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The Functions and Importance of Forests, with Applications to the Croton and Catskill/Delaware Watersheds of New York

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Report to the Croton Watershed Clean Water Coalition

18 September 2008

Introduction

The Croton Watershed

A watershed is an area that naturally captures precipitation, which then drains and infiltrates into streams and reservoirs (Mehaffey et al. 2001). The Croton Watershed spans much of Westchester and Putnam Counties, and a small part of Dutchess County, covering nearly 400 square miles (ca. 1000 square kilometers) of southeastern New York (Slawecki et al. 2003). The watershed encompasses a network of 74 subbasins and 12 primary reservoirs, the drainage areas of which are referred to as the Croton Watershed Basins. Water from upstream reservoirs eventually flows into New Croton Reservoir, the terminal reservoir for the Croton Watershed (NYCDEP 2000, Moffett et al. 2003). The major waterways of the Croton Watershed are the West, Middle and East branches of the Croton River, Titicus River, Muscoot River, and Cross River.

The watershed straddles two distinct geologic regions: the Hudson Highlands in the north, characterized by rugged, rolling mountains formed in the Middle Proterozoic; and the Manhattan Prong in the south, consisting of rolling lowlands dating to the Early Paleozoic. The Hudson Highlands are aptly named, with elevations ranging from 240 m below sea level at the bottom of the Hudson River, to 405 m above sea level on North Mount Beacon. The greatest elevation of the Manhattan Prong directly south, by contrast, reaches only about 100 m above sea level. The rocks of the Hudson Highlands underlying the Croton Watershed were formed from the deposits of a shallow sea 1.3 billion years ago. These sedimentary rocks were later metamorphosed into gneiss containing large amounts of iron ore. The bedrock is composed of stratified (layered) and nonstratified metamorphic rocks that are highly resistant to erosion (Isachsen and Gates 1991, Fisher et al. 1995). Very shallow soils with substantial areas of exposed bedrock are characteristic surficial features of much of the Hudson Highlands in the northern part of the Croton Watershed, with local pockets of deeper soil. The surface of the Manhattan Prong in the southern portion of the watershed is closely controlled by the shape of the underlying bedrock. Erosion-resistant gneiss, schist, and quartzite underlie the hills, and less resistant marble underlies the valleys. The Manhattan Prong has deep soils in the valleys with shallow soils on the upper slopes of hills. The soils of both geologic regions are generally well drained and developed on glacial till (a heterogeneous mixture of stones, gravel, sand, silt and clay). Granitic material, schist and gneiss dominate the parent material of the acid brown earths (soils), which are very strongly leached, deeply weathered, acid, and low in fertility (Cline 1963, Cadwell et al. 1989).

There is little published information on the forests of the Hudson Highlands east of the Hudson River. At Black Rock Forest, in the Highlands just west of the river, the forest is mixed hardwood dominated by sugar maple (*Acer saccharum*), northern red oak (*Quercus rubra*), white oak (*Quercus alba*), black oak (*Quercus velutina*), bitternut hickory (*Carya cordiformis*), and white ash (*Fraxinus americana*) (Ross 1958). Hemlock (*Tsuga canadensis*) and black birch (*Betula lenta*) occupy steep ravines and north-facing slopes in the Highlands in association with red maple (*Acer rubrum*), sugar maple and white ash (Charney 1980). The upland forests of the Manhattan Prong are dominated by oaks, with a component of sugar maple, American beech (*Fagus grandifolia*), black birch, yellow birch (*Betula alleghaniensis*), and hemlock (Loeb 1987). Some notes on the vegetation of the eastern Hudson Highlands is in Appendix 1.

Between 1990 and 2000, the Croton Watershed experienced significant human population growth, with populations in some subbasins increasing as much as 40 percent (Moffett et al. 2003). The Croton Watershed was approximately 60% forest, 25% residential land, 7% agricultural land, 4% commercial land, and 6% lakes and reservoirs in 1999 (Linsey et al. 1999, Burns et al. 2005).

The Catskill/Delaware Watersheds

The Catskill/Delaware Watersheds cover roughly 1600 square miles (ca. 4100 square kilometers) of Delaware, Greene, Schoharie, Sullivan and Ulster counties, west of the Hudson River. True to its name, the Catskill/Delaware Watersheds are divided into two main water supply regions: the Delaware, encompassing the Cannonsville, Pepacton, Neversink and Rondout watersheds; and the Catskill, comprising the Ashokan and Schoharie Watersheds. Streams within the watersheds flow into reservoirs in the six contributing areas; the major streams feeding the water supply reservoirs are the East and West Delaware, Esopus, Neversink, Rondout, and Schoharie (Mehaffey et al. 2001).

The Catskill/Delaware Watersheds are situated on the Allegheny Plateau, composed of sandstone and shale layers formed in the Middle and Late Devonian during the Acadian Orogeny when sediments from the Acadian Mountains were deposited in a shallow inland sea. The sediments eventually displaced the sea, forming the Catskill Delta. The northeastern section of the watershed is underlain by conglomerate and cross-bedded sandstone layers interbedded with shale layers. The rocks of this formation crown the escarpment that divides the eastern and northeastern edges of the watershed (CCCD 2001). The Allegheny Plateau is relatively high and rugged, with elevations reaching 1281 m above sea level on Slide Mountain (Rickard 1991). Soils in most of the watershed are well-drained acid brown earths with parent material largely of mixed siltstone, shale and sandstone, and lie on glacial till. These soils are very strongly leached, acid, and low in fertility. Soils developed on Recent alluvium are found where rivers in flood overflow their banks, leaving new sediment annually. Parent material of the fertile, well-drained alluvial soils in the Catskill/Delaware Watershed is generally composed of shale and sandstone (Cline 1963).

The Catskill/Delaware Watersheds are primarily forested, with an average of 92% forest cover in the individual watersheds that comprise the system (Mehaffey et al. 2003). Elevations below approximately 500 m are typically mixed oak forests, of which the dominant species is northern red oak. Forests of the high elevation (500 - 1100 m) areas of the Catskills are characteristically northern hardwood forest similar to those throughout the Northeast, dominated by sugar maple, American beech, and yellow birch, with eastern hemlock restricted to ravines and cool, north-facing slopes. Some very high elevation areas (> 1100 m) are dominated by balsam fir (*Abies balsamea*), red spruce (*Picea rubens*), and paper birch (*Betula papyrifera* var. *cordifolia*) (Lovett et al. 2000).

Active land use in the Catskill/Delaware Watershed makes up a small portion of the total area. Agriculture accounts for an average of 10% of the watersheds, while urban areas are a less important land use, accounting for an average of < 1% of the watersheds. Cannonsville, Schoharie and Pepacton basins have the highest concentration of agricultural use among the Catskill/Delaware Watersheds, mainly in the form of pastures for livestock and hay production in the northwest (Mehaffey et al. 2001, 2003).

Together, the Croton and Catskill/Delaware Watersheds supply New York City and surrounding counties with 1.3 billion gallons of drinking water each day. The Croton Watershed, the oldest of New York City's surface water systems, supplies approximately 10 percent of New York City's water, and the Catskill/Delaware Watersheds account for the remaining 90 percent (Moffett et al. 2003). Water from the Croton Watershed requires filtration before being consumed by New York residents because, according to the Environmental Protection Agency, urban development and high growth rates in the watershed would overwhelm watershed management aimed at protecting drinking water. Water from the Catskill/Delaware Watersheds is under a conditional exemption contingent upon the maintenance of watershed conditions that provide high quality drinking water (Brown 2000).

Water quality, biodiversity, and ecological health of freshwater systems depend on the function of the fine branches of headwater streams. The contributions of forests to watershed protection, as well as wildlife conservation and carbon sequestration, have been widely recognized. Despite these recognized ecosystem services, there is little legal protection for forests in the northeastern United States, and they continue to yield to urban development. Small headwater streams account for more than 70 percent of stream-channel length in the U.S., yet are often missing from maps that direct management of surrounding areas (Lowe and Likens 2005). Between 1982 and 1997, 4.2 million hectares of nonfederal forest land in the United States were developed, and nearly half of this loss occurred from 1992 to 1997 (Alig et al. 2003). The loss of forests to urban development is predicted to continue in the Northeast (Stein et al. 2005) as long as a higher value is placed on urban development than on the ecosystem services of intact forests.

The Functions of Trees in Watersheds

Forests provide a number of biotic and abiotic services to watersheds, moderating minimum and maximum stream flows and flooding, controlling surface runoff and erosion, buffering against atmospheric and terrestrial pollutants, preventing siltation and eutrophication of waterways, preserving drinking water quality, and providing unique habitat for a number of useful, rare and endangered species. The purpose of this synthesis is to review the functions of forests and their influences on environmental and resource quality with applications to the Croton and Catskill/Delaware Watersheds, so that these functions, and the effect of their removal, may be considered in light of increasing development pressure.

Surface Runoff, Flooding and Stream Flow

Precipitation travels to stream channels within a watershed in two major forms: as surface flow (runoff) and subsurface flow (groundwater). Unnaturally large amounts of runoff due to deforestation or replacement of the forest with impervious surfaces (i.e. pavement, roofs) are quickly transported to streams on the surface of the soil, leading to rapid concentration of water in stream channels and increased flood crests, as well as a suite of other physical disturbances. The moderating effect of forests on surface runoff is partly due to the surface area of the vegetation. Larger species (i.e. trees) intercept rainfall and slow runoff better than, for example, grasses, bare ground (Kittredge 1948), or impervious surfaces (Burns et al. 2005, Gaister et al. 2006), by increasing the amount of time it takes for rainfall to travel to stream channels.

In addition to providing mechanical obstruction, trees modify soils to promote groundwater flow, thereby reducing surface runoff. Organic matter within the soil, as well as pores formed by roots and soil fauna cause forest soil to be highly permeable (Kittredge 1948), allowing for higher rates of infiltration. Upland soils in northeastern forests are usually unsaturated because evapotranspiration and infiltration under most conditions reduce soil saturation during the growing season (de la Cretaz and Barten 2007). Both high infiltration capacity and slow subsurface flow in forests are the main mechanisms by which streamflow is sustained through the dry season (Heisig 2000, de la Cretaz and Barten 2007). Throughout the growing season, however, forests reduce water runoff relative to deforested areas because of water loss to transpiration (loss of water from within plants to the air).. By these functions, forests moderate stream flooding within watersheds and downstream water bodies, and can also increase the flow of streams during dry periods.

Classic studies have shown direct links between deforestation and increased surface runoff in ecosystems of the northeastern U.S. (Schneider and Ayer 1961, Eschner 1965, Hibbert 1967, Likens et al. 1970, Bormann et al. 1974, McHale et al. 2008) that resemble those of the Croton and Catskill/Delware Watersheds. In a famous study at Hubbard Brook, a northern hardwood forest in New England, vegetation was experimentally removed in a watershed and the effects of deforestation were compared to an intact control watershed nearby. Runoff increased 41% in the first year following deforestation, and remained abnormally high through the following three years. Stream flow increased 39% and also remained high throughout the study, increasing by as much as 414% in a season (Likens et al. 1970). During the growing season, transpiration normally equals or exceeds rainfall in an undisturbed forest; however, in the deforested watershed, the soil remained relatively wet, leaving little available water storage. The lack of forest canopy also led to an increase in the rate of snowmelt following deforestation (Bormann et al. 1974). In times of heavy rainfall, the increase in runoff in deforested areas could result in increased flooding downstream.

Because of anthropogenic changes to forest surfaces, soils can temporarily become saturated with water during storms, reducing the ability of the forest to minimize overland flow and slow rates of subsurface flow during periods of heavy rainfall. The urbanization of the Croton Watershed has led to an increase in surfaces impervious to water (e.g. pavement, roofs). A recent study examined runoff in heavily urbanized, moderately urbanized, and undeveloped areas of the Croton Watershed (Burns et al. 2005). Their study found that peak flows increased and recession times decreased with increasing urbanization; however, the remnant forests within the watershed played a role in moderating the responses, reducing flood risk and stream flow variability (Burns et al. 2005).

Erosion

The accelerated rate of soil erosion has been a concern in the United States since the 1930s, with experts suggesting that the soil erosion rate amounts to an environmental crisis (Steiner 1990, Pimentel et al. 1995). Much of the nation’s erosion is taking place in croplands, where native vegetation has been removed and the soil is frequently disturbed

(Trimble and Crosson 2000). Mature northeastern forests are relatively erosion-resistant ecosystems, with streams transporting between 1.4 and 9.8 kilograms of particulate matter per hectare per year (Bormann et al. 1969, 1974); however, urbanization of forested areas affects the processes and rates of erosion (Franco et al. 2008). The rate of erosion during the construction stage of development is particularly high, reaching 1.63×10^5 kilograms of particulate matter exported by streams per hectare per year (Krenitsky et al. 1998).

In forests, the rate of erosion is controlled by minimizing runoff and sheltering and stabilizing soil. The cutting and carrying power of water is disproportionately associated with its velocity (Kittredge 1948); therefore the rate of erosion may increase strikingly with a small increase in runoff. The tendency of forests to convert surface flow to subsurface flow and reduce runoff, as described above, is even more important when one considers that a small change in runoff has a large effect on erosion. Decreased runoff further reduces erosion by minimizing peak stream flow, thereby preserving soil lining stream banks (Gaister et al. 2006).

The presence of forested uplands curtails erosion because the soil below is sheltered and stabilized. Erosion increases as much as twelvefold as the size and speed of raindrops increase (Laws 1940); the canopy and litter layer provide the soil below with significant protection from the erosive energy of falling raindrops. Furthermore, roots of trees and associated vegetation not only increase soil permeability and thus decrease runoff, but also stabilize the soil itself. Stream banks are stabilized by roots of riparian vegetation and shielded by hardwood leaf litter from erosion caused by stream flow (Bormann et al. 1969, Gaister et al. 2006). Large-scale bank erosion, a major contribution to turbidity in urban watersheds (Trimble 1997), is 30 times more prevalent on banks lacking vegetation than those with vegetated banks (Naiman and Decamps 1997).

The consequences of deforestation are ubiquitous, transcending continents and ecosystems, and often varying only in degree. In the northeastern forest, experimental removal of aboveground vegetation, leaving the soil, litter and roots undisturbed, initially only doubled or tripled the erosion rate for the first two years following deforestation. During this time, roots and aboveground litter remained intact, providing a physical barrier against erosion. In the third and fourth years following removal however, soil stability deteriorated and the rate of erosion increased exponentially (Bormann et al. 1974). The absence of, for example, transpiration to reduce soil wetness and vegetation to act as a physical barrier, led to rapid disintegration of soil stability and the processes that protect the ecosystem from erosion. These consequences have been observed in a variety of disturbed ecosystems (Iwata et al. 2003, An et al. 2008, Cebecauer and Hofierka 2008, Schiettecatte et al. 2008). In the Croton and Catskill/Delaware Watersheds, tree loss and excessive erosion may lead (and this has been shown experimentally in the Catskill Watershed; McHale et al. 2008) to upland nutrient loss, lowered water quality and degraded aquatic habitat as a result of increased turbidity and eutrophication.

Nutrient Cycling and Pollutant Buffering

Within the forest ecosystem, nutrients may be found in living and dead organisms, dissolved in the soil water, or as components of soil and rock minerals. Nutrients cycle between these compartments within the ecosystem by the decomposition of organic matter, nutrient uptake, or mineral weathering. Allochthonous (external) nutrients enter the forest ecosystem from the larger biosphere through biotic and abiotic processes. Watershed forests harbor large amounts of conservatively cycled nutrients. For example, calcium, a macronutrient important in soil reactions and necessary for plant growth, can be found in forms available to plants in mineral horizons, in the forest floor, and in trees and roots. Calcium is delivered to the system via precipitation, dry deposition, and bedrock and subsoil weathering, and exits via streamflow. Studies of deforestation in the northeastern U.S. showed that forest harvesting affects calcium capital in two ways: calcium is removed from the watershed in the form of wood, and, because deforested sites are less efficient than mature forests at cycling nutrients, calcium is subsequently leached into streams and groundwater (Hornbeck and Kropelin 1982, Smith et al. 1986, Tritton et al. 1987). The disruption of the nutrient cycle by the removal of vegetation leads to large fluxes of nutrients into streams and waterways, and effects on downstream water quality and ecosystem function (Hornbeck and Swank 1992).

McHale et al. (2008) documented the effect of tree harvesting on stream water chemistry in the Neversink Reservoir catchment of the Catskill Watershed. Deforestation caused the release of significant quantities of nitrate from decaying organic matter, causing the release of inorganic aluminum from the soil, and leading to a large increase of nitrate and inorganic aluminum in stream water. The change in water chemistry caused 100 percent mortality of caged brook trout (*Salvelinus fontinalis*) during the first year, and harmed macroinvertebrate communities for two years after the harvest (McHale et al. 2008). In the Hubbard Brook ecosystem, deforestation resulted in the immediate solubilization and mobilization of nutrients. Increased runoff and discharge quickly transported nutrients to streams, leading to dissolved nutrient levels several times greater than those of the undisturbed watershed. As a result, large amounts were exported from the watershed via stream channels. Nitrate, primarily from decaying organic matter, increased 56-fold in streamwater, exceeding drinking water standards and leading to dense algal blooms for many years following deforestation (Likens et al. 1970, Bormann et al. 1974, Vitousek et al. 1979).

Interestingly, disturbances in nutrient cycling are buffered in healthy, forested watersheds. An ice storm that caused extensive tree damage along an area of forested watershed resulted in elevated nitrate levels within a northeastern stream; however, in-stream processing of nitrate by stream microorganisms was accelerated by increased light availability, effectively acting as a negative feedback mechanism for the nutrient cycle (Bernhardt et al. 2003). One study tested the capacity of an upland mixed hardwood forest to retain inputs of nitrogen, and found that 85-99 percent of added nitrogen was retained in the form of increased plant uptake and litter accumulation, and as organic forms of nitrogen in the soil (Magill et al. 1997). The effect of clearcutting on stream water chemistry in the Catskill Watershed was buffered to a degree by the acid-neutralizing, high base-cation concentration in ground water; however this neutralizing capacity was overwhelmed by the magnitude of released nitrate as a result of the disturbance. Conversely, nitrate did not increase in streams flowing from less disturbed (selectively harvested) catchments within the Catskill Watershed (McHale et al. 2008), where groundwater buffering was presumably sufficient. Studies in the Catskill Watershed also show that the amount of retained nitrogen can vary depending on the type of tree species dominating the watershed forest (Lovett et al. 2000, 2002). Another study found that ammonium uptake in streams was greater along forested reaches as a result of the tendency of riparian vegetation to preserve stream width, and thus the nutrient-processing benthic microbial communities and nutrient-absorbing benthic habitat therein (Sweeney et al. 2004).

Forest soils are also highly effective at intercepting nutrients that would otherwise end up in downstream waters. Water penetrates forest soils as dilute rainwater, snowmelt, or throughfall, and is taken up by plants, recharges ground water, or drains to streams. Interactions with the soil, largely in the form of exchange reactions with ions bound to soil particles, change the chemistry of soil water; therefore soil conditions are reflected in water quality (McHale et al. 2008). Runoff that infiltrates into the soil and root zone also contacts microbial communities, resulting in sorption of pollutants to soil mineral and organic matter and uptake by microbes and macrophytes in the soil. A study demonstrated the attenuation of 80% of nitrate and 74% of phosphate in storm runoff after exposure to riparian soils (Casey and Klaine 2001). The associated rhizoidal bacteria of birch (*Betula* sp.) are capable of degrading polyaromatic hydrocarbons (toxic, mutagenic and carcinogenic substances produced by incomplete combustion of organic compounds) in the soil (Tervahauta et al. 2008). Other soil bacteria, such as *Pseudomonas* sp., *Ralstonia basilensis*, *Pseudaminobacter* sp., and *Chelatobacter heintzii* are able to completely mineralize atrazine, a commonly used herbicide, into carbon dioxide (Monard et al. 2008). The effectiveness of soil and soil microbes in attenuating pollutants is dependent on the infiltration of runoff into soils where pollutants contact these soil communities (Casey and Klaine 2001). Therefore the restriction of infiltration by removal of aboveground vegetation and increase in impervious surfaces results in more pollution traveling to streams and reservoirs.

At a local level, forests are useful in trapping air pollutants and wind-borne nutrients in urban environments (Weathers et al. 2001), but likely more important, trees in the northern hemisphere are significant for carbon storage, and thus land-use changes in forests affect global climate change (Dale 1997, Goodale et al. 2002). Although the combustion of fossil fuels is responsible for the majority of the increase in global atmospheric carbon dioxide, deforestation (the majority of which occurs in the tropics) accounts for nearly 25% of the increase (Denman et al. 2007). Trees in New York are estimated to store approximately 24.6 million metric tons (tonnes) (24.3 tonnes per hectare) of carbon, and sequester an additional 0.8 million tonnes per year (0.8 tonnes per hectare per year) from the atmosphere (Nowak and Crane 2002). Deforestation in New York not only results in the release of stored carbon into the atmosphere, but also the loss of future stored carbon in the form of lost sequestration capacity of the forest. Conversion of forest land to other land-use activities, such as pastures, results in less carbon storage than occurs in forests (Kirby and Potvin 2007, Sheikh et al. 2008).

Ultraviolet Radiation and Temperature

Riparian forest canopies help stabilize aquatic ecosystem functions by controlling inputs of solar energy and ultraviolet radiation. Alteration of the riparian canopy through urban development or other land-use conversion may lead to increased exposure of streams to sunlight and ultraviolet radiation, and to increases in temperature, resulting in changes in the composition of aquatic communities. Compared to heavily shaded sites, areas of streams with reduced canopy had greater algal accrual (Likens et al. 1970, Bernhardt et al. 2003) and decreased macroinvertebrate biomass and diversity (Kelly et al. 2003, Sweeney et al. 2004). In small areas of healthy streams with reduced canopy, algal accrual is thought to be an indirect result of the negative effects of greater ultraviolet radiation on invertebrate grazers (Kelly et al. 2003); the combined effects of decreased grazing from invertebrates and increased nutrient input as a result of large-scale deforestation may lead to a noxious bloom (Likens et al. 1970, Schneider and Ponius 2001, King et al. 2007, Martins et al. 2008). Spikes in temperature caused by large solar energy inputs affect the rates of metabolism, growth, decomposition, and solubility of gases, and are associated with increases in fish mortality and prevalence of disease (Johnson and Jones 2000). The global declines of amphibians have been linked to increased ultraviolet radiation (Blaustein et al. 1995, Alford and Richards 1999). Salamander larvae may be more vulnerable to ultraviolet radiation than frog larvae (Bancroft et al. 2008), and several species of salamanders that occur in the New York City watersheds have stream-dwelling larvae. These findings highlight the importance of healthy riparian ecosystems in maintaining species diversity as well as watershed quality.

Siltation and Eutrophication of Surface Waters

Environmental changes in the headwaters of a river basin inevitably affect areas downstream (Haigh et al. 2004, Lowe and Likens 2005). An important consequence of deforestation in urbanized watersheds, particularly those that provide drinking water for major metropolitan areas, is the diminution of water quality from increased turbidity and eutrophication. Erosion and soil export in storm runoff, processes tightly controlled in healthy forests, are the major causes of stream siltation in watersheds (Trimble 1997). Sediment deposition in reservoirs, such as those in the Croton and Catskill/Delaware Watersheds, reduces reservoir volume and duration of use, causes deterioration of water quality, reduces flood control capability, and even augments breeding areas for mosquitoes (Haigh et al. 2004). Increased siltation in streams may result in suffocation of aquatic wildlife and abrasion of habitats (Lorang and Aggett 2005).

Eutrophication of surface water bodies constitutes one of the most serious water quality problems, affecting stream chemical and biological characteristics. Nutrient input from land conversion, agriculture and urbanization is a major contributor (Schneider and Ponius 2001, King et al. 2007), particularly following periods of intense precipitation when nutrients are transported to stream channels and ultimately into downstream water bodies. Phytoplankton and cyanobacteria (blue-green algae) thrive in the high nutrient environment, forming dense blooms and releasing cyanotoxins, thus affecting multiple water uses (Martins et al. 2008) and harming submerged vascular plants, fish and waterfowl (Howarth et al. 1991). Eutrophication can also occur as a result of deforestation in otherwise undeveloped watersheds; deforestation at Hubbard Brook resulted in significant pollution of the watershed's drainage stream and the formation of dense algal blooms for several years (Likens et al. 1970). Intact, forested watersheds are somewhat resilient to nutrient inputs (Bernhardt et al. 2003), and as described above, are useful in buffering against nonpoint source pollution (Lowrance et al. 1997, USDA 2002, Sweeney et al. 2004).

Habitat Quality and Biodiversity

Healthy aquatic and terrestrial habitats and biodiversity are intrinsic to the ability of a watershed to maintain water quality. Watersheds contain the interfaces between terrestrial and aquatic ecosystems. Because each habitat harbors unique species, the combination of upland, riparian, and aquatic habitats within watersheds accommodates significant species biodiversity (Sabo et al. 2005).

Deforestation and disturbance have negative repercussions on a number of habitat-improving services provided by trees. Trees reduce noise in habitats adjoining urban areas (Bernatzky 1982), thereby reducing disturbance to wildlife. The abundance of invasive plants is low within mature forests, but tends to be higher near forest edges and disturbed areas. This pattern suggests that invasives often colonize forest edges before spreading to forest interiors, and therefore that fragmentation compromises the ability of forests to resist invasion by exotic species (Luken and Goessling 1995, Cadenasso and Pickett 2001, Hunter and Mattice 2002).

Macroinvertebrates and other animals of the benthic community rely on the health of watersheds (Kelly et al. 2003, Sweeney et al. 2004, McHale et al. 2008), along with fish assemblages (Jones et al. 1999, McHale et al. 2008), waterfowl and other birds (Brady 1989, Johnson and Brown 1990, Keller et al. 1993, Hodges and Krementz 1996), river otters (*Lutra canadensis*; Brady 1989, DeGraaf and Yamasaki 2001), and many other rare and common animal and plant species. Breeding colonies of the Indiana bat (*Myotis sodalis*), a federally listed endangered species found in New York (NYSDEC 2008), most often utilize the underside of exfoliating bark of dead or dying trees (Carter 2006, Watrous et al. 2006). Breeding females and roosting males also often select live shagbark hickory (*Carya ovata*) trees with their characteristic exfoliating bark, or larger black locust (*Robinia pseudoacadia*) trees with deeply furrowed bark (Robyn Niver, U.S. Fish and Wildlife Service, personal communication). The Indiana bat also roosts in a number of other tree species in the Croton and Catskill/Delaware Watersheds, including bitternut hickory, northern red oak, white oak and sugar maple (USFWS 2007). The density of large snags that provide roosting habitat affects the abundance and distribution of Indiana bats (Yates and Muzika 2006), therefore loss of dead (and living) trees within the Croton and Catskill/Delaware Watersheds would result in the loss of important habitat for the endangered Indiana bat as well as a number of other important species. Other organisms that depend on forest trees and are likely to be negatively affected by forest loss in the New York City watersheds include several other species of bats as well as tree squirrels, flying squirrels (*Glaucomys* spp.), porcupine (*Erithizon dorasatum*), fisher (*Martes pennanti*), bobcat (*Lynx rufus*), wood duck (*Aix sponsa*), ruffed grouse (*Bonasa umbellus*), most of the hawks and owls and many songbirds, gray treefrog (*Hyla versicolor*), spring peeper (*Pseudacris crucifer*), mole salamanders (*Ambystoma* spp.), dusky salamanders (*Desmognathus* spp.), spring salamander (*Gyrinophilus porphyriticus*), brook trout (*Salvelinus fontinalis*), many other stream fishes, a large number of invertebrate species, and many forest wildflowers, sedges, mosses, liverworts, lichens, and fungi.

Summary and Conclusions

Forests in the watersheds of the northeastern U.S. are critical to protecting water quality of drainage streams and reservoirs. Trees reduce surface runoff by creating soil channels with roots and reducing soil wetness by transpiration, allowing precipitation to infiltrate into the ground, thus converting surface flow to subsurface flow. This function tends to decrease maximum stream flow in wet weather and increase minimum stream flow in dry weather, moderating floods and droughts. Excessive erosion and soil loss are prevented by controlled runoff, the sheltering of exposed soils by litter and vegetation, and the soil-stabilizing effect of tree roots. Nutrients are conserved and potential pollutants, including atmospheric carbon, are sequestered by the processes that cycle nutrients between the soil, litter, vegetation, and larger biosphere. The processes that reduce runoff, soil erosion and nutrient loss, and buffer against pollutants, prevent turbidity and eutrophication of streams and reservoirs, thus protecting water quality. Healthy forests also provide habitat for important wildlife such as the endangered Indiana bat.

Several characteristics of the New York City watersheds should be kept in mind when considering the role of forests in protecting water quality and biological diversity, and providing other ecosystem services. There are large areas with steep slopes, exposed bedrock, shallow soils, and low soil fertility. These environments will be especially vulnerable to erosion if deforested, and particularly hard to restore. In addition to upland forest loss, the watersheds are undergoing loss of intact wetlands and riparian habitats, which are important in conserving stream water flows and quality. The forests in this region are threatened by air pollution, introduced insects and pathogens, and climate change, making them susceptible to decline and less resistant to other stresses. Streams in the watersheds are also under stress from other factors such as pollutants from sewage, local industrial pollution, pesticides in runoff and wind drift, and atmospheric deposition of nitrogen and acidity. In order to protect streams and water, all these problems must be addressed. The beneficial influences of forests are especially important in watersheds where forests are under a high degree of threat.

Acknowledgements

We thank Gary M. Lovett for helpful comments on the manuscript.

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Appendix 1

Notes on Vegetation in the Hudson Highlands East of the Hudson River

by James G. Barbour (Hudsonia)

The lower slope forests (mostly south and west-facing) along the Hudson River south of the Bear Mountain Bridge tend to have a lot of tuliptree compared to the similar slopes of the western Highlands. These eastern slope forests do not fit the New York Natural Heritage Program (Reschke 1990, Edinger et al. 2002) description of "oak-tuliptree forest." Dominant trees are tuliptree, sugar maple and northern red oak. Witch hazel is present, and in moist soils spicebush, but mountain laurel is absent or scarce. Of course these forest types vary a lot due to quick changes of aspect, elevation and drainage.

Middle slopes have forests similar to those in the western Highlands, except they are more "generic," not fitting NHP categories. Some are blends of Appalachian oak-hickory and chestnut oak forest, but there is also what I call "oak-maple forest," mostly northern red oak, with sugar maple, red maple, or both, a very common mid-slope forest type, usually with very sparse shrub layers: maple-leaf viburnum, witch hazel, a few mountain laurel.

Ridges in the eastern Highlands seem to me drier and less densely vegetated, more stressed? This may be due to the western exposure along the river. Farther east of the river the ridges are more densely vegetated, partly because the topography is less rugged, with lower relief. Ridge flora is similar to that of the western Highlands: scrub oak, sweetfern, stunted white, chestnut and red oaks, fewer black birch, the usual heaths.

One important difference is more invasive plants in the eastern Highlands, especially Westchester County, where the understories are often nearly devoid of native herbs or even shrubs. Garlic mustard is nearly everywhere. Bell's bush honeysuckle (*Lonicera* X *bella*) and winged spindlebush (*Euonymus alatus*) are common exotic understory shrubs in mature Westchester forests.

Among rare plants cattail sedge (*Carex typhina*) occurs in ridgetop woodland pools east of the Hudson, but I know of no records for it west of the Hudson. However, the balance of rare plant occurrences is tilted toward the west, with small-flowered crowfoot (*Ranunculus micranthus*) S3 R, violet wood sorrel (*Oxalis violacea*) S2 T, black-edged sedge (*Carex nigromarginata*) S1 E, reflexed sedge (*C. retroflexa*) S2S3 R, glaucous sedge (*C. glaucodea*) S2S3 T and Wildenow's sedge (*C. wildenowii*) S3 R all known from the western, but not the eastern Highlands. This may be due in part to more botanical surveying west of the river. Virginia snakeroot (*Aristolochia serpentaria*) S1 E, for example, was known only from the western Highlands until discovered in Westchester in 2006.