. 8:559-568.

Chaloner, D. T., and M. S. Wipfli. 2002. Influence of decomposing Pacific salmon carcasses on macroinvertebrate growth and standing stock in southeastern Alaska streams. J. N. Am. Benthol. Soc 21:430-442.

Chin, A., D. L. Harris, T. H. Trice, and J. L. Given. 2002. Adjustment of stream channel capacity following dam closure, Yegua Creek, Texas. J. Am Water Resour. Assoc. 88:1521-1531.

Clarkson, R. B. 1964. Tumult on the Mountains: Logging in West Virginia – 1770-1920. McClain Printing Co., Parsons, WV. 410 pp.

Chung, K., J. B. Wallace, and J. W. Grubaugh. 1993. The impact of insecticide treatment on abundance, biomass, and production of litterbag fauna in a headwater stream: a study of pretreatment, treatment, and recovery. Limnologica 28:93-106.

Cuffney, T. F., and J. B. Wallace. 1989. Discharge-export relationships in headwater streams: influence of invertebrate manipulations and drought. J. N. Amer. Benthol. Soc. 8:331-341.

Cuffney, T. F., J. B. Wallace, and G. J. Lugthart. 1990. Experimental evidence quantifying the role of benthic invertebrates in organic matter dynamics of headwater streams. Freshwat. Biol. 23:281-199.

Cushing, C. E., K. W. Cummins, and G. W. Minshall, (eds). 1995. River and stream ecosystems, Vol. 22, Ecosystems of the world. Elsevier, Amsterdam. 817 pp.

David, M. B. and L. E. Gentry. 2000. Anthropogenic inputs of nitrogen and phosphorus and riverine export for Illinois, USA. J. Environ. Qual. 29:494-508.

Doyle, M. W., E. H. Stanley, and J. M. Harbor. 2002. Geomorphic analogies for assessing probable channel response to dam removal. J. Am. Water Resour. Assoc. 38:1567-1579.

Duncan, S. H., R. E. Bilby, J. W. Ward, and J. T. Heffner. 1987. Transport of road-surface sediment through ephemeral stream channels. Water Resour. Bull. 23:113-119.

Duncan W. F. A., and M. A. Brusven. 1985. Energy dynamics of three low-order southeast Alaskan streams: autochthonous production. J. Freshwat. Ecol. 3:115-166.

Dynesius, M., and C. Nilsson. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. Science 266:753-762.

Ensign, W. E., R. J.. Stranger, and S. E. Moore. 1990. Summer food limitation reduces brook and rainbow-trout biomass in a southern Appalachian stream. Trans. Am. Fish. Soc. 119:894-901.

Finkenbine, J. K., J. W. Atwater, and D. S. Mavinic. 2000. Stream health after urbanization. J. Am. Water Resour. Assoc. 36:1149-1160.

Fisher, S. G. 1997. Creativity, idea generation, and the functional morphology of streams. J. N. Am. Benthol. Soc. 16:305-318.

Fisher, S. G., and G. E. Likens. 1973. Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. Eco. Monogr. 43:421-439.

Forman, R. T. T., and L. E. Alexander. 1998. Roads and their major ecological effects. Ann. Rev. Ecol. Syst. 29:207-231

Gende, S. M., R. T. Edwards, M. F. Willson, and M. W. Wipfli. 2002. Pacific salmon in aquatic and terrestrial ecosystems. BioScience 52:917-928.

Gende. S. M. and M. F. Willson. 2001. Passerine densities in riparian forests of southeast Alaska: Potential role of anadromous spawning salmon. Condor 103:624-629.

Gomi, T., R., C. Sidle, and J. S. Richardson. 2002. Understanding processes and downstream linkages of headwater systems. BioScience 52:905-916.

Griffith, M. B., E. M. Barrows, and S. A. Perry. 1996. Effects of aerial application of diflubenzuron on emergence and flight of adult aquatic insects. J. Econ. Ent, 89:442-446.

Griffith, M. B., and S. A. Perry. 1991. Leaf pack processing in 2 Appalachian Mountain streams draining catchments with different management histories. Hydrobiologia 220:247-254.

Grubaugh, J. W., J. B. Wallace, and L. S. Houston. 1997. Production of benthic macroinvertebrate communities along a southern Appalachian river continuum. Freshwat. Biol. 37:581-596.

Gurtz, M. E. and J. B. Wallace. 1984. Sutstrate-mediated response of stream invertebrates to disturbance. Ecology 65: 1556-1569.

Gurtz, M. E., J. R. Webster, and J. B. Wallace. 1980. Seston dynamics in southern appalachian streams: effects of clearcutting. Can. J. Fish. Aquat. Sci. 37: 624-631.

Hall, R. O., J. B. Wallace, and S. L. Eggert. 2000. Organic matter flow in stream food webs with reduced detrital resource base. Ecology 81:3445-3463.

Hansen, W. F. 2001. Identifying stream types and management implications. For. Ecol. Manage. 143:39-46.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones III. 1998. Stream biodiversity: The ghost of land use past. Proc. Nat. Acad. Sci. 95:14843-14847.

Harding, J. S., R. G. Young, J. W. Hayes, K. A. Shearer, and J. D. Stark. 1999. Changes in agricultural intensity and river health along a river continuum. Freshwat. Biol. 42:345-357.

Hart, D. D., T. E. Johnson, K. L. Bushaw-Newton, R. J. Horwitz, A. T. Bednarek, D. F. Charles, D. A. Kreeger, and D. J. Velinsky. 2002. Dam removal: challenges and opportunities for ecological research and river restoration. BioScience 52:669-681.

Helfield, J. M., and R. J. Naiman. 2001. Effects of salmon-derived nitrogen on riparian forest growth and implications for stream productivity. Ecology 82:2403-2409.

Hilderbrand, G. V., T. A. Hanley, C. T. Robbins, and C. C. Schwartz. 1999. Role of brown bears (*Ursus arctos*) in the flow of marine nitrogen into a terrestrial ecosystem. Oecologia 121:546-550.

Hoffman, R. S., P. D. Capel, and S. J. Larson. 2000. Comparison of pesticides in eight U. S. urban streams. Environ. Toxicol. Chem. 19:2249-2258.

Hursh, C. R. 1951. Research in forest-streamflow relations. Unasylva 5:3-9.

Huryn, A. D., V. M. Butz Huryn, C. J. Arbuckle, and L. Tsomides. 2002. Catchment land-use, macroinvertebrates and detritus processing in headwater streams: taxonomic richness versus function. Freshwat. Biol. 47:401-415.

Hutchens, J. J., and E. F. Benfield. 2000. Effects of forest defoliation by the gypsy moth on detritus processing in southern Appalachian streams. Am. Midl. Nat. 143:397-404.

Hutchens, J. J., and J. B. Wallace. 2002. Ecosystem linkages between southern Appalachian headwater streams and their banks: leaf litter breakdown and invertebrate assemblages. Ecosystems 5:80-91.

Hynes, H. B. N. 1941. The taxonomy and ecology of the nymphs of British Plecoptera, with notes on the adults and eggs. Trans. Roy. Ent. Soc. London 91:459-557.

Johnson, B. R. 2002. Effects of resource manipulation on selected primary and secondary consumers in two detritus-based southern Appalachian streams. Ph.D. Dissertation, University of Georgia, Athens, GA 152 pp.

Jones, J. A., F. J. Swanson, B. C. Wemple, and K. U. Snyder. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Conserv. Biol. 14:76-85.

Kemp, S. J., and J. R. Spotila. 1997. Effects of urbanization on brown trout (S*almo trutta*), other fishes and macroinvertebrates in Valley Creek, Valley Forge, Pennsylvania. Am. Midl. Nat. 138:55-68.

Kochenderfer, J. N., and J. D. Helvey. 1987. Using gravel to reduce soil losses from minimum-standard forest roads. J. Soil Water Conserv. 42:46-50.

Larson, G. L., and S. E. Moore. 1985. Encroachment of exotic rainbow trout into stream populations of native brook trout in the southern Appalachian mountains. Trans. Am. Fish. Soc. 114:195-203.

Leopold, L. B., M. G. Wolman, and J. P. Miller. 1964. Fluvial Processes in Geomorphology. W. H. Freeman, San Francisco, CA. 522 pp.

Leopold, L. B. 1994. A View of the River. Harvard University Press, Cambridge, MA. 298 pp.

Liess, M., R. Schulz, M. H. D. Liess, B. Rother, and R. Kruzig. 1999. Determination of insecticide contamination in agricultural headwater streams. Water Res. 33:239-247.

Lohr, S. C., and J. L. West. 1992. Microhabitat selection by brook and rainbow-trout in a southern Appalachian stream. Trans. Am. Fish. Soc. 121:729-736.

Lugthart, G. J., and J. B. Wallace. 1992. Effects of disturbance on benthic functional structure and production in mountain streams. J. N. Am. Benthol. Soc. 11:138-164.

Lugthart, G. J., J. B. Wallace, and A. D. Huryn. 1990. Secondary production of chironomid communities in insecticide-treated and untreated headwater streams. Freshwater Biol. 23:417-427.

Menzel, R. G., and C. M. Cooper. 1992. Small impoundments and ponds. pp. 389-420. *In:* C. T. Hackney, S. M. Adams, and W. H. Martin, (eds), Biodiversity of the Southeastern United States, Aquatic Communities. John Wiley, New York, NY. 779 pp.

Merrill, M. D. 2001. Local and watershed influences on stream fish biotic integrity in the upper Oconee watershed, Georgia, USA. M. S. Thesis, The University of Georgia, Athens, GA. 237 pp.

Meyer, J. L. 1990. A blackwater perspective on riverine ecosystems. BioScience 40:643-651.

Meyer, J. L. 1994. The microbial loop in flowing waters. Microb. Ecol. 28:195-199.

Meyer, J. L., and C. M. Tate. 1983. The effects of watershed disturbance on dissolved organic carbon dynamics of a stream. Ecology 64:33-44.

Meyer, J. L. and J. B. Wallace. 2001. Lost linkages and lotic ecology: rediscovering small streams, pp. 295-317. *In:* M. C. Press, N. J. Huntly and S. Levin (eds.) Ecology: Achievement and Challenge. Blackwell Science, Oxford, UK. 406 pp.

Meyer, J. L., J. B. Wallace and S. L. Eggert. 1998. Leaf litter as a source of dissolved organic carbon in streams. Ecosystems 1:240-249.

Minshall, G. W., R. C. Petersen, K. W. Cummins, T. L. Bott, J. R. Sedelll, C. E Cushing, and R. L. Vannote. 1983. Interbiome comparison of stream ecosystem dynamics. Ecol. Monogr. 53:1-25.

Morse, J. C., B. P. Stark, and W. P. McCafferty. 1993. Southern Appalachian streams at risk: implications for mayflies, stoneflies, caddisflies, and other aquatic biota. Aquat. Conserv: Mar. Freshwat. Ecosyst. 3:292-303.

Morse, J. C., B. P. Stark, W. P. McCafferty, and K. J. Tennessen. 1997. Southern Appalachian and other southeastern streams at risk: implications for mayflies, dragonflies and damselflies, stoneflies, and caddisflies. pp. 17-42. *In:* G. W. Benz and D. E. Collins (eds), Aquatic Fauna in Peril: the Southeastern Perspective. Special Publication 1, Southeastern Aquatic Research Institute. Lanz Design and Communications, Decatur, GA.

Naiman, R. J., R. E. Bilby, D. E. Schindler, and J. M. Helfield. 2002. Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. Ecosystems 5:399-417.

Nakano, S., and M. Murakami. 2001. Reciprocal subsidies: dynamic interdependence between terrestrial and aquatic food webs. Proc. Nat. Acad. Sci. 98:166-170.

Nelson, J. E., and D. B. Booth. 2002. Sediment sources in an urbanizing, mixed land-use watershed. J. Hydrol. 264:51-68.

Newbold J. D., D. C. Erman, and K. B. Roby. 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. Can. J. Fish. Aquat. Sci. 37:1076-1085.

Nislow, K. H., F. J. Magilligan, H. Fassnacht, D. Bechtel, and A. Ruesink. 2002. Effects of dam impoundments on the flood regime of natural floodplain communities in the Upper Connecticut River. J. Am. Water Resour. Assoc. 38:1533-1548.

Noel, D. S., C. W. Martin, and C. A. Federer. 1986. Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. Environ. Manage. 10:661-670.

Orwig, D. A. 2002. Ecosystem to regional impacts of introduced pests and pathogens: historical context, questions and issues. J. Biogeogr. 29:1471-1474.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Ann. Rev. Ecol. Syst. 32:333-365.

Petersen, R. C., and K. W. Cummins. 1974. Leaf processing in a woodland stream. Freshwat. Biol. 4:343-368.

Peterson, B. J., W. M. Wollheim, P. J. Mulholland, J. R. Webster, J. L. Meyer, J. L. Tank, E. Marti, W. B. Bowden, H. M. Valett, A. E. Hershy, W. H. McDowell, W. K. Dodds, S. K. Hamilton, S. Gregory, and D. D. Morrall. 2001. Control of nitrogen export from watersheds by headwater streams. Science 292:86-90.

Pizzuto, J. 2002. Effects of dam removal on river form and process. BioScience 52:683-691.

Richardson, J. S. 1991. Seasonal food limitation of detritivores in a montane stream: an experimental test. Ecology 72:873-887.

Richardson, J. S. 2000. Life beyond salmon streams: Communities of headwaters and their role in drainage networks. pp. 473-476. *In:* L. M. Darling (ed), Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. 15-19 Feb. 1999. Vol. 2. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B. C. 520 pp.

Roback, S. S., and J. W. Richardson. 1969. The effects of acid mine drainage on aquatic insects. Proc. Acad. Nat. Sci. Philad. 121:81-107.

Rose, S., and N. E. Peters. 2001. Effects of urbanization on streamflow in the Atlanta area (Georgia, USA): a comparative hydrological approach. Hydrol. Proc. 15:1441-1457.

Rosi-Marshall, E., and J. B. Wallace. 2002. Invertebrate food webs along a stream resource gradient. Freshwat. Biol. 47: 129-141.

Roy, A. H., A. D. Rosemond, M. J. Paul, D. S. Leigh, and J. B. Wallace. 2003. Stream macroinvertebrate response to catchment urbanization (Georgia, U.S.A.). Freshwat. Biol. 48:329-346.

Sabo, J. L., and M. E. Power. 2002. Numerical response of lizards to aquatic insects and short-term consequences for terrestrial prey. Ecology 83:3023-3036.

Samuel, D. E., J. R. Stauffer, and C. H. Hocutt. (eds). 1978. Surface mining and fish/wildlife needs in the eastern United States. Fish and Wildlife Service, Office of Biological Services (FWS/OBS-78/81). 386 pp.

Sanzone, D. M. 2001. Linking communities across ecosystem boundaries: the influence of aquatic subsidies on terrestrial predators. Ph.D. dissertations, University of Georga, Athans, GA 263 pp.

Smock, L. A., and C. M. MacGregor. 1988. Impact of the American chestnut blight on aquatic shredding macroinvertebrates. J. N. Am. Benthol. Soc. 7:212-221.

Stanford, J. A., and J. V. Ward. 2001. Revisiting the serial discontinuity concept. Regul. Rivers: Res. Manage. 17:303-310.

Stanley, E. H., and M. W. Doyle. 2002. A geomorphic perspective on nutrient retention following dam removal. BioScience 52:693-701.

Statzner, B., and B. Higler. 1985. Questions and comments on the river continuum concept. Can. J. Fish. Aquat. Sci. 42:1038-1044.

Stepenuck, K. F., R. L. Crunkilton, and L. Z. Wang. 2002. Impacts of urban landuse on macroinvertebrate communities in southeastern Wisconsin streams. J. Am. Water Resour. Assoc. 38:1041-1051.

Stone, M. K., and J. B. Wallace. 1998. Long-term recovery of a mountain stream from clear-cut logging: the effects of forest succession on benthic invertebrate community structure. Freshwat. Biol. 39:141-169.

Suberkropp, K. and J. B. Wallace. 1992. Aquatic hyphomycetes in insecticide-treated and untreated streams. J. N. Am. Benthol. Soc. 11:165-171.

Swank, W. T., J. M. Vose, and K. J. Elliott. 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.

Swift, L. W., Jr. 1983. Duration of stream temperature increases following forest cutting in the southern Appalachian Mountains. pp. 273-275. *In:* A. I. Johnson and R. A. Clark (eds), Proceedings of the International Symposium on Hydrometerology, Denver, CO, June 13-17, 1982. American Water Resources Association, Bethesda, MD.

Swift, L. W., Jr. 1984a. Soil losses from roadbeds and cut and fill slopes in the southern Appalachian Mountains. S. J. Appl. For. 8:209-216.

Swift, L. W., Jr. 1984b. Gravel and grass surfacing reduce soil loss from mountain roads. For. Sci. 30:657-670.

Swift, L. W., Jr. 1988. Forest access roads: design, maintenance, and soil loss. pp.313-324. *In:* W. T. Swank and D. A. Crossley, Jr. (eds), Forest Hydrology and Ecology at Coweeta. Springer-Verlag, New York, NY.

Snyder, C. D., J. A. Young, D. P. Lemarié, and D. R. Smith. 2002. Influence of eastern hemlock (*Tsuga Canadensis*) forests on aquatic invertebrate assemblages in headwater streams. Can. J. Fish. Aquat. Sci. 59:262-275.

Tank, J. L., and J. R. Webster. 1998. Interaction of substrate availability and nutrient distribution on wood biofilm development in streams. Ecology 79:2168-2179.

Tank, J. L., J. R. Webster, and E. F. Benfield. 1998. Effect of leaf litter exclusion on microbial enzyme activity associated with wood biofilms in streams. J. N. Am.m Benthol. Soc. 17:95-103.

Tarter, D. C. 1976. Limnology in West Virginia: a Lecture and Laboratory Manual. Huntington WV: Marshall Univ. Bookstore. 249 pp.

Tebo, L. B., Jr. 1955. Effects of siltation, resulting from improper logging, on the bottom fauna of a small trout stream in the southern Appalachians. Prog. Fish-Culturist 17:64-70.

Trimble, G. R., Jr. 1977. A history of the Fernow Experimental Forest and the Parsons Timber and Watershed Laboratory. General Technical Report NE-28. Forest Service, U.S. Department of Agriculture, Broomall, PA 46 pp.

Trimble, S. W. 1997. Contribution of stream channel erosion to sediment yield from an urbanizing watershed. Science 278:1442-1444.

United States Fish and Wildlife Service. 1998. Permitted stream losses due to valley filling in Kentucky, Pennsylvania, Virginia, and West Virginia: A partial inventory Pennsylvania Ecological Services Field Office, State College PA 12 pp.

Vannote, R. L., G. W. Minshall, K. W. Cummins, K. W., J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37:130-137.

Vannote, R. L., and B. W. Sweeney. 1980. Geographic analysis of thermal equilibria: a conceptual model for evaluating the effect of natural and modified thermal regimes on aquatic insect communities. Am. Nat. 115:667-695.

Waldron, M. C., and J. B. Wiley. 1996. Water quality and processes affecting dissolved oxygen concentrations in the Blackwater River, Canaan Valley, West Virginia. U.S. Geological Survey. Water-Resources Investigations Report 95-4142. Charleston, WV. 85 pp.

Wallace, J. B. T. F. Cuffney, J. R. Webster, G. J. Lugthart, K. Chung, and B. S. Woldowitz. 1991a. A five-year study of export of fine particulate organic matter from headwater streams: effects of season, extreme discharge, and invertebrate manipulation. Limnol. Oceanogr. 36: 670-682.

Wallace, J. B., S. L. Eggert, J. L. Meyer, and J. R. Webster. 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. Science 277:102-104.

Wallace, J. B., S. L. Eggert, J. L. Meyer, and J. R. Webster. 1999. Effects of resource limitation on a detrital-based ecosystem. Ecol. Monogr. 69:409-442.

Wallace, J. B. and M. E. Gurtz. 1986. Response of *Baetis* mayflies (Ephemeroptera) to catchment logging. Amer. Midl. Nat. 115: 25-41.

Wallace, J. B., and J. J. Hutchens. 2000. Effects of invertebrates in lotic ecosystem processes. pp. 73-96. *In:* D. C. Coleman and P. E. Hendrix (eds), Invertebrates as Webmasters in Ecosystems. CABI Publishing, Oxon, United Kingdom. 336 pp.

Wallace, J.B., A. D. Huryn, and G. J. Lugthart. 1991b. Colonization of a headwater stream during three years of seasonal insecticidal applications. Hydrobiologia 211: 54-76.

Wallace, J. B., G. J. Lugthart, T. F. Cuffney, and G. A. Schurr. 1989. The influence of repeated insecticidal treatments on drift and benthos of a headwater stream. Hydrobiologia 179:135-147.

Wallace, J. B., J. R. Webster, S. L. Eggert, J. L. Meyer and E. R. Siler. 2001. Large woody debris in a headwater stream: long-term legacies of forest disturbance. Int. Rev. Hydrobiol. 86:501:513.

Wallace, J. B., M. R. Whiles, S. Eggert, T. F. Cuffney, G. J. Lugthart, and K. Chung. 1995. Long-term dynamics of coarse particulate organic matter in three Appalachian Mountain streams. J. N. Am. Benthol. Soc. 14:217-232.

Wallace, J. B., J. R. Webster, and R. L. Lowe. 1992. High-gradient streams of the Appalachians. pp. 133-190. *In:* C. T. Hackney, S. Marshall Adams, and W. H. Martin (eds), Biodiversity of Southeastern United States, Aquatic Communities. John Wiley and Sons, New York, NY 779 pp.

Walser, C. A., and H. L. Bart. 1999. Influence of agriculture on in-stream habitat and fish community structure in Piedmont watersheds of the Chattahoochee River system. Ecol. Freshwat. Fish 8:237-246.

Wang, L. Z., J. Lyons, and P. Kanehl. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environ. Manage. 28:255-266.

Wang, L., J. Lyons, P. Kanehl, R. Bannerman, and E. Emmons. 2000. Watershed urbanization and changes in fish communities in southeaster Wisconsin streams. J. Am. Water Resour. Assoc. 36:1173-1189.

Ward, J. D., and P. A. Mistretta. 2002. Impact of pests on forest health. pp. 403-428. *In:* D. N. Wear and J. G. Greis, (eds) Southern forest resource assessment. Gen. Tech. Rep. SRS-53. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 635 pp.

Waters, T. F. 1995. Sediment in streams: source, biological effects, and control. American Fisheries Society, Bethesda, MD 251 pp.

Wear, D. N. 2002. Land use. pp. 153-173. *In:* D. N. Wear and J. G. Greis, (eds.) Southern forest resource assessment. Gen Tech. Rep SRS-53. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station. 635 pp.

Weaver, L. A., and G. C. Garman. 1994. Urbanization of a watershed and historical changes in a stream fish assemblage. Trans. Am. Fish. Soc. 123:162-172.

Webster, J. R., and S. W. Golladay. 1984. Seston transport in streams at Coweeta Hydrologic Laboratory, North Carolina, USA. Proc. Int. Assoc. Theor. Appl. Limnol. 22:1911-1919.

Webster J. R., M. E. Gurtz, J. J. Hains, J. L. Meyer, W. T. Swank, J. B. Waide, and J. B. Wallace. 1983. Stability of stream ecosystems. pp. 355-395. *In:* J. R. Barnes and G. W. Minshall (eds), Stream Ecology. Plenum Press, New York, NY 399 pp.

Webster, J. R., and J. L. Meyer (eds). 1997. Stream organic matter budgets. J. N. Am. Benthol. Soc 16:3-161.

Webster, J. R., J. L. Tank, J. B. Wallace, J. L. Meyer, S. L. Eggert, T. P. Ehrman, B. R. Ward, B. L. Bennett, P. F. Wagner, and M. E. McTammany, 2000. Effects of litter exclusion and wood removal on phosphorus and nitrogen retention in a forest stream. Proc. Int. Assoc. Theor. Appl. Limnol 27:1337-1340.

Webster, J. R. , J. B. Wallace, and E. F.Benfield. 1995. Organic processes in streams of the eastern United States. pp. 103-164. *In:* C. E. Cushing, G. W. Minshall, and K. W. Cummins (eds), River and stream ecosystems (Ecosystems of the World, vol. 22). Elsevier Science, Amsterdam. 817 pp.

Wemple, B. C., F. J. Swanson, and J. A. Jones. 2001. Forest roads and geomorphic process interactions, Cascade Range, Oregon. Earth Surf. Process. Landf. 26:191-204.

Whiles, M. R., and J. B. Wallace. 1992. First-year benthic recovery of a headwater stream following an insecticide-induced disturbance. Freshwat. Biol. 28:81-91.

Whitworth, W. E., and R. J. Strange. 1983. Growth and production of sympatric brook and rainbow trout in an Appalachian stream. Trans. Am. Fish. Soc. 112:469-475.

Winter, J. G., and H. C. Duthie. 2000. Export coefficient modeling to assess phosphorus loading in an urban watershed. J. Am. Water Resour. Assoc. 36:1053-1061.

Wipfli, M. S., and D. P. Gregovich. 2002. Export of invertebrates and detritus from fishless headwater streams in southeastern Alaska: implications for downstream salmonid production. Freshwater Biology 47:957-969.

Wipfli, M. S., J.Hudson, and J. Caouette. 1998. Influence of salmon carcasses on stream productivity: Response of biofilm and benthic macroinvertebrates in southeastern Alaska, U.S.A. Can. J. Fish. Aquat. Sci. 56:1600-1611.

Wolf, Ph. 1956. Utdikad Civilization. (Drained Civilization). Gleerups. Melmo, Sweden.