

3 Principles Guiding Watershed Management

The science of watershed management continues to evolve, although many basic principles have been long-established and are now widely accepted as the basis for informed management. It is the intent of the Division to constantly review the literature and stay on top of research developments that impact management decisions. The purpose of this section is to describe current principles in the general areas of watershed protection, water yield and quality maintenance, the value of forest cover for water protection, the impact of a wide variety of disturbances on the delivery of high quality water from forested watersheds, and the role of active management in developing resistance and resilience in the water supply protection forest.

3.1 General Watershed Management Principles

3.1.1 Watershed Protection

- Forested watersheds generally yield higher quality water than non-forested cover types. Most urban, suburban and agricultural land uses contribute in some way to lowered water quality.
- Uncontrolled human activities on water supply watersheds represent a major source of potential contamination. Efficient and effective water quality protection on both filtered and unfiltered water supplies requires control over human activities.
- Watershed cover conditions differ in their regulation of certain nutrients (e.g., nitrates, phosphates). Within the variety of watershed land cover types, the best regulation of nutrients is provided by maintaining vigorously growing forest across the vast majority of watershed sites. Forests developed through silvicultural methods that include the range from single-tree to small group and patch regeneration cutting will include a range of size and age classes, as well as a mix of species across the continuum from shade tolerant to shade intolerant.
- Fire protection, watershed ranger and police surveillance, water sampling, and other watershed management activities, including forest management, all depend upon an adequate, well-maintained watershed road system. Poorly designed or inadequately maintained roads represent the greatest potential source of sediment inputs to tributaries on undeveloped watersheds.
- The proper management and protection of wetland and riparian zones is a critical component of watershed protection, in part because these frequently are concentrated water supply source areas and because they represent the final opportunity to capture mobile sediments/nutrients before they enter surface waters.

3.1.2 Water Yield

- Water yields are influenced by precipitation amounts, site conditions (such as slope, aspect, and soils) and the intensity and type of watershed cover management.
- Water yields are affected directly by evapotranspiration rates of the watershed cover. Therefore, management activities that result in decreased evapotranspiration also result in increased water yield, while those that increase evapotranspiration decrease water yield.

- Intensive, even-aged management of forested watersheds provides consistently greater water yields than uneven-aged management, multi-aged management or the absence of active management.
- Water yields decrease as young forests grow. As forests become more open, water yields increase. When watershed forests are disturbed, water yields initially increase. As disturbed areas fill with young forests, water yields decrease.
- Paired watershed experiments across many regions have demonstrated that until approximately 20-30% of the forest is cut, there is generally no measurable increase in water yield, and furthermore that water yield generally returns to or below the pre harvest baseline within 3-10 years unless there is a significant change in species composition (e.g., from deciduous species to evergreens), in which case some degree of change in yield will persist indefinitely.

3.1.3 Water Quality

- Surface waters collected from fully vegetated watersheds with minimal exposed soils generally carry very low turbidity.
- Bacteria counts in surface waters may or may not be buffered by vegetated land cover, depending on the sources of bacteria present and their contact with these waters.
- Critical protection of water quality for predominantly forested and actively managed watersheds includes the following principles:
 - Minimizing land use/land cover changes in order to maintain forest cover across the majority of the watershed provides the most effective primary barrier for protecting tributary and reservoir water from pollutants.
 - In actively managed forests, Conservation Management Practices, correctly designed and applied effectively will protect water sources from sediment/nutrient losses otherwise associated with forest management activities.
 - The most common sources of water quality degradation by timber harvesting are intersections between harvesting roads or staging areas and water sources. Disconnecting roads/staging areas from water sources prevents water quality degradation.
 - To prevent contamination of surface and ground waters, petroleum products on water supply watersheds must be tightly contained or replaced with biodegradable alternatives.
 - Maintaining a species and age/size-diverse forest cover may increase that cover's resistance to disturbance and ability to recover quickly when disturbance occurs. Active management can increase size and species diversity where past land use or natural disturbances have homogenized the forest cover.

3.1.4 Unfiltered Water Supply Regulatory Requirements

Unfiltered surface water supplies are mainly regulated under two EPA promulgated rules: the Long Term Surface Water Treatment Rule (LT2) and the Stage 2 Disinfectants and Disinfection Byproducts Rule (Stage 2). Both rules were released by EPA in January, 2006.

In order to maintain its status as an unfiltered water supply system, the Quabbin Reservoir must meet the main aspects of the LT2, which are:

- Non-filtered water supplies must maintain a watershed control program. A well managed forestry program is an integral part of any watershed management program.
- Turbidity levels must not exceed 5 Nephelometric Turbidity Units (NTU). A well managed forest over the watershed prevents erosion and ensures that turbidity levels as measured at the CVA average under 0.5 NTU.
- Fecal coliform concentrations must be less 20/100 ml prior to disinfection in 90% of samples taken during any consecutive six month period. At Quabbin, MWRA meets all regulatory standards although the source water criteria require the implementation of an active seagull harassment program starting in late fall and often through early spring.
- MWRA will be required to conduct two years of source water monitoring at Quabbin for the presence of *Cryptosporidium* oocysts. All the results will be averaged, and if the mean is less than 0.01 oocysts/L, MWRA will be required to provide at least 2-log *Cryptosporidium* inactivation. If the mean is greater than 0.01 oocysts/L, MWRA will then be required to provide at least 3-log *Cryptosporidium* inactivation. Based on data collected over several years indicate that the mean will most likely be less than 0.01 oocysts/L (Lasky, 2006). The existence of an active beaver and muskrat exclusion program over the Pathogen Free Zone on the Winsor Dam Basin is one of the main reasons for the low mean.
- In order to meet the water quality standards of Stage 2, source water has to meet the following standards with regard to disinfection by-products (DBPs): Average maximum contaminant levels (MCLs) for trihalomethane (THM) of 80 parts per billion (ppb) and 60 ppb for haloacetic acid (HAA). Current levels are in the 4 – 12 ppb range for both. The following is background information on DBPs (Garvey *et al.*, 2001):
 - DBPs are formed by reactions between oxidants, usually chlorine based, and organic compounds.
 - Natural organic matter (NOM) in the water is composed of both dissolved and particulate organic substances, and can originate on land (allochthonous) or within the aquatic system (autochthonous).
 - NOM is measured as total organic carbon (TOC). TOC is composed of dissolved organic carbon (DOC) and of particulate organic carbon (POC). DOC normally accounts for approximately 90% of TOC. DOC can be a precursor for DBPs.
 - Planktonic algae account for the most significant percentage of autochthonous inputs. Most DOC in Quabbin is autochthonous.
 - DOC is directly related to the eutrophic state of a water body. Eutrophication is related to the amount of nutrients, mainly nitrogen and phosphorous, available for algal growth, with phosphorous usually the most limiting. Eutrophication is driven by external loading of organic matter, nutrients and silt. A well managed, forested watershed sequesters nutrients.

3.2 Basic Principles of Forest Hydrology

Hydrology is the scientific study of the liquid and frozen waters of the earth, their properties, circulation, and distribution on and under the Earth's surface and in the atmosphere, from the moment of precipitation until these waters are returned to the atmosphere through evapotranspiration or are discharged into the ocean. Forest hydrology is more specifically the study of the circulation of water into and through forested lands, and details the effects of the forest on water at scales ranging from a single tree to a partially or fully forested watershed. Among the most critical investigations in forest hydrology is the impact of the forest on water yield, the difference in the amount of water that arrives from the atmosphere via precipitation and the amount that ultimately leaves the watershed via draining tributaries.

Increasingly, as drinking water supplies become more critically limiting, the relationship between forest cover and water quality has become a central topic in forest hydrology, in part because of the links between water yield and the yield of sediments and nutrients to water supplies.

3.2.1 Forests and Water Yield

- Evergreens (generally conifers) use more water than deciduous species over the course of a year, in part because they continue to transpire after deciduous trees have dropped their leaves.
- Generally, where precipitation exceeds potential evapotranspiration ($PET = \text{the amount of water that could be removed by a crop of short vegetation in given conditions of heat and wind energy}$), forests can grow. Where forests are removed in whole or in part from these areas, actual ET decreases and yield increases. The water balance is generally expressed as $P - ET - Q \pm \text{change in } S \pm L = 0$, where P =precipitation; ET =evapotranspiration; Q = water yield; S =storage, and L =leakage. Decades of forest hydrology research have shown that for many forests, removal of forest cover must reach 20-30% before increased yields are detectable.
- As much as 20-30% of rain and snow falling on a given forest is intercepted by forest cover, and a portion of this evaporates before reaching the ground, a portion runs down the leaves, branches, and stems to reach the ground, and a portion accumulates to larger drops and falls toward the ground from the canopy.

3.2.2 Forests and Water Quality

- No other land cover has been shown to protect the quality of drinking water better than forest cover.
- The accumulation of organic matter, the growth of fine and coarse roots, the actions of soil-dwelling microbes, invertebrates and vertebrates, and other natural processes develop properties of infiltration, hydraulic conductivity, and water storage (porosity) that are unique to forest soils and contribute to the protection of water quality.
- The interception of rain by forests modulates its kinetic energy, reducing its ability to dislodge soil particles and cause erosion. However, a raindrop can regain terminal velocity in approximately 60 feet (Chang, 2003), so it is the vegetation beneath the forest canopy and the accumulation of organic materials on the forest floor that provides the greatest assurance of reducing the erosive energy of rain (Stuart and Edwards, 2006).

3.3 The Value of Forests in Protecting Drinking Water Supplies

Water arrives in the watersheds that supply cities and towns of Massachusetts as precipitation in all of its forms, in annual amounts averaging 44 inches of rain, and carrying a wide range of airborne pollutants acquired when water droplets form around atmospheric particulates and when condensing clouds dissolve atmospheric gasses, producing sulphur dioxide, nitrous oxides, ozone, and ammonia. It leaves these watersheds, to enter the drinking water supply systems, after traveling a path ranging from direct to torturous, through land cover ranging from pavement to dense forest, and after a time period ranging from minutes to years. In a forested watershed, the erosive energy of even the most driving rainfall is absorbed and the relatively long path and timeframe from precipitation to the tap purges pollutants, minimizes erosion of sediments and nutrients, and delivers high quality raw water.

There are exceptions to this general rule, e.g., when forest-based wildlife carries pathogens of human consequence to the watercourse. In general, however, there simply is no other watershed cover or land use that exceeds the purifying role of forests for protecting drinking water supplies nor that provides this protection more reliably, through the wide range of weather and climate extremes from severe drought to extreme rainfall events. Forested watersheds supply this unparalleled drinking water protection and can simultaneously deliver undeveloped open space and its associated values, protection for both rare and common species and their habitats, and renewable, sustainable wood production that supports rural economies and reduces dependence on long-distance transportation of natural resources.

The interactions of the components of the protective buffering provided by forest cover are complex, but include at least the following:

- The forest canopy intercepts precipitation before it reaches the forest floor, initially removing some pollutants (e.g., through denitrification), and reducing the velocity of raindrops before they reach potentially erodible soils.
- The midstory and understory vegetation further reduce the kinetic energy of falling rain, passing it more gently to the forest floor, and also aggressively capture mobile, inorganic nutrients from the soil water as they grow and accumulate biomass.
- From the point in time at which it is regenerated, a typical stand of trees in a northern temperate forest continues to accumulate biomass, first quite rapidly and then more gradually, until about year 50-60 (Bormann and Likens, 1979b). Beyond this point, the ratio of respiratory to photosynthetic tissues increases, so that net accumulations slow.
- The infiltration rates of forest soils, with high contents of organic materials, and porosity maintained by burrowing fauna and the penetration and decay of roots of all sizes, are seldom exceeded by the rainfall rates of precipitation events, so that overland flow and its associated delivery of sediments is an exceedingly rare event on fully forested watersheds. Even where 'hardpan' layers of soil exist within the forest soil profile, the result is generally lateral interflow rather than overland flow.
- Surface litter on the forest floor presents a Manning's coefficient (the roughness, or resistance to overland water flow) of approximately 0.4, and, where forest understory vegetation is dense, the coefficient can reach 0.8, in sharp contrast to the resistance provided by asphalt (0.012) or even dense turf (0.35) (Novotny, 2003) further reducing the surface transport of sediments and nutrients.
- Even following a major loss of overstory, e.g., resulting from a catastrophic storm event, the

dense organic matter of the fallen forest continues to resist erosion of particulate matter, and, so long as regeneration capacity is not restricted, rapidly regains control over mobilized nutrients (Bormann, et al., 1974; Foster, et al., 1997). This condition may be enhanced or diminished by active forest management, depending on a wide variety of factors.

- Thick organic soils support microbial denitrification (the conversion of nitrate, a potential water pollutant, to nitrogen gas) which represents a loss of inorganic nitrogen to forest vegetation, but also prevents excesses of that mobile nutrient from entering water sources.
- For partially forested watersheds, in which sources of pollutants exist up gradient from forested sections, biological remediation in forested systems can effectively filter and reduce potentially toxic components such as heavy metals (Chen and Cutright, 2003; Pulford and Watson, 2002; French et al., 2006), pesticides (Arora, et al., 1996; Paterson and Schnoor, 1992), and nitrogen in wastewater (Aronsson and Perttu, 2001) or from agricultural sources (Mayer, et al., 2006).and a wide variety of organic pollutants (Aiken, et al., 1991).

The protection provided by forest cover functions at every scale, from the individual tree, to a forest stand (by definition, relatively homogeneous in composition and structure), and most importantly, on the landscape scale of a relatively more diverse forested watershed.

An individual tree serves to capture and slow precipitation, processes pollutants through both mechanical trapping (http://www.sussex.ac.uk/press_office/media/media12.html) and biochemical processing (http://aeismag.com/issues/2001/june_july/bioremediation.htm), and buffers against local soil saturation through passive interception and evaporation and active transpiration (collectively, ‘evapotranspiration’). When rain falls on an individual tree, a significant portion of that rain is intercepted by leaves, branches, and/or the trunk of the tree (when the rain falls as snow, an even greater proportion is intercepted, especially by evergreens). An individual rain drop may be held until it is evaporated (approximately 10-15% of rainfall), it may simply run down the trunk of the tree to the ground (~5%), or it may drop to the forest floor directly, often after coalescing with other raindrops, or simply fall through gaps in the trees cover. While trees serve to break the fall of raindrops, these drops can regain terminal velocity within about 60 feet (Chang, 2003), so that a groundcover of plants and young trees and/or layers of accumulated organic materials are necessary to limit the erosive power of rain on the soils beneath a maturing forest.

While many factors determine evapotranspiration rates, a mature, open-grown deciduous tree is estimated to have in excess of 200,000 leaves, which, on a summer day, can transpire as much as 900 gallons of water (DeCoster and Herrington, 1988). The evapotranspiration associated with an individual tree limits the frequency with which the soils it occupies become saturated, thereby maintaining infiltration and limiting overland flow of water and the associated transport of nutrients and sediments. In addition, trees directly process pollutants in a variety of beneficial ways. Some airborne pollutants are simply trapped on the surfaces of the tree, removing them from the ambient air and temporarily stalling their entry to the water system. Biochemical reduction of pollutants by plants (the basis of “bioremediation”) is a varied and complex combination of processes that includes: improved degradation by soil microorganisms through rhizosphere enhancement (primarily nutrient additions); the uptake, translocation, and volatilization of unmetabolized compounds; and the uptake and metabolism or storage of other compounds (<http://www.cpeo.org/techtree/ttdescript/phytrem.htm>). So long as pollutant levels are not toxic to the trees, these processes clean the water that moves through the forest.

In addition to these direct influences on water quality, individual trees also: anchor soil; produce soil macropores as roots penetrate and die, increasing infiltration capacity and reducing overland flow; capture and utilize inorganic nutrients in the soil for growth and metabolism; provide shade that regulates decomposition processes and the temperature of streams; deliver organic materials (leaves, twigs) to the

forest floor, thus reducing erosion; and produce seed that enhances the forest's ability to recover from disturbances.

A forest stand protects water supplies through the multiplication of the effects of individual trees and understory plants, but also provides collective effects that go beyond those of individual plants. When an individual tree in a stand matures and begins to decline, it also begins to lose its ability to affect water quality. Leaf area, transpiration, root penetration, growth and nutrient uptake, shade, and eventually seed production all decline. This process may result from simple stem exclusion, through which an initial stand of perhaps a million seedlings per acre is reduced by competition to a mature forest of 100-200 trees. Or it may result from a large array of defoliators, fungi, or viruses, or from injuries following wind or ice storms. Regardless of the cause, the influence of a forest stand is that the living, thriving trees surrounding a tree in decline will utilize the resources made available by the dying tree, including sunlight, water, and nutrients, and the result for water quality is uninterrupted protection. When a disturbance forces groups of trees into rapid or gradual decline, the protection provided by the stand relies upon regeneration to replace the functions of the dying trees, rather than upon surrounding, vigorous trees.

Stand types (roughly homogeneous combination of species and age classes) produce categorically similar effects on water quantity. Evergreen conifer stands generally reduce water yield below that produced by deciduous trees on similar sites, primarily because evergreens continue transpiring throughout the year and because they intercept a higher percentage of snowfall, a portion of which either evaporates or directly sublimates. Stand age affects water quality, also in roughly predictable ways. Young, established stands of any species mix are accumulating biomass more rapidly than older, maturing stands, and therefore assimilating available nutrients more aggressively due to higher biotic demand for these nutrients (Bormann and Likens, 1979; Vitousek and Reiners, 1975). As expected, this demand is highest during the growing season, which is reflected in the seasonal patterns of nutrient flux. As decomposition and respiration rates begin to balance or exceed the rate of primary production, the capacity for nutrient assimilation by older stands begins to decline. Although nutrients are still held tightly by older stands, outside additions (e.g., atmospheric transport of nitrous oxides) can overwhelm this assimilation capacity and result in leaching and hydrologic losses, and long-standing accumulations of organics can lead to higher losses of organic forms of nutrients to adjacent waters (Hedin, et al., 1995).

The forested watershed accumulates the effects of individual trees and forest stands to provide highly resilient protection for drinking water supplies. Even after intense land use practices have pushed the forest toward homogeneity, the range of seed sources, topographic positions, water regimes, aspects, soil types, and bedrock composition conspire to maintain a diversity of stand types. The mix of types across the watershed at any given time produces a predictable yield, a predictable volume of water delivered to the reservoir or river system, while the inherent diversity in species composition provides the watershed forest with a level of redundancy in maintaining itself that rivals the most responsibly engineered water treatment plant. The diverse structure in the living green filter across the watershed, like diversity in an investment portfolio, yields more consistent performance through the vagaries of climate fluctuations, wind, snow, and ice, rainfall intensity, and damaging native and alien pests than a forest (or an artificial filter) built to a single design. The range in structural and species composition across the forested watershed represents built-in multiple barriers, providing a forest biofilter that functions 24 hours a day on free solar energy (Barten, 2006)

3.4 The Effects of Disturbance on the Watershed Protection Forest

3.4.1 General Types, Frequency, and Principles of Disturbance

Disturbances can be broadly categorized as *endogenous* (autogenic, originating within the ecological community, e.g., through the death and subsequent fall of a single large tree) or *exogenous* (allogenic, originating from forces outside the ecological community, e.g., wind thrown trees that occur as a direct result of catastrophic tropical storms) (Attiwill, 1994a; Bormann and Likens, 1979). Endogenous disturbances generally remain localized and pose minimal threats to water supplies, while exogenous disturbances can create either chronic or catastrophic landscape scale changes that may result in direct or indirect effects on these supplies. Attiwill (1994b) addressed the ability of a forest ecosystem to respond to disturbance in terms of *resistance* (factors which delay or prevent movement from a pre-disturbance state to a disturbed state) and *resilience* (the ability to return quickly to the pre-disturbance state).

There is a significant body of literature regarding the use of natural disturbances as a model for regulating the pace and design of silviculture and timber harvesting, driven in part by concerns for long-term sustainability (Attiwill, 1994b; Armstrong, 1999; Franklin et al., 2002; Larsen and Johnson, 1998; Roberts and Gilliam, 1995). Runkle (1985) calculated that on average, between major disturbances, regular, endogenous disturbances regenerate 0.5% to 2.0% of the temperate forest annually, suggesting that the natural forest is well adapted to deliberate silviculture that occurs at this rate. In the context of water supply forestry, these concepts may or may not be critical, depending on the objectives for the property and the intensity of silviculture that is possible. It is critical, on the other hand, to understand the temporal and spatial scales of natural disturbances for any given water supply protection forest, in order to design silvicultural treatments that work within this context to retain or increase resistance and resilience in that forest.



A recent "microburst" disturbance.

That the forests surrounding the Quabbin Reservoir will be regularly and at times catastrophically disturbed is without question. Throughout its past and recent history, this forest has been repeatedly visited by: snow and ice storms; strong winds that accompany thunderstorms, nor'easter's, microbursts, or hurricanes; occasional fires; intense precipitation events; chronic environmental changes such as air pollution or global warming; and a very broad spectrum of both native and alien insects and diseases. The estimated background level of endogenous disturbance for forests in this region of about 0.5% - 2.0% of the forest area annually, in part accounts for the relatively uncommon presence of trees or stands that have persisted for longer than 100-200 years (where such older trees or stands persist, they provide a valuable component of landscape-level biodiversity and may contribute important genetic diversity as well).

The science of forest disturbance and response is complex. The following are some generalized principles, from the disturbance literature:

- Overstory wind throw, in the absence of rapid regeneration, can temporarily increase erosion and nutrient leaching, by disturbing soils, increasing decomposition rates, and causing a setback in biomass accumulation rates.

- Severe forest fires can significantly reduce soil infiltration, thereby increasing overland flow of water, sediments, organic materials, and nutrients.
- A forest that is diverse in age structure and species composition limits the impacts of age- and species-specific disturbances.
- Forests with advance tree regeneration in the understory will recover more quickly from disturbances to the forest overstory than will forests with poor understory development.
- Younger, shorter trees will sustain less damage from severe windstorms than taller, older trees, due both to their lower tendency to “catch” the wind, and to stem flexibility.
- While tightly grown, aerodynamically smooth *stands* may deflect wind better than those that are aerodynamically rough, *individual trees* that have been grown in more open stands will develop strongly tapered stems that resist wind better than the non-tapered stems of trees grown in tight stands.
- Saturated overland flow from infrequent, large storms with intense rains and rapid spring snowmelt account for much of the annual particulate, sediment, and dissolved nutrient outputs from watersheds in any given year.

3.4.2 Specific Disturbances

3.4.2.1 Weather Events

There are many ways in which weather affects the growth and development of the watershed protection forest and in turn can affect quality and quantity of associated water resources, the most common are ice and heavy (wet) snow, wind (hurricanes, microbursts, intense thunderstorms, tornadoes), large accumulations of snow followed by rapid snowmelt, intense precipitation, lightning, and long periods of drought.

3.4.2.1.1 Ice or Heavy Snow Build-up

Ice or heavy snow build-up, caused by a variety of weather anomalies that are becoming more common as average winter temperatures rise, puts excessive weight on branches and eventually on stems, causing breakage that weakens or kills the tree. Conifers that dominate northern snowy regions have evolved a conical shape that sheds most snows successfully, without lasting damage, but the weight and tenacity of wet snow and ice can exceed even this adaptation. The most common cause for ice build-up is the melting of snow falling from higher, colder air masses as it passes through a warm layer of air before hitting forest canopy, where it refreezes and accumulates as ice. The ice storm of January 1998 was among the most extensive of these in recent history, covering a large area north and east of Lake Ontario, and depositing ice as thick as 4”, leading to massive loss of limbs and trees.

While these storms can be devastating set backs in the growth of trees and the production of everything from maple syrup to sawlogs, their impacts on drinking water supply may be less obvious. Hooper, et al., (2001) quantified the effects of the storm of 1998 and calculated that 33.6 cubic meters of woody debris was created by the loss of approximately 10% of the above-ground biomass in the forests of Quebec. This overstory reduction likely also resulted in an increase in decomposition rates and nitrification/mineralization rates, as well as a loss of evapotranspiration during the period from the storm until both the surviving overstory and the understory recovered. The massive additions of coarse and fine woody debris

to the forest floor would have prevented erosion from heavy precipitation, but the losses of evapotranspiration could result in higher water yields and the associated movement of nutrients from the forest and of stream deposits due to more frequent high flows and related bank erosion.

3.4.2.1.2 Wind

Wind is a constant component of the watershed forest and its effects have been widely studied. Trees have adapted to this ever-present force, much as conifers have adapted to the presence of snow. Tree crowns and stems are flexible and can bend sharply without breaking. Tree, branch, and even leaf structure allow the crowns to fold into an aerodynamic and compact form during average wind storms, without damage. For example, sugar maples with crown widths of 5-6m on a calm day have been observed to contract to 2m widths in storms of 110-120 km/h (Frelich, 2002) and conifers can reduce their surface area by 45% in winds of as little as 40 km/h (Banks, 1973).

There are limits to the resiliency of individual trees. Trees in dense stands may rely on the support of adjacent trees to overcome their otherwise wind-susceptible form, with small root masses and minimal stem taper, compared to open-grown trees in well-thinned stands. The risk in opening these stands is determined by the return interval of wind disturbance versus the thinning response time required for the residual trees to develop wind-resistant taper and root mass, both of which are difficult to predict accurately. Each species and size-class has a maximum sustainable wind speed. For sitka spruce this critical wind speed for tree failure has been calculated as 184 km/h for 18 m tall plantations with 3 m spacing (Blackburn and Petty, 1988). In addition to stand density, wind damage to individual trees is related to height (winds are stronger at greater heights above the ground) and trunk size (larger diameter trees have stiffer stems, which reduces their susceptibility to breakage, but increases the transfer of canopy wind force to the roots, where it can result in root failure and then toppling of the tree (Frelich, 2002; King, 1986)). Finally, age can affect the susceptibility of trees to wind damage simply because the likelihood of stem rot from a wide variety of sources increases with age (Frelich, 2002).

Occasional losses of individual or small groups of trees due to wind are not generally a problem for water supplies. The potential water quality effects of catastrophic winds that damage large areas of the watershed forest are variable, but may present risks. The loss of a block of trees on a stable site that is not adjacent to the water supply may result in only gradual, relatively minor adjustments to ecosystem processes, including nutrient losses from the site (Foster, et al., 1997). However, the uprooting of streamside canopy trees by hurricane force winds can result in a four-fold increase in groundwater nitrate and a doubling of stream water nitrate (Yeakley, et al., 2003). The impact on the forest canopy from catastrophic wind events was documented following the hurricane of 1938, by researchers at the Harvard Forest in Petersham, MA. On level or windward slopes, >75% of softwoods greater than 34 feet and hardwoods greater than 74 feet tall were damaged, and the landscape pattern of disturbance ranges from individual trees to areas as large as 35 ha (Foster and Boose, 1992). Depending on the saturation level of the forest soils and the type of trees affected, catastrophic winds may either break stems and branches or uproot whole trees. The later presents the possibility of moving exposed soils in subsequent or concurrent rain events, although intact surrounding forest floor would likely mitigate this affect.

Hurricanes have passed through the Quabbin forest every 20-40 years, although catastrophic storms (Category III or higher with winds above 110 mph; the Hurricane of 1938 had winds of 120 mph) have historically occurred just once every 100-150 years (Foster, 1988b). The most recent hurricanes to make landfall in the region as Category 1 or higher hurricanes were Gloria (1985) and Bob (1991). Hurricane Floyd (1999) weakened to a tropical storm before making landfall in New England.

Some wind events of concern are not as wide as hurricanes but may include stronger winds. Tornadoes (winds from 40 mph (F0) to 318 mph (F5), and widths 300 ft to 6,500 ft) or microbursts (downdraft winds with a horizontal extent of less than 2.5 miles but with winds approaching 170 mph) can cause extensive

damage to limited blocks of the forest. While climate changes may increase their frequency, Massachusetts typically receives 2 or fewer observed tornadoes annually (compared, at the other extreme, to 100-180 per year in Texas). Among the most damaging recent tornadoes in Massachusetts was the Great Barrington Memorial Day tornado of 1995, an F4 tornado (wind speeds from 207 to 260 mph) that caused 3 fatalities and 23 injuries and did extensive damage to forests and structures in its path. While they are uncommon in Massachusetts, the straight-line, destructive windstorm referred to as a “derecho” can reach wind speeds well in excess of 100 miles and persist for long distances as clustered downbursts associated with a rapidly moving band of thunderstorms (Frelich, 2002). At least three such storms have been recorded in Massachusetts since 1995 (<http://www.spc.noaa.gov/misc/AbtDerechos/derechofacts.htm#historic>).

3.4.2.1.3 Rapid Snowmelt and Intense Precipitation

Rapid snowmelt and intense precipitation share some common effects on the water supply. Both can overwhelm infiltration rates even when storage capacity by detention (large pores filled by gravity) and retention (small pores in which water is held against gravity by capillary or matric, adsorptive forces) in forest soils is still available. The result is high peak flows in the hydrograph, which often correlates to bank scouring and erosion of sediments and nutrients. It has been shown repeatedly in the Northeast that a very few high intensity storm and snowmelt events are responsible for the vast majority of the annual transport of sediments by any given stream.

3.4.2.1.4 Lightning

The effects of lightning on the watershed protection forest are generally of limited scale, often resulting in damage only to a single unfortunately positioned tree. It is not uncommon to see tall, dominant white pine trees in the Quabbin forest that have been struck by lightning, leaving a spiraling split in the bark from the top of the tree to the ground. This wound may heal over, but often it is the site for further damage to the tree as insects or diseases take advantage of the break in the tree’s defense. However, the loss of individual trees does not threaten water supply. An uncommon but potentially greater impact occurs when lightning strikes during a dry period and ignites a fire. There have been few incidences of this combination within the Quabbin forest during its history as a water supply protection forest.

3.4.2.1.5 Drought

Drought may not have direct consequences on the water supply other than the obvious reduction in yields. Severe droughts can cause mortality in the understory and overstory vegetation, but it is extremely uncommon for this to be extensive. More common are fires that follow long dry periods. The effects of fire on the watershed protection forest are reviewed separately in Section 3.4.2.2.

3.4.2.2 Fire

Fires in the conifer-dominated western U.S. have been shown to cause problems for water quality (Beschta, 1990), depending on the severity of the fire, the timing of following precipitation events, and the biogeochemistry of the region of the fire (Dissmeyer, 2000). Catastrophic wildfire is less common in the broadleaf forests of northeastern U.S. in modern times, in part because of suppression efforts but also due to the generally higher moisture content of trees and understory of the region during the growing season. Fires that do occur in these forests are frequently set by careless human activity and are most likely to occur when the snow is off the ground and leaves are off the trees.

Most fires in the Quabbin forest in recent years have been set by humans and confined to areas of less than 10 acres. They have been primarily low, ground fires that burn surface organic material and small trees and shrubs, but do not kill or move to the crowns of larger trees. There have been exceptions, including two fires in the 1950s (north of Route 122 and on the north end of Prescott Peninsula) that killed overstory trees and burned hundreds of acres at high temperatures. The Division's fire policy (Section 5.1.5) and regular improvements in suppression response have limited the impact of fires in the past several decades.

The potential impact of severe fire on water supplies includes the simultaneous death of overstory and understory vegetation and the exposure of mineral soils due to the wholesale consumption of organic materials on the forest floor. The loss of these critical layers exposes the forest soils to erosion by rainfall that reaches mineral soil without the dampening influence of vegetation or organic duff. The loss of organic materials also increases the transport of suspended sediments to watercourses during storm events that challenge the infiltration rate of the soils, especially on steeper slopes (Dissmeyer, 2000). Severe fire that kills riparian vegetation can result in increased stream temperatures and associated changes in stream water quality (Levno and Rothacher, 1969).

Nutrient losses to streams as a result of fire occur when relatively insoluble oxides of cations carried in ashes react with water and carbon dioxide and become more soluble, leading to increased potential for leaching (Debano et al., 1998), and through the acceleration of mineralization and nitrification (the conversion of organic nitrogen to inorganic forms) (Vitousek and Melillo, 1979). Immediately following fire, there is also reduced uptake of these nutrients by plants (because there are fewer living plants), further increasing the likelihood of leaching losses. Nitrate losses to streams and reservoirs can be more pronounced in areas that were approaching nitrogen saturation prior to the fire (Dissmeyer, 2000).

3.4.2.3 Insects and Diseases

Insects and disease-causing organisms are natural components of the forest ecosystem that under ordinary circumstances play a vital role in general biodiversity, decomposition and nutrient cycling, and predator-prey relationships. On the other hand, these organisms are occasionally capable of large-scale infestation and damage, in particular when the specific organism is imported from outside the area and therefore not subject to its normal suite of population-controlling predators. Insects and diseases are a major problem in the Quabbin forest only when their impacts conflict with the Division's objective of creating and maintaining a watershed protection forest. For the most part, this includes only large-scale outbreaks that threaten to alter tree species diversity or forest structure. Chestnut blight, which appeared in central Massachusetts in the first decade of the twentieth century, is an example of such a disease. Before the blight, chestnut was one of the dominant trees in the forest; today, it is essentially a minor shrub.

While native insects and diseases are generally kept in check by their predators, imported or "alien" species can cause significant damage if their natural controls are not introduced at the same time. The potential problems associated with insects and diseases, relative to water supply protection, include the sometimes very rapid defoliation of either a single species or several host species. The foliage of a water supply protection forest controls the erosive force of precipitation, slows decomposition rates by moderating solar radiation, and is the source of evapotranspiration, which moderates soil saturation and helps maintain potential infiltration and storage as a result, all of which work to maintain water quality. Where individual or even small groups of trees are defoliated, these impacts may not be significant, but when a significant percentage of a watershed area is defoliated in a short period of time, the impact on water yield and on the movement of nutrients and sediments can be significant, especially if the understory is not well-developed. For example, large scale defoliation by the fall cankerworm (*Alsophila pometaria* (Harris)) on the Coweeta Basin in North Carolina resulted in a fourfold to fivefold increase in

annual weighted nitrate nitrogen concentrations in otherwise untreated streams, compared to reference streams (Swank, et al., 1981).

3.4.2.4 Climate Change

The relationship between the management of watershed protection forests and climate change is complex. Evidence is overwhelming that the climate is changing and in particular, that mean annual global temperatures are rising. A wide variety of associated changes is predicted, many of which have implications for the long-term management of forests and forested water supplies. It is important to recognize that adaptive changes in forest management in response to predicted changes in the climate may or may not bring about the desired changes in forest structure in time to accommodate the effects of climate change. Forestry has always involved uncertainty in predicting changes and disturbances at the local and landscape scale. Global climate change increases the level of uncertainty managers are required to consider. A broad and conservative response to predicted fluctuations in the global climate is to enhance the forest's inherent natural resilience in order to maintain its ability to adapt quickly to change.

3.4.2.4.1 Weather Extremes

Weather extremes may have begun to rise in magnitude and frequency, although there remains debate about the range of inter-annual variability (Houghton, ed., 1996). More frequent storms with high winds and more intense precipitation, as well as more common occurrence of ice storms and heavy snows have been predicted or have begun to occur. Droughts are more likely as higher temperatures increase evaporation. Higher summer temperatures may favor the geographic expansion of some pathogens or insect pests. Ice storms and heavy snow may weaken individual tree crowns and therefore increase the vulnerability of the tree to pathogens (Broadmeadow, ed., 2001). A reduction in deep winter snow could increase the survival and impact of browsing ungulates. Winter cold injury to trees may be reduced.

3.4.2.4.2 Changes in Species Composition

Changes in species composition may occur in response to higher average temperatures and other changes (Iverson, et al., 1999). Species currently at the northern limit of their range may migrate further north and become locally more common. Species at the southern edge of their limit may become less common. Diseases and insects that affect the current and developing range of species in the Quabbin forest could likewise benefit or lose ground. Alien, invasive plant and insect species may become more of a problem.

3.4.2.4.3 Changes in Seasonal Patterns

Changes in seasonal patterns will have a variety of impacts. The synchrony between hosts and pest development may be altered, with positive and negative results (Broadmeadow (ed.), 2002). While winter cold injury to trees may be lessened, early spring bud break may leave trees more vulnerable to late season frosts. Early spring flushing of forage may enhance the survival of winter-stressed browsing mammals.

3.4.2.4.4 Effects of Rising CO₂

Effects of rising CO₂ levels have already been shown to include increased growth rates due to increased carbon uptake. Leaf area increases that result from increased growth rates also increase water use by plants through increases in transpiration and rainfall interception. Rising CO₂ may also lower the age at which trees become mature and begin producing viable seed (LaDeau and Clark, 2001).

3.4.2.4.5 The Effects of Climate Change on Soil and Water

The effects of climate change on soil and water may include long-term depletion of soil carbon stocks (via increased decomposition rates), although the increased productivity of the forest may counteract this for the foreseeable future, and predicted changes in soil carbon include increases or decreases (Zhou, et al., 2006; Jones, et al., 2005). Predicted increases in foliar densities would depend upon sufficient soil water supply, which is subject to the vagaries of precipitation. While a greater annual average precipitation rate could support greater foliage densities, the distribution pattern of this precipitation will likely have a greater impact than the increase in average volume. If it occurs as intense precipitation that challenges infiltration rates and results in higher peak flows, interspersed with longer dry periods, an increase in annual watershed yield may be more likely than an increase in foliage densities and transpiration rates.

3.4.2.5 Air Pollution and Forested Water Supplies

Air pollution is a chronic disturbance that influences the watershed forest. It also increases the forests value in water supply protection.

- Forests serve as “sinks” for various environmental pollutants, retaining them and slowing their movement into water supplies. A tall, dense, and layered forest serves this function more effectively than a short, sparse forest.
- Environmental pollution has been linked to general forest decline, which increases the susceptibility of those forests to insects, diseases and other impacts.
- Air pollution contributes to nitrogen saturation of forest ecosystems. Nitrogen saturation can cause elevated nitrate, aluminum, and hydrogen levels in streams and losses of cation bases from soils due to soil acidification. These impacts can be compounded by acid precipitation and ozone pollution, and ameliorated by the accumulation of biomass and storage of nutrients in an actively growing forest. Actively growing forests with a diversity of species and sizes may therefore help buffer the impacts of acid precipitation on water supplies.

Concerns about the influence of air pollution focus both on the direct impacts of air pollution on watershed forests and the impacts of resulting ecosystem degradation on water quality. It is extremely difficult to isolate the effects of air pollution from the many other processes and stresses occurring in forest ecosystems (climatic stresses, insects, diseases, fire, ice, wind, etc.). It is also difficult to isolate the impact of one specific pollutant, e.g., ozone or nitric acid, from the composite of impacts affecting a forest. Klein and Perkins (1988) state:

It is now recognized that no single causal factor is responsible, but that there are a variety of anthropogenic causal factor complexes interacting with natural events and processes that, together, induce stresses in forests that culminate in declines of individual plants and of ecosystems.

3.4.2.5.1 Acid Deposition

Carlton (1990) provides an excellent overview of the impact of acid deposition upon watersheds. In Massachusetts, data indicate that the average pH of precipitation is 4.2, which is six times more acidic than uncontaminated precipitation (Godfrey 1988, as cited in Carlton 1990). In New England, approximately 60-70% of the acid falls as sulfuric acid and 30-40% as nitric acid (Murdoch and Stoddard 1992; Rechcigl and Sparks 1985, as cited in Carlton 1990). Murdoch and Stoddard (1992) note a study in Maine that showed the sulfuric acid component decreasing in recent years, while the nitric acid

component is increasing, leaving the pH of precipitation fairly constant. For example, Stoddard (1991) reported that sulfate deposition had decreased by 1.8% from 1970 to 1984 in the Catskill Mountains of New York. However, the acidity remained the same due to equal increases in the nitric acid component. In Massachusetts, depositions amount to 0.3 to 0.7 pounds of hydrogen ion, 16.2 to 27.5 pounds of sulfate, and 8 to 22 pounds of nitrate per acre per year (Petersen and Smith 1989).

Sulfuric and nitric acids tend to accelerate replacement of aluminum, calcium, magnesium, and other base cations in the soil with hydrogen ions (Hovland et al., 1980, as cited in Carlton 1990). In this way, acid deposition will increase soil acidity and directly impact biological activity, soil fertility, and cation-exchange capacity (Carlton 1990). Acid precipitation can also leach aluminum directly into streams causing potential negative water supply and aquatic and fish impacts (McAvoy 1989). Key factors in determining the susceptibility of watersheds to acid inputs include: the supply of base cations in soils; the percentage of base-rich groundwater flow versus storm flow; the relative importance of snowmelt events; the average storm rainfall intensity, volume, and duration; and the soil depth, texture, pH, and cation exchange capacity (McAvoy 1989; Peters and Murdoch 1985; Veneman 1984). Records at Hubbard Brook, New Hampshire show that while sulfate inputs have declined, base cation inputs from precipitation have also declined (145 micro eq/liter in 1963 to 104 micro eq/liter 1989) causing sensitivity to acidification to actually increase (Driscoll et al., 1989). Decreases in base cations are generally related to a large reduction in suspended particulates since 1970, due to reduction of coal and open burning emissions.

Some researchers have questioned the extent of the impact of acid precipitation. For example, Krug and Frink (1983) feel that most aluminum in streamwater is due to acid soils (caused by natural humic acids) not acid rain. Krug and Frink (1983) and Veneman (1984) note that streamwater can become more acidic as the acid humus layer increases with forest age and because thick humus layers may reduce the amount of water percolating into the subsoil and increase saturated overland flow. Studies in Connecticut and the Berkshires of Massachusetts show that soil acidity increases with forest age (Art and Dethier 1986; Krug and Frink 1983). In Connecticut, litter pH changed from 5.5 to 3.9 from 1927 to 1980 and the mineral soil pH from 5.1 to 4.6 during this period. A study in Norway also concluded that changing land use and consequent vegetational succession was largely responsible for acidification of soils and water (Krug and Frink 1983).

Reuss and Johnson (1986) identified the key difference between natural and anthropogenic acid inputs as the ability of the stronger nitric and sulfuric acids to leach through to stream waters, whereas the weaker natural organic acids will leach from upper to lower soil horizons, acidifying soils but not stream waters. Therefore, a key factor in identifying systems acidified by pollution is whether pH is attributed to organic acids or sulfates and nitrates.

Driscoll et al., (1989) noted that the “acid rain” and “acid soil” argument is largely due to the lack of long-term data on basin soil and water quality. To help resolve this controversy, the authors compared two similar basins, one in New Hampshire (NH) where acid deposition is significant (pH 4.1) and one in British Columbia (BC) where acid deposition is insignificant (pH 5.0). The basins have similar bedrock, glacial history, and soils but differed in vegetation type and precipitation amounts. Both headwater streams were acidic. The key difference was that the BC stream was dominated by weak organic acids, had low aluminum concentrations, and low sulfate loading, while the NH stream was dominated by strong acids (nitric and sulfuric), had high aluminum concentrations, and high sulfate loading.

Two streams in the Quabbin watershed, the West Branch of the Swift River and the East Branch of Fever Brook, received similar analysis to those in NH and BC (Rittmaster and Shanley 1990). The concentrations of sulfate and hydrogen ions in precipitation were significantly higher at Quabbin than at the New Hampshire site. While both Quabbin streams had high aluminum concentrations during high

flow periods, Fever Brook aluminum was in an organic form that is not toxic to fish. Fever Brook also had one half the net export of sulfate of the Swift River, a result of sulfate reduction in the extensive beaver flowage at Fever Brook.

Veneman (1984) rated the ability of the soils of Massachusetts to buffer acid inputs using many of the criteria outlined above. Of the 25 soil types that make up almost all of the DCR lands at Quabbin, only four (all wetland soil types) were classified as “acid precipitation will have no negative impact on water quality,” whereas sixteen types are listed as “acid precipitation will have a moderate or significant impact on water quality.” Baker (1984) re-measured soil parameters at eight sites at Quabbin that had been measured in 1962. He found that soils had increased in acidity and exchangeable aluminum and were now releasing sulfate, whereas they were adsorbing sulfate in 1962. These changes have reduced the neutralization capacity of the soils.

3.4.2.5.2 *Interaction between Air Pollution and Forests*

Reuss and Johnson (1986) use the term “canopy leaching” for the process where hydrogen ions replace base cations in the forest canopy. Krug and Frink (1983) report that 90% of the hydrogen ions in acid rain at Hubbard Brook, NH are neutralized in the northern hardwood canopy during the growing season (rain pH of 4.1 changed to 5.0 in throughfall). In studies in the west-central Adirondack Mountain region of New York, Peters and Murdoch (1985) noted that throughfall in deciduous forests was less acid than rain, while throughfall in coniferous forests was more acid than rain.

As the forest flora exist in several layers above and below the ground surface, the accumulation/neutralization that occurs at these various layers tells a great deal about how the forest processes incoming acid deposition. Yoshida and Ichikuni (1989) studied the chemical changes to precipitation as it passed through the canopies of three different types of Japanese forests. They reported that from 49-74% of the total incoming acid deposition was neutralized by the forest canopies, with deciduous oak forests neutralizing the least and cedar forests neutralizing the most. Virtually all of the cations and anions studied, with the exception of the hydrogen ion, increased as precipitation fell through the canopy (the authors studied Ca^{2+} , Mg^{2+} , K^+ , Na^+ , NH_4^+ , H^+ , Cl^- , NO_3^- , SO_4^{2-} , and Al^{3+}). This indicates the process of “canopy leaching” is evident in these forests. The authors note that similar occurrences have been documented in New England by other authors.



Local sources of air pollution.

Laboratory studies indicate acid precipitation increases leaching of calcium and potassium from vegetative foliage (Smith 1981). In order for the forest canopy to replace the cations and anions lost, similar amounts of these substances must be taken up from the soil. In some cases, acid conditions cause these nutrients to be leached below the root zone where they become unavailable to plants (Klein and Perkins 1988). The net effect of the above processes is to acidify the soils and damage forest ecosystems (Yoshida and Ichikuni 1989).

An increase in the acidity of soil water causes the leaching of aluminum, which is an element of increasing concern to water supply managers. Aluminum also damages fine tree roots and inhibits the uptake of calcium, a nutrient vital to plant growth. This situation leads to further imbalance in nutrients and increases susceptibility to drought stress, decline in growth, and increased mortality (Johnson and

Siccama 1983, as cited in Art and Dethier 1986; Petersen and Smith 1989; Smith 1981). For example, soil acidity is a potential contributor to increased nitrate leaching from forests (Vitousek 1977). Klein and Perkins (1988) report that temperature, moisture, light, nutrients, and soil factors all contribute to susceptibility to disease. This type of pollution may also affect recovery from winter injury.

According to Klein and Perkins (1988), trees undergoing nutrient stresses may be predisposed to decline when natural and pollution-caused stresses are added. Forests in a condition of decline go through a process of “reorganization” during which increased nutrients are leached from the system into tributaries. This increased loss of nutrients may in turn perpetuate the forest decline.

Soil acidity will vary relative to air pollution levels, as well as other factors including soil type and horizon, underlying geology, and successional stage of forest cover (Art and Dethier 1986). In general, the soils of the New England region have a low acid neutralizing capacity or “ANC” (Godfrey 1988, as cited in Carlton 1990). Art and Dethier (1986) studied the relationship of land use and vegetation to the chemistry of soils in the Berkshires. Acidity of the upper-most soil layer was positively correlated to species composition and stand age, with stands less than 140 years averaging pH 4.21 and those over 140 years averaging pH 3.92. Several studies verify an increase in soil acidification with successional sequences following agricultural abandonment (Robertson and Vitousek 1981, Thorne and Hamburg 1985, Krug and Frink 1983, all as cited in Art and Dethier 1986). Acidity varied with land use history, with previously pastured lands having significantly lower pH in the upper horizons than previously cultivated lands. The conclusion is that past land use has a significant impact on species composition and overall soil acidity (Art and Dethier 1986). These studies are useful in considering overall differences in chemical processing in various types and ages of forests and in assessing the potential susceptibility of various forests to impacts of acid deposition.

Soil water pH generally decreases deeper into the soil profile. For example, in a study of eight forest soils in central Massachusetts, mean pH in the A and C horizons were 4.39 and 3.58 respectively; an increase in acidity of eight times. Exchangeable aluminum in the A horizons was nearly four times as high as in the C horizons (Baker 1985, as cited in Carlton 1990).

High levels of ozone cause injury to leaf surfaces of sensitive tree species such as white pine, black cherry, and white ash, especially during summer months. Ozone also reduces photosynthetic rates and the supply of carbohydrates to the roots (Petersen and Smith 1989; Reich and Amundson 1985; Smith 1981). High levels of ground level ozone occur at Quabbin Reservoir, with readings recorded at Quabbin Hill sometimes exceeding other state recording stations including those in Boston.

The combined effects of acid deposition and ozone pollution may be contributing to a measurable decline in Massachusetts forests. A statewide study of the Massachusetts forests identified 24,000 acres that show signs of decline, including yellowing leaves, dead branches, and standing dead trees. This represents a 10% increase in forest decline over twenty years ago (Parker 1988). In addition, the growth rate on one third of the red and white pines studied has dropped 20-50% since the 1960s (Freeman 1987). The overall impact of air pollution predisposes trees to insect and disease outbreaks. For example, research shows that air pollution predisposes pine trees to bark beetle infestations and makes several tree species more susceptible to root rotting fungus (Smith 1981).

In Massachusetts, the decline of red spruce and sugar maple has been examined most closely. Studies of red spruce on Mt. Greylock found that this decline involved a combination of factors, including pathogens, insects, and ice, snow, and wind. However, the decline studied was attributable only in small part to these factors. The high acidity of rain and fog, the high soil acidity, and the low soil nutrient content (including low calcium) at these sites point towards air pollution as a chief cause of the decline of

red spruce. The study of sugar maple decline also concludes that many trees are in a weakened condition, which magnifies the impact of other detrimental factors (Petersen and Smith 1989).

In addition to acid deposition and ozone pollution, current air pollution contains metals, polychlorinated biphenyls (PCBs), alkanes, and various polycyclic hydrocarbons and organic acids (Rechcigl and Sparks 1985, as cited in Carlton 1990). Soil and vegetation surfaces are the major “sinks” for pollutants in terrestrial ecosystems (Smith 1981, as cited in Carlton, 1990). For example, the leaves and twigs of an average sugar maple tree 12 inches in diameter will remove the following elements from the air in one growing season: 60 mg of cadmium, 140 mg of chromium, 5800 mg of lead, and 820 mg of nickel (Smith 1981). Klein and Perkins (1988) reported that the accumulation of metals affects nitrogen transformations in hardwood forests.

Forest soils serve as sinks for lead, manganese, zinc, cadmium, nickel, vanadium, copper, and chromium; tree trunks also serve as sinks for large amounts of trace metals including nickel, lead, chromium, cadmium, and manganese (Smith 1981; Driscoll et al., 1989). The U.S. Environmental Protection Agency designed a 40-acre “model forest” containing several hardwood species and white pine (Smith 1981, as cited in Carlton 1990). The model predicts that, within five years of planting, this hypothetical forest and its soils would annually remove the following pollutants:

- 96,000.00 tons/year of ozone
- 748.00 tons/year of sulfur dioxide
- 2.20 tons/year of carbon monoxide
- 0.38 tons/year of nitrogen oxides
- 0.17 tons/year of peroxyacetylnitrate

The net effect of air pollution on a forest ecosystem is a combination of decreased photosynthesis, decreased growth, increased respiration, reduced biomass, and possible reductions in reproduction. These impacts produce a range of symptoms that together are termed “forest decline.” The severity of the decline depends on the amount of pollutants, and the species and site conditions involved. An additional impact of air pollution is alteration of forest ecosystem composition and structure, through selectivity of impact. More severe air pollution, and air pollution on naturally stressed sites, serves to simplify the overall make up of the ecosystem and make it less diverse and less stable (Klein and Perkins 1988; Smith 1981). Smith (1981) defines three classes of air pollution impacts:

- *Class I:* low dosage, where the ecosystem serves as a sink for pollutants.
- *Class II:* intermediate dosage causing nutrient stress, reduced photosynthesis and reproductive rate and increased predisposition to insects and diseases.
- *Class III:* high dosage where mortality is widespread and gross simplification of the ecosystem alters hydrology, nutrient cycling, erosion, microclimates, and overall ecosystem stability.

Klein and Perkins (1988) reviewed more than 400 studies relating to forest decline and concluded:

There are interactions between primary causal complexes and their direct effects and secondary causes and consequences of forest decline discussed here, so that the web of interactions becomes formidable. Nevertheless, a start must be made on these analyses, not only to understand forest decline holistically, but also because of the pressing need to develop concepts and strategies to ameliorate or reverse the imminent collapse of forested ecosystems. Recognizing that species sensitivities to causal factor complexes vary greatly, inevitable simplification of ecosystems will drastically affect their ultimate stability.

3.4.2.5.3 *Nitrogen Saturation*

3.4.2.5.3.1 *Overview*

The potential problem of nitrogen saturation, defined as the declining ability of an ecosystem to retain added nitrogen, was only identified in 1981 (Aber 1992). Researchers are concerned that acid deposition may also be adding significant amounts of nitrogen, originating chiefly from nitrogen oxides in air pollution. The effects of nitrogen saturation include elevated nitrate, aluminum and hydrogen ion concentrations in stream water (Van Miegroet and Johnson 1993). Monitoring of nitrates is required for drinking water (standard=10 ppm) because of health effects upon infants and potential formation of carcinogenic byproducts (Skeffington and Wilson 1988). Nitrates can also cause algal blooms in lakes and reservoirs. Excess nitrogen deposition may also affect forest composition and productivity (Aber 1992).

Bormann and Likens (1979b) report a doubling in nitrate concentration in precipitation since 1955. Schindler (1988) reports that deposition of nitrogen oxides has increased much more rapidly than sulfates in recent decades. Ollinger et al., (1994) report that there is a more than twofold increase of wet nitrate deposition from east to west between eastern Maine and western New York State. The authors mapped broad-scale wet and dry nitrogen deposition across the Northeast, with the Catskill region in the highest category (10.34-12.66 kg N/ha/yr) and the Quabbin region in the lower (7.99-9.16 kg N/ha/yr) category.

3.4.2.5.3.2 *Processes Involved*

The processes related to nitrogen saturation are more complex than those related to precipitation inputs of sulfates, mainly because nitrogen can be both an acid and plant nutrient component and due to the complex interactions between soils and plants and the various compounds of nitrogen. In the ammonium form, nitrogen is a nutrient for the plant/soil biota complex. In the nitrate form, nitrogen can be a nutrient for biota but can also be a very mobile and dominant anion involved in base cation depletion and mobilization of aluminum through the soil and into stream water.

A key reaction in this process is nitrification, the conversion of ammonium to nitrate. Others are denitrification (in which atmospheric nitrogen is released from nitrates) and nitrogen mineralization (the process by which ammonium is formed from organic nitrogen in soils). Mineralization is an important process, as the storehouse of nitrogen in soils far exceeds that in the plant system (75-97.5% of nitrogen is in inorganic form in soils) but the nitrogen can be more mobile in the plant system. As long as the soil system delivers an amount of nitrogen less than or equal to the capacity of the plant system, nitrogen is held within the system. Thus, nitrogen saturation requires both the soil and plant systems to be saturated.

The interaction of these three processes – nitrification, denitrification, and nitrogen mineralization – is dependent upon various bacteria, pH levels, season and climate, as well as variations in plant/soil composition. An added complication is the process of nitrogen fixing, by which plants transform nitrogen gas (the most prevalent component of the atmosphere) to nitrogen in a usable form in the soil/biota system. The relative importance of nitrogen fixation is dependent on the composition of nitrogen-fixing plants in the system. Bormann and Likens (1979b) estimate that 70% of the nitrogen store at Hubbard Brook, NH is derived from fixation and the remainder from deposition. In general, predictions of the timing of the onset of nitrogen saturation are limited by the lack of understanding of soil properties and the complex processes at work there (Schofield et al., 1985; Agren and Bosatta 1988; Nadelhoffer et al., 1984; Aber 1992,1993).

Disturbance of the plant/soil system by natural or anthropogenic events tends to increase mineralization of nitrogen and consequent nitrification in the system. Vitousek et al., (1979) analyzed processes that keep nitrate leaching in balance. These include the accumulation of ammonium in soil solution on cation exchange sites in the soil, and lack of soil water for nitrate leaching. A delay in nitrate movement after disturbance is critical as this allows vegetation to develop and take up much of the available nitrate before it can leach into stream waters.

Van Miegroet and Johnson (1993) summarize the complexity of the nitrogen saturation process:

This soil condition is the integrated result of vegetation type, age and vigor, past N accumulation history, climatic conditions, and current and past N input regime and soil characteristics.

Aber et al., (1989) have developed equations based on field work that can help model the nitrogen cycle using soil litter analysis.

3.4.2.5.3.3 Symptoms and Site Susceptibility

Aber (1992) describes the characteristics – including annual stream water nitrate trends – of nitrogen-limited, nitrogen-transition, and nitrogen-saturated systems. In general, nitrogen-limited systems have low nitrate loss during snowmelt, high carbon to nitrogen ratios in soil litter, and high soil dissolved organic carbon concentrations. Nitrogen-saturated systems exhibit the reverse conditions for these three criteria. The identification of elevated nitrates in storm events, especially during snowmelt, may be a first indication that system inputs are exceeding capacity, at least temporarily. For example, researchers at the New York City water supply watersheds in the Catskills are concerned about peaks of nitrates in the spring (up to 128 micro eq/l) combined with elevated summer levels (Murdoch and Stoddard 1992). Rittmaster and Shanley (1990), in a study of two tributaries at the Quabbin, reported that nitrate concentrations were generally low, but nitrate peaks of 20 and >35 micro eq/l were reported in the two streams during the snowmelt period. The authors attributed these peaks to short soil contact time during storms. There are no other records of nitrate peaks at Quabbin, but limited storm sampling has been done.

Brown et al., (1988) recommend consideration of vegetation type and age, site history, carbon: nitrogen ratios in soil organic matter, external inputs, and nitrogen turnover rates to thoroughly evaluate the condition of a system with regard to nitrogen saturation. The authors note that because natural plant communities change, nitrogen saturation is a “moving target.” Van Miegroet and Johnson (1993) reported that forests with small soil nitrogen pools, due to either limited accumulation history or frequent disturbance such as fire, generally have low nitrification potential and insignificant nitrate leaching, irrespective of age or vigor of the forest. Sites that have high soil nitrogen content coupled with a low carbon to nitrogen ratio have a high nitrification potential, and under these conditions the annual leaching of nitrates is strongly dependent on atmospheric inputs, forest age and tree nitrogen uptake rates.

3.4.2.5.3.4 Impacts of Forest Succession and Disturbance

Stand age is an important factor in determining nitrogen uptake and annual nitrogen accumulation rates in tree biomass. A declining trend in nitrogen immobilization as a stand matures may explain why nitrate leaching losses are typically larger in mature versus vigorously growing forests. Long periods without disturbance may allow high nitrogen accumulation and low carbon to nitrogen ratios and increased nitrification potentials (Van Miegroet and Johnson 1993). Hemond and Eshleman (1984) note that both higher plant uptake and microbial immobilization contribute to limiting nitrate losses from Temperate Zone mid-successional forests.

Murdoch and Stoddard (1992) state:

In watersheds where forests are accumulating biomass, biological demand for nitrogen is often sufficient to retain virtually all atmospherically deposited and mineralized nitrogen during the growing season and reduces net nitrate release to stream water.

In their analysis of elevated summer nitrate levels in Catskill Mountain streams, Murdoch and Stoddard hypothesized that the older forests in the Catskill Preserve may have a low demand for nitrogen and may therefore be unable to retain all of the atmospheric nitrogen entering the watersheds. In a study of N-cycling within the long-undisturbed Biscuit Brook watershed in the Catskills, annual rates of N leaching from this older forest were more influenced by temperature-related differences in N processing (nitrification, mineralization) than by differences in N deposition rates, leading to speculation that climate warming could accelerate the pace at which an area approaches N saturation (Murdoch et al., 1998).

Aber et al., (1991) note that changes in species composition may affect the ability of a forest to absorb nitrogen. For example, due to longer needle retention, pine takes up less nitrogen than oak or maple. The authors also modeled the timing of nitrogen saturation of a hypothetical forest under different scenarios. For example, forest harvesting (removal of nitrogen) slowed the onset of saturation; ozone pollution reduced net primary productivity and moved the onset of saturation up from 300 years in the future (without ozone pollution) to 50 years into the future (with ozone pollution); and alteration of forest species from low nitrogen-demanding to high nitrogen-demanding species delayed the onset of saturation. This modeling exercise did not examine the impact of forest succession. In the Catskill Mountains, differences in N saturation and retention among forested watersheds seems to be related to differences in species composition and forest history, which in turn relate to forest succession (Lovett et al., 2000).

3.5 Resistance and Resilience in the Watershed Forest and the Role of Management

3.5.1 Forest Filtration: Redundancy and Diversity

A frequently applied principle in civil engineering is to design *redundancy* into systems built to serve and protect public health by duplicating critical components so that failure of primary systems does not present insurmountable risks. The drinking water supply biofilter provided by the forest is naturally redundant in the protection that it provides. This redundancy includes multiple opportunities for mitigating the kinetic energy of rainfall, the source of erosion, through layers of vegetation in the overstory, midstory, and understory, and the accumulated organic debris on the forest floor. In a fully functioning forest, redundancy includes the regular production of seed by mature trees, enabling their rapid regeneration when needed. Chemical filtration redundancy in the forest includes denitrification of inorganic nitrogen to nitrogen gas in the canopy as well as at the forest floor coupled with microbial remediation of pollutants within the forest soil. Roots within this filter provide reinforcement that adds to forest soil stability while they are alive, and increase soil porosity when they die and decay, therefore increasing the generally high infiltration rates of forest soils. In spite of its redundancy, the forest is unlike an engineered water filtration plant in that it is a living, dynamic system whose components change constantly, in response to growth and competition, mortality, and a wide range of disturbances that affect it on scales ranging from a fraction of an acre to the majority of a watershed. Therefore, building redundancy requires consideration not just of the structure of the static biofilter that exists at a given point in time, but also of working with natural processes to build long-term *resistance* and *resilience* in these natural filtration systems.

Ecological principles of diversity relate to the engineering concept of redundancy in the context of a biofilter. Both lead to stability in the system, to its ability to continue to function in spite of disturbance,

to resist changes in its desired functions and/or to recover those functions quickly following disturbance. A forest that is low in species diversity is more susceptible to mortality than a mixed species forest when species-specific pests arrive (e.g., gypsy moth in an oak-dominated forest). A forest that is predominantly composed of tall trees may be more susceptible to wind damage than a forest with mixed height classes. Beyond these self-evident sources of system stability through diversity, regions within these watershed forests are further categorized by their relative influence on water resources. Riparian forests are expected to consistently provide water quality control through high soil infiltration rates and rapid nutrient uptake. Upland forests are expected to yield water at consistent rates while preventing the loss of sediments or nutrients to the riparian forests or directly to streams through subsurface flow. To avoid sudden changes in these functions, these forests are expected to be able to resist damage and/or to regenerate quickly when damage occurs.

The most catastrophic damage that typically occurs in the New England forest has historically been produced by low-frequency, very high intensity hurricanes. The amplitude of the changes brought about by these disturbances is a function both of the intensity of the storm and the susceptibility of the forest. The hurricane of 1938 did vast damage, but the New England forest at that time was still rebuilding following farm abandonment that began in the mid to late 1800s and large scale cutting of old field pine at the turn of the century. Were the same storm to strike today, it would impact a more vulnerable forest, one that has grown consistently older and taller across the landscape except where management or smaller scale disturbances have rejuvenated it since 1938. Among the objectives for active management of watershed forests is to reduce the amplitude of unavoidable natural disturbances, through deliberate rejuvenation of a small portion of the forest each year. This general concept is simplified and depicted graphically in **Figure 12**, showing the amount of forest that was or might have been disturbed with and without management, by the hurricanes of 1635, 1788, 1815, and 1938.

3.5.2 Nutrient Dynamics

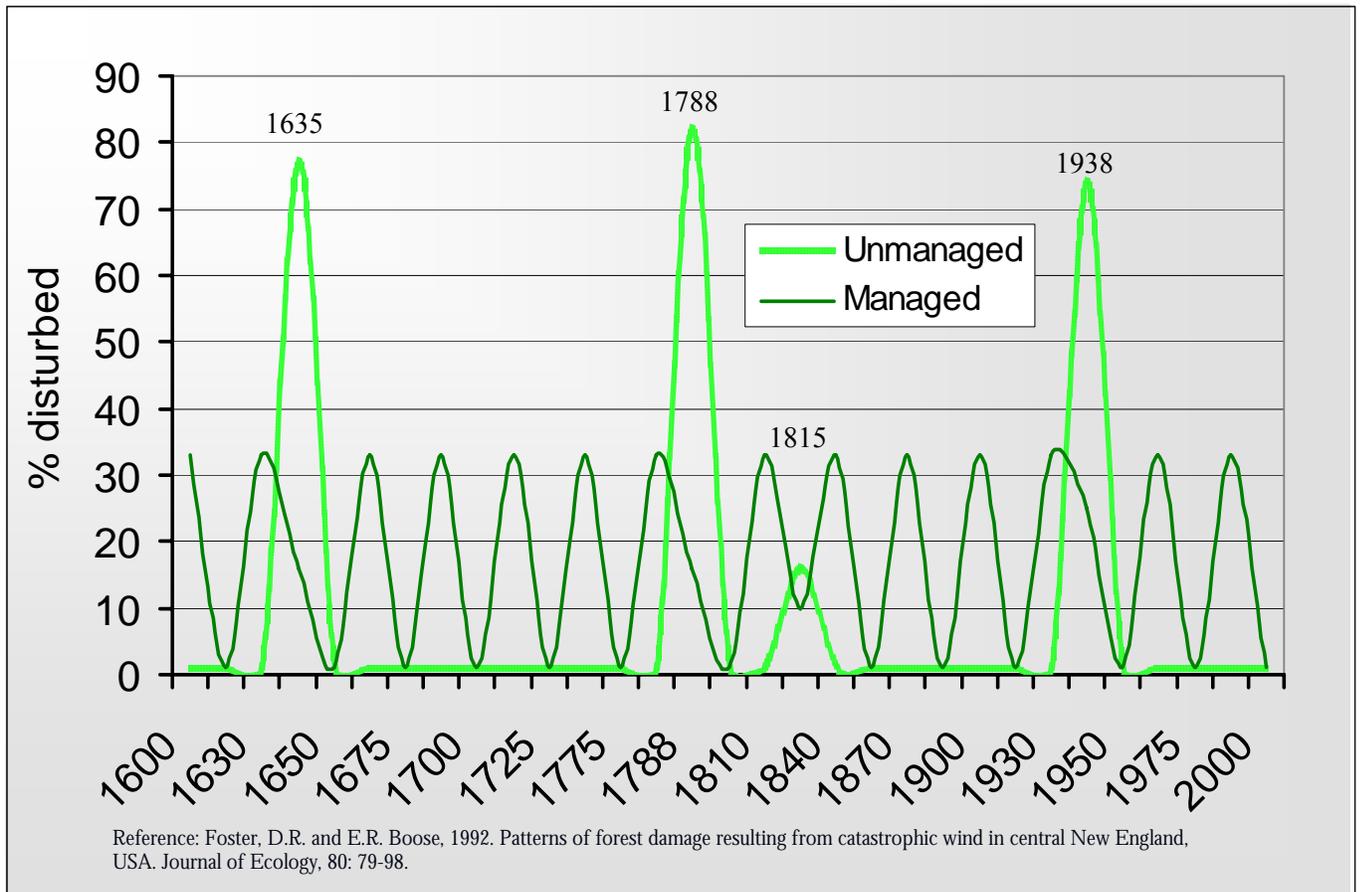
Understanding the influence of the forest on water resources requires an understanding of nutrient dynamics within these ecosystems. U.S.D.A. Forest Service researchers have detailed the hydrologic and nutrient dynamics in experimentally manipulated northern hardwood forests at Hubbard Brook in NH (Bormann and Likens, 1979). They describe the stages that a forest passes through following disturbance to the overstory without destruction of the biological, hydrological, and nutrient properties of the soils.

1. A relatively brief *reorganization phase*, during which the ecosystem loses total biomass despite accumulation of living biomass.
2. A much longer *aggradation phase*, during which the system continuously assimilates nutrients and accumulates biomass.
3. A variable length *transition phase* that occurs as the forest reaches a point where biomass accumulation slows and age-related mortality increases.
4. A *steady state*, at which point biomass losses from mortality are balanced by biomass accumulation that results from regeneration.

The progression of the forest along this path assumes that no additional large-scale catastrophic disturbance intervenes. The nutrient dynamics that accompany these phases also vary through time. The greatest regulation of nutrient export occurs during the aggradation phase, while losses are associated with both the reorganization and transition phases, and a relatively consistent level of control that is somewhat lower than during the aggradation phase is associated with the steady-state phase. Control over nutrient export following disturbance is provided by incorporating nutrients in accumulating biomass. If

it were possible to maintain an entire watershed forest in an aggrading condition, nutrient control could be held at an optimum level. But the forest is a dynamic system, moving along these phases steadily unless disturbances interrupt this progression.

Figure 12: Amplitude of Wind Disturbance in Unmanaged vs. Managed Forests



3.5.3 Forest Structure and Water Yield

Studies throughout the eastern United States have documented changes associated with removing overstory forest from a forested watershed. At Coweeta Hydrological Laboratory, in Franklin, North Carolina, increases in streamflow correlated well to both the percentage of basal area removed and the solar radiation load, which relates strongly to aspect (Douglas and Swank, 1975). Removal of vegetation temporarily reduces evapotranspiration, so that the largest increases in streamflow on cut versus uncut watersheds occur during the growing season. Studies at the Fernow Experimental Forest in West Virginia documented a strong correlation between soil moisture in the upper two feet of forest soils and streamflow, so that growing season soil moisture deficits in fully forested stands are reduced with cutting, resulting in increased discharge (Kochenderfer and Aubertin, 1975). Cutting alone will not produce this increase in discharge, but will cause it when subsequent precipitation events eliminate soil moisture deficits (Hornbeck and Federer, 1975). For partial cuttings that retain residual forest, the exposure of edge trees adjacent to openings increases their rate of growth and transpiration, resulting in a reduction of the yield that would be associated with full clearcuts (Hornbeck et al., 1975). Partial cutting may also

reduce snowmelt maximum discharge; the more rapid melting of openings desynchronizes the overall watershed melt (Hornbeck et al., 1975).

Forest hydrology research from 1900 to 1970 focused on water yield and sedimentation concerns. It was demonstrated repeatedly that producing measurable increases in yield required reducing watershed forest canopy by at least 20-30%, and furthermore that sediment loss following these canopy changes was almost entirely the result of poorly designed or constructed roads (Ice and Stednick, 2004). Many of these experiments involved paired watershed comparisons of undisturbed forest to forests that were either completely or partially clearcut (Stednick, 2000), but few were designed to test systematically dispersed partial harvesting or small group selection cutting.

The Leading Ridge Watershed Research Unit in Pennsylvania demonstrated that these are important matters by testing the yield results of cutting the upper versus the lower half of a watershed. Hubbard Brook researchers tested strip cuts, not a common practice in the Northeast and generally limited to plantation management. Harr and Frediksen (1988) tested “patch cut” partial harvesting in the watersheds that supply Portland, Oregon, but the smallest patch studied was 7.9 acres in size, an opening size that would be considered to be an even-aged clearcut in the Northeast U.S. A water yield experiment in a second-order watershed of Quabbin Reservoir tested small group patch cuts, but also cut or killed all trees within 100 feet of the tributary to test the water yield effects of greatly reducing riparian zone evapotranspiration (Mader et al., 1972). Cutting on Catchment 3 at Fernow included 2 “intensive selection cuts” removing 13% and 8% of the stocking, followed by a cutting of 0.5 acre patch cuts that removed an additional 6%, and finally a clearcut of the remaining stocking (Hornbeck et al., 1993). In general, once basal area removal exceeds 25% of the watershed total, the change in yield during the first year following removal is roughly proportional to the percent of the total basal area that was removed.

3.5.4 The Importance of Pattern

The importance of the configuration of partial cuts in predicting water yield was clarified in paired watershed experiments by comparing partial cuts ranging from 24% to 33% of the catchment basal area, but distributed differently, at Hubbard Brook, Fernow, and Leading Ridge experimental forests. The greatest yield of these partial cuts came from cutting 24% of Catchment 2 at Leading Ridge, which produced nearly twice the yield increase produced by cutting about 33% of the stocking on Catchment 2 at Fernow and Catchment 4 at Hubbard Brook. The basal area reduction at Leading Ridge was produced by clearcutting the lowest portion of the catchment, while cutting at Hubbard Brook was in alternating strips and cutting at Fernow was a diameter-limit removal of scattered individual trees (Hornbeck et al., 1993). Clearly, pattern, placement, and distribution of cutting have an effect on water yield, which in turn has implications for water quality.

Vitousek (1985) addressed pattern in determining nutrient cycling within patches, suggesting that there is a critical patch size below which nutrients and water made available by disturbance are likely incorporated in the intact forest at the edge of the opening (and above which nutrients and water may be more mobile). He further speculates on the watershed-level effects of patch disturbance:

A final question is, to what extent can vegetation patches within a watershed interact to influence ecosystem-level nutrient dynamics? Nutrients leaching through the soil from disturbed patches toward streams or groundwater could be taken up by adjacent aggrading patches or by the riparian vegetation near streams if patch sizes are small and if percolating water is not too deep. In such a case, the watershed as a whole could retain nutrients more effectively than any individual patch, and natural vegetation made up of a mosaic of patches could be significantly more retentive than vegetation managed

in large patches. Some experimental forest management schemes based on this possibility have been implemented (Hornbeck et al., 1975) or contemplated (Jordan, 1982). If these practices are useful and widely applicable, the management of patch dynamics would become an important way to manage the nutrient capital of terrestrial ecosystems.

Satterlund and Adams (1992) further discuss the importance of pattern in controlling water yield, which in turn controls the associated loss of nutrients and sediments. While there are issues of practicality in implementation, disturbing (harvesting) 25% of a watershed's forest cover through widely scattered small openings will yield less water and associated nutrients than concentrating this disturbance in a single block. The difference is in the control over yield that is exerted by the intact forest at the edge of the openings. The greater the ratio of edge length to opening size, the greater the control over yield will be. Therefore, managing the dynamic changes in the watershed forest's stage of development, by deliberately rejuvenating small, well-dispersed patches, could maintain an aggrading, nutrient-controlling condition across the watershed without exposing adjacent tributaries to the nutrient loss spikes associated with disturbances that affect large, homogeneously vulnerable patches.

The importance of forest pattern in developing resilience in the biofilter is also evident in peak flows associated with snowmelt. Snowmelt is often the strongest peak flow of the water year. Verry (1986), discussing the differences in long wave radiation energy added to the snow pack by mature versus young stands points out that "because there is a five-day difference between peak snowmelt flows in clearcut and older stands, harvesting will desynchronize snowmelt within a forested area and actually reduce flood peaks by 30 percent when a mosaic of young and older stands exist in the same area." Satterlund and Adams (1992) echo this, stating that "All in all, it appears that management systems that are designed to increase the natural heterogeneity of a watershed will flatten and broaden the snowmelt hydrograph. Cutting systems that increase homogeneity will sharpen it." Stronger peak flows carry the possibility of stronger losses of sediments and nutrients to tributaries and receiving reservoirs.

3.5.5 Age Structure and Resiliency

Murdoch and Stoddard (1992), in describing the potential sources of nitrogen they observed being flushed from streams in the Catskills during storms and spring snowmelt, offered the following as one possible explanation:

Neither the major source of nitrate during storm flow and snowmelt, nor the causes of long-term increases in nitrate concentrations in Catskill streams, can be identified with certainty. In all likelihood, both atmospheric deposition and natural processes contribute to the increasing nitrate concentrations in Catskill streams. Changes in nitrogen deposition rates alone cannot account for the nitrate trends in streams observed here, as mentioned earlier. However, episodes of high nitrate concentrations will result if the terrestrial ecosystem fails to retain atmospherically deposited nitrogen. The headwater forests of the Catskill Mountain region have not been logged since 1870, when they were incorporated into a state preserve, and major forest fires have not been reported since the 1840s. In general, young, rapidly growing forests will retain more nitrogen than forests that are mature or growing slowly. The Catskill forests may therefore have a low demand for nitrogen because they are at or near maturation, and can no longer retain all of the atmospheric nitrogen entering the watersheds.

The concept of nutrient "leakiness" of very old forests, which proposed that gradual declines in net productivity as senescence begins to balance growth might explain nutrient losses from these forests

(Vitousek and Reiners, 1975), has been further tested in unmanaged forests in Chile. In old-growth temperate forests in southern Chile, it is clear that nutrient uptake potential is less than in an aggrading, younger forest, but even in very old forests in this region, nutrients are still very tightly cycled between litter and living biomass, and only very minor amounts are lost to water courses (Hedin et al., 1995).

The difference between these forests and those of the Catskills, where old, undisturbed forests are losing nitrogen to the streams, may be primarily the difference in atmospheric inputs. The Catskills are among the regions on the receiving end of atmospheric nitrogen that accumulates from the burning of coal to produce power in the Midwest. Prevailing westerlies carry this nitrogen source to the Catskills, and the relatively low buffering potential results in losses. The old forests in Chile may be at similarly low uptake capacities, but do not receive the atmospheric inputs, and therefore do not flush nitrogen during peak flows. Nonetheless, these older forests are apparently more vulnerable when excess nitrogen does arrive, and the general conclusion of Vitousek and Reiners (1975), that “intermediate-aged successional ecosystems will have lower nutrient losses than either very young or very old (mature) ecosystems” may hold up well in areas receiving increased atmospheric nitrogen inputs.

3.5.6 The Effects of Management on Resistance and Resilience: A Working Hypothesis

The watershed protection forest, as described in Section 3.5.5, has built-in processes that, with or without active management, collectively resists change and rapidly works to recover equilibrium following a natural or deliberate disturbance. Experience has demonstrated, however, that there are opportunities to enhance these natural processes through active management. Past land management practices have had a lasting impact on Quabbin’s watershed forest in the following ways:

- For many decades, the loss of both natural predators and hunting by humans left deer populations as the dominating influence over the regeneration of the forest following disturbance. Deliberate management of this population, beginning in 1991, has restored the regeneration process, although a growing, unchecked moose population has begun to challenge it again.
- The artificial planting of several thousand acres of agricultural fields to white and red pine homogenized the age and species composition of these new forest stands, many of which were planted on sites on which they were eventually susceptible to root rot and wind throw. Removing these plantations in stages has resulted in new age classes of site-suited natural regeneration that is diverse in species composition, a generally more stable forest structure.
- With the regeneration process restored, management has moved from improvement thinnings to regeneration harvesting designed to create a mosaic of age classes across the forest that mimics the annual 0.5 to 2.0% natural disturbance cycle but in a more regulated pattern. While infrequent catastrophic disturbances will still arrive on the watershed, the forest that these will affect will include well-distributed patches of forest that are resistant to these disturbances, due to enhanced vigor or to age, size, or species diversification, or to a combination of these elements.

Other evidence from the science of forested watershed management offers additional and compelling reasons to consider active manipulation:

- Snowmelt, which generates some of the highest peak flows in a water year, is synchronized to the extent that a subwatershed is homogenized in species or size class,; it all occurs within a single relatively narrow time period, maximizing the peak event. On the other hand, a subwatershed of mixed species composition and a variety of age classes tends toward desynchronization, because patches of snow gathered within differing stands will melt at different rates, resulting in a longer

length, but lower amplitude peak flow. This in turn limits bankfull stream conditions and the associated erosion of accumulated sediments and organic materials.

- Experimentation with strip and small patch cutting has verified the ability of retained patches, if the balance is carefully designed and maintained, to capture and utilize water and/or nutrients that have been mobilized by the cutting of adjacent stands. Within some site-specific limits, the diversification of size classes within the forest can be accomplished without exceeding the ability of the residual, undisturbed, adjacent forests to control losses of water, nutrients, and sediments. Once this diversification has been accomplished, the standing forest structure should be more capable of resisting and rapidly recovering from large scale, outside disturbances.
- Research in nutrient cycling indicates that the maintenance of a steady component of aggrading, middle-aged stands within a watershed forest that includes all age classes should optimize the buffering of further nutrient inputs, for instance when these arrive via atmospheric deposition. Maintaining this forest structure requires steady recruitment of younger forest to replace middle-aged stands that have matured.

Active management carries risks that can reduce or eliminate the potential gains associated with deliberate manipulation. Current research on these risks indicates that by following a few clear rules, these risks can be controlled or eliminated:

- Researchers have shown that a minimum of 20-30% of the stocking of a forest must be cut within a short time (1-3 years) to increase water yield. Conversely, to minimize yield increases and the loss of nutrients or sediments that may accompany yield increases, managers need to limit harvesting to not more than about 25% of the stocking on a forested watershed in any given 5-10 year period.. Using GIS and GPS technologies, it has become possible to maintain these standards fairly efficiently.
- Separating the roads and staging areas from water resources is among the basic rules to protect those resources from negative impacts due to logging. Roads should be designed to minimize stream crossings and stormwater drainage structures need to be properly designed and maintained. Staging areas should be kept far enough away from water resources to be hydrologically remote.

The working strategies for actively managing the Quabbin watershed forest to take advantage of some of these principles are:

- To maintain the ability of the forest to regenerate itself.
- For the next decade, to annually regenerate approximately 1% of the actively managed forest, using small group selections or patch harvesting to maintain multi-age class structure and diverse species composition.
- To strictly adhere to Conservation Management Practices that have been customized for drinking water supply protection.
- To limit harvesting to no more than 25% of the stocking of any given subwatershed during any given 10 year period.

3.5.7 *Wildlife Effects*

The wildlife effects of greatest concern in the development of resistance and resilience in the watershed protection forest are those that limit the establishment and development of plants, particularly trees. The stability and long-term functioning of the biological filtration provided by the watershed forest is dependent upon its ability to grow and reproduce perpetually, following a wide range of disturbances.

While low populations of browsing ungulates (deer, moose) can be accommodated within the watershed protection forest while maintaining sufficient forest structure, there are limits to this accommodation. Deer populations in excess of 15-20 per square mile begin to limit both the species composition and the ability to meet minimum densities of tree regeneration following disturbances to the overstory. Moose weigh nearly 10 times as much as white-tailed deer and consume 50-60 pounds of vegetation per day. In addition, they are capable of feeding on tall saplings, by breaking their stems to bring the canopy within reach or by simply walking over them to bend them to the ground. In order to escape moose browsing, trees need to be much larger than the 4-5 feet tall that is considered to be beyond regular deer browsing reach. In extreme cases of heavy browsing, tree regeneration is nearly absent, so that the values for water quality protection that are associated with maturing trees (Section 3.3) are also absent.