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Preparing for the Onset of Hemlock Mortality in Great Smoky Mountains National Park: An Assessment of Potential Impacts to Riparian Ecosystems

Scott Wesley Roberts
University of Tennessee - Knoxville

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To the Graduate Council:

I am submitting herewith a thesis written by Scott Wesley Roberts entitled "Preparing for the Onset of Hemlock Mortality in Great Smoky Mountains National Park: An Assessment of Potential Impacts to Riparian Ecosystems." I have examined the final electronic copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Master of Science, with a major in Geography.

Ken Orvis, Major Professor

We have read this thesis and recommend its acceptance:

Carol Harden, Roger Tankersley, Jr.

Accepted for the Council:
Carolyn R. Hodges

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)

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Preparing for the Onset of Hemlock Mortality in Great Smoky Mountains National Park:

An Assessment of Potential Impacts to Riparian Ecosystems

A Thesis
Presented for the
Master of Science Degree
The University of Tennessee, Knoxville

Scott Wesley Roberts
May 2006

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Abstract

Currently the Hemlock Woolly Adelgid (*Adelges tsuga*) is spreading across forests in eastern North America, causing the decline and mortality of the eastern hemlock and the Carolina hemlock. Investigation into the impact of hemlock mortality on ecosystem processes has only recently begun and is not yet fully understood. The loss of hemlock from riparian forests in Great Smoky Mountains National Park (GSMNP) could reasonably be expected to result in significant alterations to stream environments. The goal of this study was to assess the influence of riparian hemlock stands on stream conditions and estimate possible impacts from hemlock loss in GSMNP. I paired hardwood- and hemlock-dominated streams that were similar in topography, geology, land use, and disturbance history using Geographic Information Systems (GIS) analysis and statistical techniques. I then monitored each stream pair for water temperature, nitrate concentrations, pH, and discharge over eleven months. I found that differences between hemlock- and hardwood-dominated streams could not be explained by dominant forest type alone; the presence of hemlock or hardwood riparian forest does not appear to exert a consistently dominant signal on measured conditions of headwater streams in GSMNP. The variability in the results suggests that other landscape variables, such as the influence of understory Rhododendron species, may exert more control on stream conditions than differences between hemlock and hardwood canopies. For example, Rhododendron was found to reduce light levels reaching the forest floor and streambeds in both hemlock- and hardwood- dominated forest stands.

Evidence from recent peer-reviewed literature suggests that short-term stream condition impacts from forest disturbances can be severe. However, research also indicates that conditions can return to pre-disturbance levels within five to ten years. In GSMNP, the return to long-term stability of stream conditions after hemlock mortality will depend on the type of replacement species and how quickly the replacement species can establish in disturbed sites. There is evidence that deciduous hardwood species are most likely to replace hemlock. The results of this study suggest that hemlock and hardwood stream conditions are similar in GSMNP. Therefore, *if* hardwood species are able to replace hemlock in GSMNP and streams are able to recover from short term impacts, the long term impacts from hemlock mortality on stream conditions will be minimal. However, the presence of Rhododendron in riparian hemlock forests in GSMNP may prevent hardwood species from effectively replacing hemlock, which could hinder the return to long-term stability.

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Chapter 1. Introduction

1.1 Subject of the Thesis

In the interconnected world in which we live, invasive exotic pests are perhaps the most immediate threat to the conservation and preservation of our natural areas. The introduction of exotic pests can be detrimental to biological diversity, economies, aesthetics, and human health, among other things (White 1997). Natural ecosystems are particularly vulnerable to the introduction of exotic species. In forested ecosystems, exotic pest infestations can significantly alter species composition, stand structure, and forest functions (Jenkins et al. 1999). Tree mortality as a result of infestation can modify ecosystem processes by impacting stand dynamics, succession, and nutrient cycling, and can make ecosystems more vulnerable to additional disturbances (Orwig et al. 2002). Although the spread of exotic species has occurred throughout history, it has significantly increased during the last 100 years due to the increased frequency of anthropogenic introduction of exotics (Kizlinski et al. 2002).

Today eastern North America's forests are being threatened by a number of exotic pests, including the Hemlock Woolly Adelgid, *Adelges tsugae* (HWA). The HWA is currently spreading across the forests of eastern North America, causing the decline and mortality of one of the eastern temperate forest's most important tree species, the eastern hemlock (*Tsuga canadensis* (L.) Carr.). Decline and mortality of hemlocks due to HWA infestation have already occurred in the mid-Atlantic, but HWA has only recently made its way into the southern Appalachians. In 2002, the adelgid was found within the borders of Great Smoky Mountains National Park (GSMNP) (Johnson et al. 2005). The consequences of hemlock mortality on ecosystem processes have not been investigated until relatively recently. Although the eastern hemlock is thought to play a unique role in eastern forests, its particular influence on ecosystem functions is not fully understood. In GSMNP, where hemlocks commonly grow along riparian corridors, the loss of hemlock from these riparian forests could significantly alter stream environment conditions.

The threat of the loss of hemlock from riparian forests in GSMNP has caused a growing concern among scientists about the impacts of hemlock mortality on stream environment conditions. The type and magnitude of these impacts are not fully understood. In an attempt to understand these impacts, I raise the following questions:

1. To what degree does hemlock-dominated riparian forest influence stream conditions and the adjacent riparian environment?
2. What will be the initial, *short-term* impacts to stream conditions and the riparian environment caused by the decline and mortality of hemlock?
3. What *long-term* changes will occur with the eventual replacement of formerly hemlock-dominated forest with non-hemlock dominated forest?

This thesis will explore these questions and their significance for GSMNP.

1.2 Purpose

The purpose of this thesis was to assess the influence of riparian hemlock stands on stream environment conditions and to explore potential short-term and long-term responses of stream environment conditions to hemlock loss in GSMNP. The response of stream conditions to hemlock mortality will occur at two different temporal scales: 1) The initial, short term, immediate response of stream conditions to hemlock decline and mortality; and 2) The long-term, eventual response of stream conditions to the replacement of hemlock-dominated forest with non-hemlock-dominated forest. In this thesis, I report the results of an assessment of the potential impacts at both of these temporal scales. In order to investigate the unique influence of riparian hemlock stands on stream conditions, it is important to understand differences between a functioning hemlock-dominated riparian forest and a functioning hardwood-dominated forest. To evaluate this difference, I developed a terrain-based site selection and sampling methodology that allowed direct comparison of hemlock-dominated riparian environments with topographically similar hardwood-dominated riparian environments. I then compared stream temperature, pH, nitrate concentrations, discharge, and available photosynthetic light on paired sites dominated by hemlock and hardwood. These observations and sampling were conducted monthly for one year. This comparison allowed me to characterize the relationship between hemlock forests and adjacent stream water conditions. Although only eleven months of data are presented here due to time constraints, twelve months of data were collected and may appear in future publications.

A thorough assessment of the initial short-term impacts of hemlock decline and mortality would require long-term consistent monitoring throughout the transition from a healthy hemlock stand to HWA infestation to hemlock decline and mortality. Since my study for practical reasons was limited to only one year of data collection, such long-term monitoring was not possible. Instead, the data I collected serve as baseline data, with which additional observed changes in future stream conditions can be compared to evaluate the magnitude of short-term changes that occur after my baseline data are measured. Additionally, a section of the literature survey provided in this thesis explores potential short-term impacts to riparian environments from hemlock loss by surveying literature related to the impacts of other forest disturbances on riparian environments.

Hardwood species are the most abundant species observed to regenerate in HWA-damaged forests in south central Connecticut, New York, and Pennsylvania (Orwig and Foster 1998; Yorks et al. 2003). With the onset of hemlock mortality, hemlock-dominated riparian systems may eventually be replaced with hardwood-dominated riparian systems. With this in mind, I hypothesize that observations of similarly structured hardwood-dominated stream environments, will serve as a predictor of the long-term outcome of changes in stream environments formerly dominated by hemlock. My study was intended to provide an established baseline to which the expected long-term transition from hemlock to hardwood forests can be compared.

1.3 Justification and Objectives

HWA is expected to cause decline and mortality of hemlocks throughout GSMNP. Hemlock mortality in riparian stands, in turn, may cause severe impacts to stream environment conditions. These impacts may be inevitable because hemlocks are an important component of riparian forests in GSMNP, and because hemlocks show no sign of resistance to HWA infestation. To date, there have been no studies of the immediate response of stream conditions and ecosystem processes to riparian hemlock mortality in GSMNP. Additionally, the long-term impact of this change in riparian forest composition on stream environmental conditions is unknown. In GSMNP, there have been no direct comparisons of the functional differences between streams draining hemlock-dominated forest and streams draining hardwood-dominated forest.

Therefore, in order to assess the potential long-term impacts of hemlock mortality and associated forest dynamics, I examined the differences in environmental conditions between streams draining forests dominated by hemlocks and streams draining forests dominated by mixed hardwoods within GSMNP. With the anticipated changes in forest ecosystems due to hemlock mortality, it is imperative that we strive to comprehend the potential ecosystem functions that may be lost through the decline of hemlock among riparian forests. My study investigates this relationship before HWA-induced hemlock mortality reaches the levels that are already observed in the Mid-Atlantic and Northeastern United States.

I used the following methods in this investigation: 1) The use of Geographic Information Systems (GIS) and Remote Sensing data to characterize watersheds within GSMNP based on available geographic data; 2) The identification of terrain-based paired watersheds with hemlock and hardwood dominant forests; 3) The use of monitoring techniques to conduct a year-long assessment of environmental conditions including stream temperature, stream pH, stream nitrate concentrations, and forest floor insolation at the selected sites.

1.4 Study Area

The study area for this investigation is Great Smoky Mountains National Park, located in the southern Appalachians along the border of North Carolina and Tennessee (GSMNP; Figure 1). GSMNP is one of the largest federally protected areas in the eastern United States, encompassing 212,000 ha (525,000 acres). The Park is an International Biosphere Reserve and a World Heritage Site. With 10 million visitors per

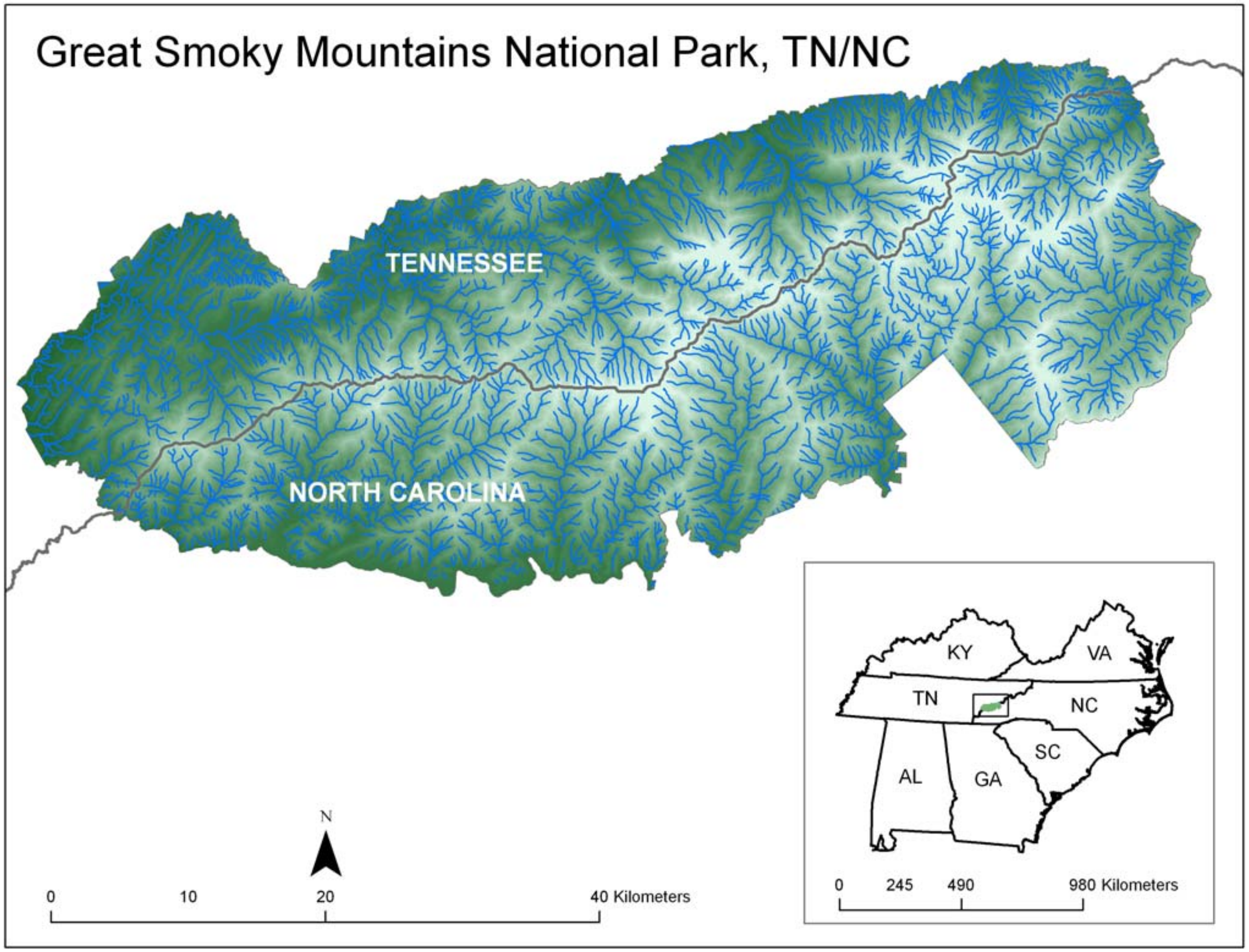


Figure 1: Great Smoky Mountains National Park

year, GSMNP is the most visited National Park in the United States. Visitors are attracted to the Park for its natural beauty and many other aesthetic qualities, such as its cool, dark hemlock groves.

Topography varies greatly in GSMNP and elevations range from 256 m (840 feet) near Abrahms River to 2024 m (6643 feet) at the summit of Clingman's Dome**.** The complex topography and rugged terrain allow for a diverse mix of forest communities, microclimates, and habitat. GSMNP is world-renowned for its biodiversity, as it contains over 1500 species of flowering plants including 100 species of native trees (Walker 1991; Kemp 1993). There are 12 general vegetation types in GSMNP: Cove Hardwood, Grape Thicket, Grassy Bald, Heath Bald, Mesic Oak, Mixed Mesic Hardwood, Northern Hardwood, Pine, Pine-Oak, Spruce-Fir, Treeless, and Tulip Poplar (Johnson 1995). The eastern hemlock is a dominant species of at least five vegetation types and is an associate species in most of the other seven (Johnson 1995). Hemlock forest is widespread throughout the Park, covering 1545 ha (3820 acres) (Johnson et al. 1999). GSMNP contains a substantial amount of old-growth eastern hemlocks, including the largest and oldest eastern hemlock in existence (Johnson et al. 1999). Hemlock in GSMNP is found mostly along lower elevation, sheltered streams, but also on exposed ridges and north-facing slopes. GSMNP contains over 3200 km (2000 miles) of stream channels. High-gradient streams in GSMNP provide habitat for a diverse aquatic biota, including several endangered species such as species of salamanders and the native brook trout.

GSMNP is an ideal location for research because of the abundance of geographic data available. The Park is a perfect setting for my study because HWA infestation is at an early stage, and hemlock mortality could have substantial impacts on broad-scale ecosystem components such as nitrogen saturation of watersheds and endemic populations of brook trout.

I focus on first- and second-order headwater streams in GSMNP. Study sites are located entirely within the GSMNP boundary, thus diminishing the influence of modern land use or human-induced disturbance. Small headwater streams were chosen for this study for a number of reasons. Headwater streams are important habitats for macroinvertebrates, fish, and amphibians (Meyer and Wallace 2001). Due to their geographic isolation, they support genetically isolated populations that are important for local biodiversity (Gomi et al. 2002). Headwater streams are more closely tied to landscape processes and have a greater response to disturbances than larger network streams (Gomi et al. 2002). This creates a tight interconnectivity between headwater stream characteristics and the landscape attributes and processes that occur within the stream’s catchment area. Headwater streams, because of their topography and small stream channel size, typically have vegetative canopy closure above the stream channel, while network streams typically have more open canopies above the stream channel. Thus, riparian vegetative canopy may have greater influence on headwater stream temperatures than on those of network streams. Smaller streams also carry lower volumes of water than larger streams, making them more susceptible to changes in water temperature. Moreover, typical low flow in small headwater streams occurs in late summer, when water temperatures reach their annual maxima, making such streams more vulnerable to the impacts of increases in solar radiation caused by hemlock loss. So it is logical for this study to focus on streams that currently may be influenced the most by hemlocks and that may be the most susceptible to disturbance from the loss of hemlock in the future.

1.5 Thesis Organization

This thesis is organized as follows: Chapter 2 provides a discussion of the literature that is applicable and relevant to this study with emphases on the eastern hemlock, the hemlock woolly adelgid, and the potential impacts of hemlock mortality on riparian environments; Chapter 3 discusses the use of statistics and GIS analysis in the selection of research sites for this study, methodologies used in field data sampling, laboratory analysis, and statistical analysis; Chapter 4 presents the results of the comparisons between paired watersheds; and Chapter 5 discusses the relevancy of these results, potential future research, and forest management implications.

Chapter 2. Background

2.1 Range and Ecology of Eastern Hemlock

The eastern hemlock (*Tsuga canadensis* (L.) Carr.) is an important eastern conifer, occurring from southern Ontario east to Cape Breton Island, and south through the Appalachian Mountains (Little et al. 1980). The main population of eastern hemlock is found in the northeastern United States and Canada, where hemlocks occur throughout a wide range of topographic positions. However, in the southern Appalachians, their intolerance of drought and/or heat stress restricts them to moist cool coves, rock outcrops, and north-facing ridges (Little et al. 1980). The eastern hemlock is one of the longest living tree species in the Appalachians. Hemlock can live for up to 800 years, reach a height of over 44 m (175 feet), and grow to a diameter of 1.8 m (6 feet) (Ward et al. 2004). It is one of the most shade tolerant tree species in the Eastern United States and is able to survive in the understory with only 5% of full sunlight (Ward et al. 2004). Its dense, evergreen canopy casts deep shadows on the forest floor, creating a distinct microclimate.

The eastern hemlock is often categorized as a “foundation species” because of the unique communities it creates in forest ecosystems. A “foundation species” has been defined as “a single species that defines much of the structure of a community by creating locally stable conditions for other species, and by modulating and stabilizing fundamental ecosystem processes” (Dayton 1972; Ellison et al. 2005). Indeed, the attributes of eastern hemlocks create forest communities that are distinctive and significantly different than nearby deciduous-dominated forest communities. Hemlock-dominated communities are characterized by deep perennial shade, a sparsely vegetated understory, slowly decomposing litter, a deep duff layer, retention of moisture, and moist, acidic, nutrient-poor soils (Godman and Lancaster 1990; Jenkins et al. 1999).

These conditions have a significant impact on the microclimate and environment of the hemlock understory and forest floor, creating an important habitat for plant and animal species. The dense shade is used for thermal cover by many mammalian species including whitetail deer (*Odocoileus virginianus*), ruffed grouse (*Bonasa umbellus*), and turkeys (*Melegris gallopavo*) (Anderson and Loucks 1979). Several species of birds have been found to have a significant association with hemlock forest including black-throated green warbler (*Dendroica virens*), blackburnian warbler (*Dendroica fusca*), hermit thrush (*Catharus guttatus*) and Acadian flycatcher (*Empidonax virescens*) (Tingley et al. 2002). A variety of aquatic species exhibit a strong association with riparian hemlock habitat including brook trout (S*alvelinus fontinalis)*, brown trout (*Salmo trutta)*, and several aquatic invertebrate species (Snyder et al. 2002; Ross et al. 2003). In headwater streams of the Delaware River basin, brook trout and brown trout populations were found to be two to three times as prevalent in hemlock-dominated streams compared to hardwood-dominated streams (Ross et al. 2003). Of particular concern for GSMNP is the loss of important habitat for the native brook trout. The National Park Service and other wildlife and fisheries management agencies have invested a great amount of time and resources in the preservation and conservation of genetically pure brook trout populations currently living in the Park. There is concern that the loss of riparian hemlock habitat in GSMNP would threaten to reduce, if not eliminate, existing brook trout populations and significantly diminish the success of these agencies' investments (Ross et al. 2003).

In addition to the eastern hemlock's numerous ecological values, the tree also has strong aesthetic qualities, evident in various acts of reverence in American culture. Recreational areas such as campgrounds and picnic areas are often found in or near hemlock stands because of the public's fondness of hemlock's aesthetic qualities. It is the state tree of Pennsylvania and even appears in a poem by Robert Frost (Frost 1923). Additionally, the eastern hemlock is valued for its use in horticulture and landscaping. Eastern hemlock is commonly used in landscaping because of the general appeal of its evergreen foliage. Although timber production of hemlock peaked in the early 1900s, hemlock is still harvested for pulpwood (Godman and Lancaster 1990).

Another species of hemlock, the Carolina hemlock (*Tsuga caroliniana* Engelm.), is endemic to the southern Appalachians and only occurs in the Blue Ridge Mountains of eastern Tennessee and western North Carolina. Although Carolina hemlock occurs in the Blue Ridge Mountains just to the east of GSMNP, it has not been found to occur in GSMNP. In locations where the Carolina hemlock and the eastern hemlock both occur, the Carolina hemlock is smaller in size, but almost indistinguishable in appearance.

Hemlock stands are an important component of the forests of GSMNP, covering approximately 1545 ha (3820 acres) and often occur in old growth virgin forests that include trees over 400 years old (Johnson et al. 1999). Hemlock stands in GSMNP frequently occur along mountain streams, where they affect a number of environmental conditions and ecosystem processes including water quality, light and energy insolation, nutrient cycling, and type of aquatic habitat (Johnson et al. 1999). The loss of hemlock from these riparian sites may inevitably have significant consequences, but the specific impacts are not yet fully understood and have received little attention.

2.2 The Hemlock Woolly Adelgid

The HWA is currently spreading across the forests of eastern North America, causing the decline and mortality of both the eastern hemlock and the Carolina hemlock (Kizlinski 2002). The HWA is an aphid-like insect, native to Japan, and was first discovered in the forests of Virginia in the early 1950s (Orwig et al. 2002). Since its arrival, it has rapidly spread north, but has only recently made its way into the southern Appalachians. In 2002, HWA populations were discovered near Fontana Dam in GSMNP (Johnson 2005). High rates of hemlock mortality due to HWA infestation have been observed in the Mid-Atlantic and Northeastern United States, and hemlock mortality is likely to occur in GSMNP (Yorks et al. 2003). The hemlock is not the first tree species in the Park to be threatened by an exotic pest or pathogen. GSMNP has already lost most of its chestnuts (*Castanea dentata)* to the chestnut blight, and most of its Fraser fir (*Abies fraseri*) to the balsam woolly adelgid.

The HWA can be devastating to forests due to its rapid reproduction cycle of two generations per year, high rate of migration, and variety of dispersal vectors via wind, birds, mammals, and humans (McClure 1990; Orwig et al. 2002). There has been only limited success in developing methods to control populations of HWA through insecticides, native predators, or exotic predators (Orwig et al. 2002). To date, the most promising method of biological control is the exotic beetle, *Pseudoscymnus tsugae* (P.T. beetle), which is currently being reared in a number of laboratories including the Institute of Agriculture at the University of Tennessee.

Recent studies have reported that HWA populations decline when exposed to extreme winter temperatures of –30° C (–22°F), but since temperatures in the GSMNP rarely reach that low, cold temperature will not be a factor in controlling HWA populations in the South (Skinner et al. 2003). Additionally, I hypothesize that the high levels of atmospheric nitrogen deposition that occur in GSMNP may result in increases in HWA populations. McClure (1991b) found that HWA population densities were more than five times higher on hemlocks exposed to nitrogen fertilizer than on unfertilized hemlocks. HWA survival rates and egg production were also found to be higher on nitrogen fertilized hemlocks (McClure 1991b). With this in mind, hemlock stands in GSMNP receiving high levels of atmospheric nitrogen deposition and experiencing nitrogen saturation may be more susceptible to HWA infestation and may undergo a more rapid rate of mortality.

Once a hemlock stand is infected by HWA, it may suffer complete mortality within 4 to 10 years (Orwig et al. 2002). To date, hemlock stands have shown no sign of developing a resistance to HWA infestation or any sign of recovery once infestation occurs (Snyder et al. 2002). Moreover, unlike other pathogens or pests, HWA infestation has been found to cause defoliation in all life stages, saplings and seedlings as well as overstory trees (McClure 1991a). Researchers investigating the susceptibility of hemlock to HWA infestation have found that landscape variables such as forest composition, forest structure, slope, and elevation do not play a significant role in a particular hemlock stand’s susceptibility or mortality (Orwig et al. 2002). Therefore, an infested hemlock stand on any type of landscape could potentially be eliminated from a site in less than a decade. Infestation of HWA should be perceived as a significant threat, and the decline of hemlock forest in GSMNP may be inevitable.

2.3. Potential Impacts from Hemlock Decline and Mortality

2.3.1. Observed Initial Impacts from Hemlock Decline and Mortality

Although Hemlock Woolly Adelgid-induced hemlock decline and mortality are only beginning to occur in GSMNP, it has already occurred in the mid-Atlantic and northeastern United States where the following changes to forests have been documented: 1) a reduction of overstory canopy; 2) a net reduction of nutrient uptake by vegetation; and 3) a reduction of evapotranspiration rates (Orwig and Foster 1998; Kizlinski et al. 2002; Snyder et al. 2002; Yorks et al. 2003). In forests where hemlock is the dominant tree species, these responses to hemlock mortality can significantly alter ecosystem processes such as nutrient cycling and surface water quality. The reduction of overstory canopy causes more light to reach the forest floor, and thus increases soil and stream water temperature (Kizlinski et al. 2002). Soil samples from hemlock stands undergoing decline have indicated the accumulation of nutrients, net increases in mineralization of nitrogen, increases in soil pH, lower C:N values, and increases in nitrification and nitrogen saturation (Jenkins et al. 1999; Yorks et al. 2003). These soil observations are likely the result of the reduction of nutrient uptake by declining hemlocks and of increases in detritus (Yorks et al. 1999). Increases in soil moisture in HWA infected hemlock stands have been observed as a result of reduced evapotranspiration (Yorks et al. 2003). The abundance of soil moisture and accumulation of nutrients lead to significant leaching of nutrients from the soil (Jenkins et al. 1999). Excessive concentrations of ions have been measured both in the soil and stream water of declining hemlock stands (Jenkins et al. 1999; Snyder et al. 2002; Yorks 2003). These high ion concentrations are of concern because they indicate the leaching of cations (Ca^{2+}, Mg^{2+}) and anions (NO_3^-,

SO_4^-), the mobilization of metals (Al^{3+}), and the acidification of soil and surface water (Yorks et al. 1999). These observations indicate that the initial impacts from hemlock decline and mortality could cause a depletion of soil nutrient capital and increases in nutrient loading, acidification, and metal concentrations in surface water.

Other observations in hemlock stands undergoing decline or mortality include alterations of litter-fall input and forest floor decomposition rates, as well as increases in the emergence of shade-intolerant vegetation (Jenkins et al. 1999; Yorks et al. 2003). The emergence of shade-intolerant vegetation is attributed to changes in forest floor insolation due to canopy decline.

In GSMNP, hemlocks commonly occur along streams where altered riparian forest conditions can rapidly influence stream chemistry and water quality. Of particular concern in GSMNP are alterations to stream water temperature, solar radiation received by stream water surfaces, and stream nitrate concentrations caused by hemlock decline and mortality. This thesis will address the potential impacts to stream water temperature, pH, discharge, and nitrate concentrations in GSMNP.

2.3.2 Stream Water Temperature and Solar Radiation

Importance of Stream Water Temperature

One of the most likely impacts to stream environmental conditions from hemlock loss is alteration of stream water temperature regimes. Researchers hypothesize that the perennial shade provided by riparian hemlock stands exerts a strong influence on stream water temperatures (Snyder et al. 2002). The loss of hemlock's perennial shade is expected to alter stream water temperature regimes and possibly result in adverse impacts to aquatic biota. In order to assess this potential impact, it is important to understand the role of stream water temperature as a component of aquatic habitat.

Water temperature is an important stream attribute, as it drives many physical and biological processes, especially in small streams (Beschta et al. 1987; Mellina et al. 2004). For example, stream temperature has been found to be the most important factor controlling the distribution and migration patterns of trout (Greene 1950). Stream temperature can guide the timing of life cycles of aquatic biota, such as emergence, spawning, and migration. Water temperature drives the functions of aquatic biota, including metabolism rates and growth rates (Johnson and Jones 2000). Modifications to natural stream water temperatures caused by forest disturbances can have numerous biotic impacts. In cold water systems, such as headwater streams in GSMNP, disturbances that lead to increases in maximum stream water temperatures are of great concern. Exothermic species, such as trout and invertebrates, are sensitive to the temperature of the water surrounding them. Trout species are especially sensitive to and limited by high stream water temperatures (Greene 1950). Higher stream temperatures require more energy from trout to sustain biological processes and functions (Tate et al. 2005).

Although very little is known about the impact to stream temperatures from hemlock mortality, the impacts of other disturbances on stream temperatures are well documented. Land-use change, wildfire, and forest harvesting have led to changes in the seasonal and diurnal timing of maximum temperatures as well as the occurrence of elevated maximum temperatures (Greene 1950; Bartholow 2000; Johnson and Jones 2000; Sullivan et al. 2000; Mellina 2002; Johnson 2004). Extreme disturbances can create lethal stream water temperature levels, causing mortality of stream biota. More commonly, however, disturbances create sub-lethal stream water temperature levels which cause thermal stress in stream biota (Sullivan et al. 2000; Tate et al. 2005). Thermal stress effectively can result in mortality through alterations to organism functions including reproductive success, alterations of the timing of life cycle events, and increased incidence of disease (Becker and Fugihara 1978; Johnson and Jones 2000; Tate et al. 2005). Researchers have found that stream temperature controls the rate at which salmon eggs mature, the timing of emergence of larval salmon, and the timing of migration (Sullivan et al. 2000; Tate et al. 2005). Modifications to stream water temperatures could alter these processes and the timing of their occurrence. Local increases in stream water temperatures can also create thermal barriers in streams, preventing certain aquatic species from migrating past the barrier (Richter and Kolmes 2005).

The temperature ranges that aquatic species can tolerate differ from species to species. Sullivan and others (2000) have suggested that temperatures over 25 °C (77 °F) are lethal and temperatures ranging from 19.4 °C (67 °F) to 24.4 °C (76 °F) are sub-lethal for Salmonids in the Pacific Northwest of the United States (Tate et al. 2005). The eastern brook trout (*Salvelinus fontinalis*), which is native to GSMNP, has an ideal stream temperature of 18.9 °C (66 °F) and a maximum limit of 23.9 °C (75 °F) (Greene 1950). If stream water temperatures exceed these suggested ranges and fish populations are not able to migrate to more suitable habitat, they may experience thermal stress.

The concern for stream water temperatures reaching lethal and sub-lethal levels has led the Environmental Protection Agency (EPA) to include high temperatures in the

Federal Clean Water Act's list of potential pollutants. The EPA has developed a critical threshold for stream temperature. If a stream's temperatures are above the critical threshold, it is listed on the "water quality limited" list (Dodge 2005). However, creating a specific critical threshold for stream temperature can be difficult. The responses of aquatic biota to stream temperatures vary by species, geographic location, and life stage (Beschta et al. 1987; Sullivan et al. 2000). Therefore, the critical thresholds for stream temperature must be developed for individual reaches based on the species and life stages present. Additionally, critical thresholds for stream temperatures must take into account multiple species and multiple life stages that occur within the same reach.

Stream water temperature also drives physical processes such as organic matter decomposition rates, nutrient cycling, and the solubility of gases (Johnson 2004). For example, as stream water temperatures increase, dissolved oxygen decreases. Reduced dissolved oxygen levels caused by increased stream water temperatures can cause significant stress in fish species (Richter and Kolmes 2005). Additionally, researchers have found that the toxicity of some organic chemicals and metals increases in response to elevated stream water temperatures (MacLeod and Pessah 1973; Howe et al. 1994; Richter and Kolmes 2005).

Solar radiation and other factors influencing stream water temperature

In order to assess the degree to which the perennial shade of riparian hemlocks influences stream water temperature, it is necessary to understand all factors that contribute to the thermal regimes of mountain stream water. Stream water temperature is a function of multiple environmental factors. These include shade, riparian vegetation, air temperature, substrate, conduction, groundwater influence, discharge, stream channel geometry, and direct solar radiation (Johnson 2004). There is disagreement within the peer-reviewed literature about the magnitude of influences that each natural contributing factor has on stream temperatures (Johnson 2004). This is in part due to the complex interconnectivity of natural ecosystems and the inherent difficulty in examining the influence of each factor independently. However, advances in temperature monitoring technologies and methodologies have provided data that suggest that the major factor influencing stream water temperature is incoming solar radiation (Johnson 2004).

Because water has a high specific heat, differences in the amount of solar radiation striking water surfaces strongly influences water temperature regimes. The amount of incoming solar radiation that strikes surface water depends on a number of factors. As a first-order control, topography and the shape of terrain can greatly influence the amount of incoming solar radiation. A stream flowing through a concave gorge, hollow, or canyon will receive less solar radiation than a stream flowing through a broad valley due to the terrain's potential for intercepting solar radiation. As a second-order control, riparian vegetative cover also determines the amount of solar radiation that a stream can receive. Riparian vegetation can shade a stream and keep incoming solar radiation from striking the water surface. A stream reach that has no riparian vegetative cover will receive much more solar radiation than a similar reach that has an abundance of riparian vegetative cover (Brosofske et al. 1997). In return, differences in solar radiation will create differences in water temperature regimes between the two reaches. Johnson (2004) conducted an experiment examining the influence of shade on stream water temperatures. Johnson measured air and water temperature for three weeks before shading, three weeks during shading, and three weeks after shading. Shading was simulated using constructed black plastic sheets supported by a suspension system approximately 2 m above the surface of the stream water. Johnson found that maximum stream water temperatures decreased significantly during shading. However, it is not known how representative the constructed “shading” was of the shade that natural riparian vegetation provides.

The ability of riparian vegetation to intercept solar radiation differs based on its form and structure. Trees generally have a greater potential for intercepting solar radiation than grasses (Bartholow 2000). Additionally, different tree species have different leaf area indexes and thus some species produce denser shade than others. Riparian vegetation not only decreases the exposure of a stream to direct solar radiation but also insulates the stream water from heat loss at night. The potential for insulation differs between riparian species as well. For example, an evergreen tree, such as hemlock, can provide insulation to a stream all year long, while deciduous trees lose some of their potential for insulation during winter when they lose their leaves.

Although incoming solar radiation is the major factor influencing stream water temperature, other factors have strong influences as well. However, these other factors are directly related to incoming solar radiation. Stream discharge and channel geometry can influence water temperature, but their influence is ultimately based on the influence of incoming solar radiation. The rate of potential temperature change from solar radiation depends on the volume of water and rate of flow in a stream. A stream with a smaller volume of water can change temperature faster than a stream with a larger volume of water. Simply put, it takes more solar radiation to heat a larger volume of water and less solar radiation to heat a lesser volume of water. With this in mind, it is logical that streams with a smaller volume experience greater diurnal and seasonal fluctuations of water temperature than streams with a larger volume. Additionally, this explains why annual maximum water temperatures occur during the late summer, when streams typically have the lowest volume and rate of flow and direct insolation is at its peak (Moore and Minor 1997). Stream channel geometry also plays a role in determining water temperature. The surface area of a stream is a function of its width. A wide stream has a large water surface area and is able to receive greater amounts of solar radiation than a narrow stream containing a similar volume of water (Moore and Minor 1997). Thus, streams that are wide, shallow in depth, and contain a small volume of water are particularly susceptible to disturbance-induced water temperature changes.

In the past, researchers have argued that air temperature is a major factor influencing stream water temperature. Models that are used to predict stream water temperature over broad areas often use air temperature as the main independent factor. However, Johnson (2004) argues that, although air temperature and water temperature are correlated, the correlation does not necessarily imply causation. Air temperature, as well as water temperature, is driven by incoming solar radiation, and, based on her field experiment studies, Johnson concludes that solar radiation has a much greater influence on stream water temperature than air temperature has.

Conduction, another factor that is considered to influence stream temperature, is also a function of incoming solar radiation. When a reach receives solar radiation, some energy from the solar radiation is absorbed directly by the substrate and then transferred to the water through conduction, influencing its temperature. Different types of substrate absorb and conduct energy differently. For example, Johnson (2004) found that the magnitude of diurnal water temperature fluctuations in a bedrock reach was greater than in a gravel reach, while average daily water temperatures were similar.

Stream temperatures are also regulated by the mixing of surface flows with subsurface, or hyporheic, flows. Hyporheic flows have been found to have lower magnitude diurnal fluctuations than those observed in stream surface flows (Evans and Petts 1997; Johnson 2004). In some streams, surface water is lost to subsurface flow and then reemerges downstream. In other streams, significant influxes of groundwater enter into the in-stream surface flow. The inflow and outflow of surface and hyporheic waters and the influx of groundwater can contribute greatly to the overall water temperature of a stream. The influence of groundwater on stream water temperatures is especially important in the summer, when groundwater is typically cooler than the surface water (Moore and Minor 1997). The influx of groundwater in the summer can help cool surface waters to water temperatures that are more ideal for stream biota and thus prevent thermal stress.

Importance of Riparian Vegetation

In GSMNP, hemlock commonly occurs along streams and functions as valuable riparian vegetation. The threat of the loss of hemlock as a riparian species in GSMNP has caused a growing concern among scientists. Although the impacts from the loss of hemlock as a riparian species are unknown, the overall value of riparian vegetation in general is well understood. Additionally, it is logical to expect impacts from disturbance to hemlock riparian forest to be similar to impacts caused by other disturbances to riparian forest.

Vegetation is an important component of the riparian environment and is essential for its well-being. Riparian vegetation plays an integral role in moderating stream water temperatures by intercepting solar radiation and providing insulation to stream water. Riparian vegetation also creates microclimatic conditions, which vary by species but greatly influence riparian environmental conditions as a whole (Snyder et al. 2002). For example, riparian hemlocks provide deep shade, woody debris, and shelter from wind (Orwig and Foster 1998). Riparian vegetation also controls certain processes of fluvial geomorphology. By stabilizing stream banks, riparian vegetation can help determine stream channel geometry, including stream width-to-depth ratios and exposed water surface areas, both of which influence stream water temperature. It is also the source of woody debris that accumulates in and around the stream bed. Woody debris itself provides habitat and is a food source for a variety of stream biota. Different shrub and tree species provide different types and amounts of woody debris (Ross et al. 2003). Some researchers have suggested that woody debris from riparian hemlock stands creates a unique habitat for aquatic biota that is not found in hardwood stands (Snyder et al. 2002; Ross et al. 2003).

Some researchers have suggested that the impacts from the loss of riparian canopy caused by hemlock mortality will be similar in type to impacts caused by other disturbances to riparian forest. The effects of riparian vegetation disturbance on stream temperatures have received considerable attention in the recent peer-reviewed literature. Although less is known about the ecological effects of forest pest disturbances, substantial attention has been given to understanding the ecological effects of forest harvesting within riparian areas. When riparian vegetation is removed by forest harvesting, stream temperatures significantly increase (Bartholow 2000; Johnson and Jones 2000). Johnson and Jones (2000) found that maximum stream temperatures occurred earlier in the summer and increased 7 °C (12.6 °F) after clear-cutting of riparian vegetation in western Oregon. They also documented that after 15 years of regrowth of riparian vegetation, stream temperatures returned to pre-harvest levels. In another example, researchers in the southern Appalachians found evidence that riparian forest harvesting had caused brook trout to migrate upstream in search of cooler water temperatures (Greene 1950).

There is disagreement in the literature about whether impacts to stream temperatures caused by forest harvesting are carried downstream to riparian areas where forest harvesting did not occur. Warmer stream water temperatures caused by forest harvesting may have an effect on downstream areas, or it is possible that once the affected stream water flows far enough downstream, stream water temperatures return to normal as it mixes with water from undisturbed sources (Bartholow 2000).

In addition to the impacts to stream temperature, riparian disturbances such as forest harvesting can also result in alterations to a stream’s fluvial geomorphology. For example, once riparian vegetation is removed, stream banks are no longer stabilized by vegetation and heavy erosion from stream banks can occur (Bartholow 2000).

In an attempt to minimize the impacts of forest harvesting on streams, forest harvest best management practices now suggest leaving a substantial buffer of riparian vegetation around streams. Researchers have found that when only the overstory riparian vegetation is removed for harvest, there is less impact to streams than when understory vegetation is also removed (Bartholow 2000; Johnson and Jones 2000).

Riparian vegetation and associated stream temperatures are also disturbed by other changes in land use. Researchers at the Coweeta Hydrologic Laboratory in western North Carolina conducted a study comparing stream temperatures in an undisturbed forested watershed to stream temperatures in a farmed watershed. The farmed watershed's riparian vegetation had been replaced with a mixture of cultivated land and pasture. During one year of measurements, weekly maximum stream temperatures in the farmed watershed were found to be 6.4 °C (11.5 °F) warmer on average than the undisturbed forested watershed (Greene 1950). Under agricultural best management practices, farmers are encouraged to leave an undisturbed buffer of riparian vegetation around streams on farmland in order to alleviate negative impacts to stream temperatures.

Small-scale localized disturbance to riparian vegetation has also been found to cause detrimental effects to the riparian environment. In Pennsylvania, excessive deer-browsing of willows that were shading trout-inhabited streams caused the trout to migrate to other locations where intact riparian vegetative cover remained (Greene 1950).

2.3.3 Stream Nitrate Concentrations and Nutrient Cycling

Another anticipated impact to stream environmental conditions from hemlock loss is the alteration of nutrient cycling and stream nitrate concentrations. In areas where hemlock mortality has already occurred, the reduction of nitrogen uptake by declining hemlocks and the input of nitrogen from increased hemlock litterfall have significantly altered overall nutrient cycling (Yorks et al. 2003). As a result of these alterations, net losses of nitrogen to surface water and increased stream nitrate concentrations are expected to occur. In GSMNP, where hemlock decline and mortality has not yet occurred, the potential impacts to nutrient cycling and stream nitrate concentrations are not known. However, it is possible that impacts from hemlock loss could be similar in type to impacts observed from other forest disturbances.

During the past few decades, researchers have documented overwhelming evidence that forest disturbances, such as timber harvesting, windthrow events, and insect defoliation, can increase nutrient export to stream water and alter nutrient cycling of forested watersheds (Johnson et al. 1982; Swank 1988; Eshleman et al. 1998; Yeakley et al. 2003; Yorks et al. 2003). Nitrate (NO_3^-) is the ion that is most likely to reach elevated levels, which persist in streams for many years following disturbance (Swank 1998). Stream nitrate concentrations have increased dramatically following riparian forest harvesting (Bormann and Likens 1979; Swanson et al. 2000; Townsend et al. 2004). Stream nitrate concentrations often peak one to three years after harvesting and return to pre-harvest levels after five to ten years (Bormann and Likens 1979; Townsend et al. 2004). Following a significant blowdown of canopy trees at the Coweeta Hydrologic Laboratory in the southern Appalachians, researchers observed 500-fold increases in soilwater nitrate, four-fold increases in groundwater nitrate, and a doubling of stream water nitrate (Yeakley et al. 2003). A strong link between increased stream nitrate concentrations and the defoliation of canopy trees by insects has been documented. Eshleman and others (1998) found substantial export of nitrate from oak-dominated watersheds defoliated by gypsy moth infestation in the central Appalachians. In Japan, Ohte and others (2003) documented a three-fold increase in stream water nitrate concentrations as a result of a partial dieback of a pine forest.

In GSMNP, elevated stream nitrate concentrations as a result of hemlock decline and mortality are of particular concern. GSMNP receives some of the highest levels of atmospheric nitrogen loading in the United States, leading to the occurrence of nitrogen saturation (Johnson and Lindberg 1992). As a result of hemlock mortality, altered nitrogen cycling could increase the extent of nitrogen saturation in the Park. Nitrogen saturation occurs when available nitrogen exceeds the biotic demand and thus results in high nitrogen export fluxes (Aber et al. 1989). Streams draining nitrogen-saturated watersheds are characterized by high nitrate concentrations. According to Fenn and others (1998), watersheds draining high elevations, steep slopes, shallow soils, or old growth forest are more prone to nitrogen saturation and nitrate export than other watersheds. High elevations, steep slopes, and/or shallow soils cause rapid runoff and provide little opportunity for nutrient absorption or biological uptake. Old growth forests typically do not take up as much nitrogen as younger forests, and thus are more prone to nitrogen saturation (Foster et al. 1997). Other studies have suggested that conifer stands are more prone to nitrogen saturation than hardwood stands (Aber et al. 1995). Many of these conditions are present in hemlock-dominated watersheds in GSMNP, suggesting that these watersheds may be particularly vulnerable to nitrogen saturation.

Flum and Nodvin (1995), in a publication that assessed stream water chemistry in GSMNP, predicted that streams experiencing nitrogen saturation and stream water acidification will increase in number in the future due to continued atmospheric nitrogen loading. Increased nitrogen saturation and stream acidification caused by hemlock mortality may accelerate nitrogen saturation and stream water acidification, especially in these vulnerable hemlock-dominated watersheds.

Following the occurrence of a major disturbance, nitrate concentrations in streams have exceeded the U.S. federal drinking water standard of 10 mg/L (Riggan et al. 1994; EPA 2003). There is a legitimate concern that nitrogen-saturated, hemlock-dominated watersheds suffering from complete hemlock mortality could yield concentrations that exceed the federal drinking water standard. High nitrate levels in stream water can contribute to the eutrophication of lakes and ponds as well as pose a threat to human health via drinking water (Swanson et al. 2000). Many communities surrounding GSMNP depend on streams originating in the Park for their water supply, which would be degraded by elevated nitrate concentrations.

Researchers suggest that the best method of reducing nitrate loss to stream water is to preserve an effective riparian buffer zone (Haycock et al. 1993). The loss of hemlock from riparian forest stands in GSMNP could lead to increased nitrate export to streams and a reduction of the effectiveness of the vegetation buffer.

2.3.4. Potential Long-term Impacts from the Loss of Hemlock

To date, most of the relevant literature has focused on the immediate, short-term consequences of hemlock decline and mortality. Research on the impacts of forest disturbances on stream conditions also has focused on immediate, short term effects; very little is known about the long-term impacts of hemlock mortality on environmental conditions. Evidence suggests that nutrient cycling, ecosystem processes, and stream conditions could return to pre-disturbance levels when replacement species begin to take up nutrients again (Johnson and Jones 2000; Yorks et al. 2003). If this occurs, soil and stream water conditions could slowly return to normal levels over time, but this would depend on the rate of uptake and the types of species that emerge to replace the hemlock. Species of Birch (*Betula)*, Oak *(Quercus)*, and Maple (*Acer)* are replacing the eastern hemlock in HWA-damaged forests in south central Connecticut (Orwig and Foster 1998). In the Catskill Mountains of New York, Striped Maple (*Acer pensylvanicum)*, Red Maple *(A. rubrum)*, and Yellow Birch *(B. alleghaniensis)* are predicted to replace the eastern hemlock (Yorks et al. 2003). However, the forest composition in GSMNP is significantly different from that of forests of New York or Connecticut, and currently the species that will emerge to replace hemlock in GSMNP are not known. In GSMNP, dense thickets of Rhododendron (*Rhododendron*) are common in the understory of hemlock stands and may play a significant role in regeneration after hemlock mortality. Furthermore, high levels of nitrogen saturation, like those seen in GSMNP, could prevent nutrient cycling and ecosystem processes from returning to pre-disturbance levels by inhibiting forest regeneration (Silsbee and Larson 1982; Riggan et al. 1985; Fenn et al. 1998).

Very few studies have attempted to investigate the long-term changes to stream conditions and ecosystem processes that will occur with the replacement of formerly hemlock-dominated forest with non-hemlock forest. These long-term changes could be addressed by investigating the difference between a functioning hemlock-dominated riparian forest and a functioning hardwood-dominated riparian forest. In Massachusetts, an ongoing study has found that evapotranspiration rates differed significantly between hemlock and hardwood forests and that hemlock forests contained more soil water than hardwood forests in late summer (Hadley et al. 2005). These differences indicate that small perennial hemlock-dominated streams could become intermittent in the late summer if hemlock is eventually replaced by hardwood species.

Such changes could affect the diversity of niches available for stream biota. Snyder and others (2002) investigated the difference between streams draining hemlock and mixed hardwood forests for an assessment of the potential association between aquatic invertebrates and riparian hemlock habitats in Delaware Water Gap National Recreation Area (DWGNRA) in New Jersey. They found that streams draining hemlock forests had a greater diversity of microhabitat types and had more stable thermal and hydrologic regimes than streams draining mixed hardwood forests. Snyder and others (2002) also found significant differences in species compositions and trophic structures between hemlock- and hardwood-dominated streams. They documented that macroinvertebrate predators were more common and grazers less common in hemlock-dominated streams (Snyder 2002). Additionally, brook trout were found to be four times more abundant in hemlock-dominated streams than in hardwood-dominated streams (Snyder 2002).

However, the thermal and hydrologic regimes as well as trophic structures of hemlock- and mixed hardwood-dominated streams in GSMNP may or may not be similar to those observed in DWGNRA. The terrain and forest species composition are more complex in GSMNP than in DWGNRA. Therefore, a thorough investigation is needed in GSMNP in order to assess the long-term impacts of hemlock mortality on stream environmental conditions.

Chapter 3. Methods

3.1. Paired Watershed Methodology and Site Selection

Due to the proximity of hemlock stands to stream corridors in GSMNP, changes to stream conditions may be a significant ecological impact of hemlock mortality. This investigation is the first attempt to assess the influence of riparian hemlock stands on stream conditions and the possible impacts from hemlock loss in GSMNP. In order to isolate the effect of riparian hemlock forest on stream conditions, I compared hemlock-dominated watersheds with non-hemlock-dominated watersheds. This comparative method allows the prediction of the long-term impacts to stream conditions and ecosystem processes that will occur with the postulated replacement of formerly hemlock-dominated forest with non-hemlock-dominated forest. The difficulty in this type of comparative study lies in site selection.

Due to the complexities of natural variation in landscapes, controlling for all site factors in a comparative field study is nearly impossible (Jenkins et al. 1999). Careful site selection is imperative in order to draw strong inferences from comparative analyses. I devised a GIS-based site selection methodology, modified from a methodology developed by Young and others (2002). In this design, the influence of landscape variability was minimized so that the specific effect on stream chemistry by forest type could be detected. The overall goal of my site selection design was to select stream monitoring sites in which it would be possible to isolate differences in stream conditions and water quality due to forest type with all other factors being as equivalent as possible.

The first task of the site selection process was to obtain digital data of stream and watershed locations for GSMNP from a Digital Elevation Model (DEM). I accomplished this by using ESRI ARC GIS's Hydrology toolset to delineate streams of an appropriate flow, establish stream order, and then delineate watersheds. In order to focus on first- and second- order streams, I defined the minimum accumulation area for stream delineation as 100,000 m^2. In an attempt to examine only those watersheds with minimal human impact, I developed a GIS model to identify watersheds in which no significant historical disturbance has occurred. I used GSMNP fire history and disturbance history GIS shapefiles in the model to identify areas where fire had not occurred in the last 50 years and areas where intensive logging had never occurred. In addition, I identified all areas that do not have underlying bedrock of the shale-dominated Anakeesta Formation, which has the ability to yield sulphuric acid and significantly influence water chemistry and nutrient cycling (Flum and Nodvin 1995). The result of the model identified all areas in GSMNP where fire had not occurred in the last 50 years, intensive logging had never occurred, and Anakeesta was not the underlying bedrock. I used on-screen digitizing to select first- and second-order watersheds that met these criteria. The selection of first- and second-order streams allowed me to focus on a finer scale, avoiding the influence of the diverse mixtures of landscapes and forest types that are typically drained by larger watersheds (Swanson et al. 2000).

The result was the selection of 298 candidate watersheds draining first- and second-order streams that have minimal documented human impact. The average size of these 298 watersheds was 182 ha (450 acres). I then characterized these watersheds using nine terrain variables. Two variables, mean elevation and range of elevation, were calculated for each entire watershed area. I also calculated the mean elevation of the actual stream channel. I then calculated mean elevation, range of elevation, mean slope, terrain shape index, slope/aspect transformation, and topographic radiation index within a 100 m riparian buffer surrounding each stream channel segment. I calculated these variables within a riparian buffer in order to effectively assess the direct influence of the terrain surrounding the streams. Terrain shape index is a measure of local convexity or concavity and was derived by calculating the elevation of a point and subtracting the mean elevation of the surrounding 23 Ha. A resulting positive value indicates a convex terrain shape, such as a ridge. A negative value indicates a concave terrain shape, such as a steep gorge. A terrain shape index value near zero indicates that the terrain is planar (McNab 1989). The slope/aspect transformation index calculates slope multiplied by the cosine of aspect. The result is a continuous value from –1 to 1, which indicates the degree to which the slope is facing north (1) or south (–1) (Stage 1976). The topographic radiation index is a measure of how much solar radiation an area should receive based on its aspect. It is calculated by the following formula:

$$\text{Topographic Radiation Index} = \frac{1 - \cos\left((\pi / 180)(\text{aspect} - 30)\right.}{2}$$

The result is a continuous value from 0 to 1 in which zero indicates locations that are typically cooler and wetter while a value of one indicates locations that are typically hotter and dryer (Roberts and Cooper 1989).

I calculated all nine terrain variables for each watershed using ArcGIS's zonal statistics tool. I then analyzed the terrain statistics of the 298 watersheds using a K-means Cluster Analysis classification, which was reiterated numerous times in order to choose an appropriate number of clusters. After analyzing iteration history and terrain characteristics represented by final cluster centers, I determined that five clusters were necessary to account for the variability in aspect and terrain shape that occur in the complex topography of GSMNP. I then implemented the cluster assignments of each of the 298 watersheds into ArcGIS for further analysis. The result was the classification of all 298 watersheds into five terrain clusters, or classes (Figure 2).

I did not want to compare watersheds that were different sizes, had different geological substrates, or received different levels of atmospheric deposition. All three of these factors have the potential to influence stream water quality. For example, differences in underlying geology can influences stream chemistry (Zhi-Jun Liu et al. 2000). In a central Appalachian watershed, concentrations of Ca^{2+}, Mg^{2+}, pH, total alkalinity, and conductivity were found to be closely related to underlying carbonate bedrock (Zhi-Jun Liu et al. 2000). Additionally, streams draining watersheds that receive different levels of atmospheric deposition can have significantly different water quality properties including pH and nutrient concentrations (Flum and Nodvin 1998).

Therefore, I characterized the watersheds by an additional set of landscape variables including watershed size, level of atmospheric deposition, and geologic substrate. I classified watershed size into three classes by quantile: 1) 69–183 2) 184–299; and 3) 300–962 hectares. I created a model of probable atmospheric deposition for GSMNP that is a function of elevation and forest type (Weathers 2000). Higher elevations receive more atmospheric deposition than adjacent lower elevations due to orographic effects and cloud deposition (Lovett and Kinsman 1990). Coniferous forests receive higher amounts of atmospheric deposition because of their ability to absorb more

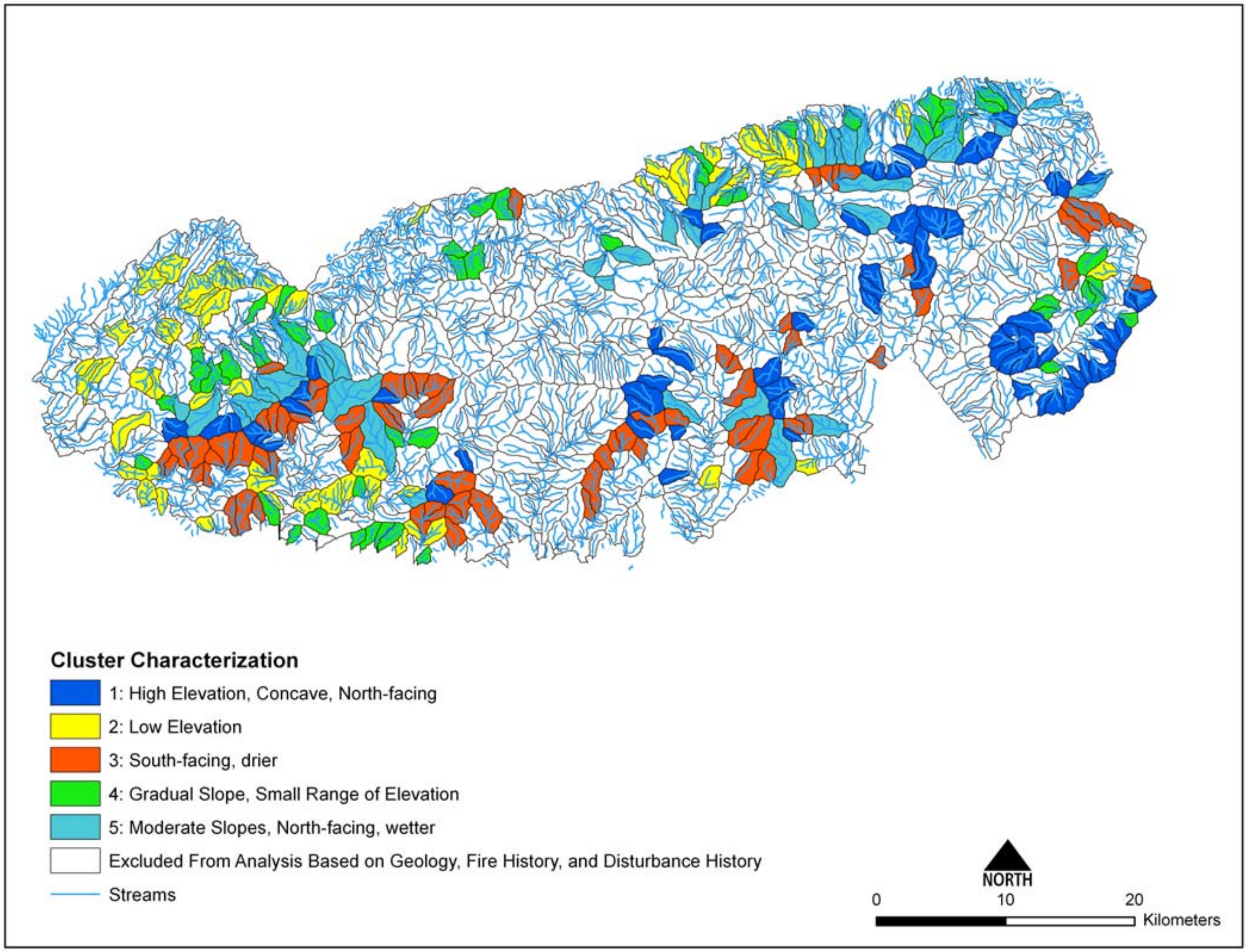


Figure 2: Map displaying the characterization of watersheds in GSMNP into five terrain classes

particles and gases than deciduous forests (Weathers 2000). I classified atmospheric deposition into five classes representing different levels of probable deposition from low to high. I derived geologic substrate from digital geologic data from the National Park Service's legacy data. Geology in the Park varies, but is dominated by metamorphosed sedimentary rocks (King et al. 1968). There are 25 different classifications of underlying bedrock identified in the GSMNP geology database. Because of the limited spatial extent of some of the geologic units included in the geology layer, I chose to omit several from this study. I classified the remaining geological types as either sandstones or siltstones and used these two categories.

I developed a model that identified watersheds that were members of the same suite of landscape classifications. For example, the model selected watersheds that were members of terrain cluster one, watershed size class one, probable atmospheric deposition level one, and had a geologic substrate of sandstone. I altered the parameters of the model and reiterated the model until every possible combination of the four variables had been selected. The result was a series of "similar watersheds grids" containing watersheds that had similar topography, geology, watershed size, and modeled atmospheric deposition. I then examined each "similar watershed grid" to ensure that watersheds did indeed have similar topography, geology, watershed size and modeled atmospheric deposition.

Within each "similar watersheds grid," I assessed watersheds for riparian hemlock cover in order to identify one hemlock-dominated watershed and a similar deciduous hardwood-dominated watershed for comparison. I defined hemlock-dominated watersheds as watersheds where canopy tree species of the riparian corridor were dominated by greater than 60% hemlock. I defined deciduous hardwood-dominated watersheds as watersheds where canopy tree species of the riparian corridor were less than 15% hemlock. Riparian corridors were established by creating a buffer of 100 m along each stream segment in each watershed. I concentrated on the canopy species of riparian corridors in order to focus on forest stands that have the most direct impact to stream environment conditions. I will refer to “deciduous hardwood-dominated watersheds” as simply “hardwood-dominated watersheds” for the remainder of this paper in order to avoid needless verbiage. I obtained presence of hemlock in GSMNP from a detailed vegetation database developed by Welch and others at the University of Georgia, which classified the occurrence of hemlock as either dominant, co-dominant, secondary, or inclusive. (Welch et al. 2002). For the purposes of this study, I only included dominant and co-dominant classified hemlock stands in order to more accurately identify the occurrence of hemlock-dominated areas on the landscape. Although the GSMNP vegetation dataset appears to accurately identify canopy trees on the landscape, it may not identify important understory species, such as rhododendron.

Once I identified hemlock-dominated watersheds, I paired each with a hardwood-dominated watershed that had similar topography, geology, watershed size, and modeled atmospheric deposition. In order to pair a hemlock-dominated watershed with a hardwood-dominated watershed that is as topographically similar as possible, I used Euclidean distance dissimilarity matrix, a statistical technique. The dissimilarity matrix is created based on user defined input variables. For this study, I used the original nine terrain variables again. The Euclidean distance dissimilarity matrix produces a table that assigns a value to the statistical distance between two cases, or for this purpose, watersheds. A pair of watersheds with a small Euclidean distance value indicates that the pair is more similar to each other in regard to the terrain variables than a pair of watersheds with a larger Euclidean distance value. Within each “similar watersheds grid,” I paired each hemlock-dominated watershed with a hardwood-dominated watershed based on the smallest Euclidean distance value found. Thus, I paired each hemlock-dominated watershed with the most statistically similar hardwood-dominated watershed. In all, I identified a total of 10 pairs of geographically similar hemlock and hardwood-dominated watersheds to serve as study sites.

Following the final selection of stream study site pairs, I conducted field checking in order to assess the validity of forest type classification and to observe any landscape considerations not apparent from digital data. For example, I measured stream flow and stream width in order to ensure that paired stream sites were as structurally and geographically similar as possible. I also assessed accessibility of paired stream sites based on travel time required and the difficulty of travel in order to reach sites. Based on this feasibility assessment, I dropped four pairs from the study because they were either difficult to access or would require too mush time to reach. Therefore, I conducted this research at six pairs of streams, or 12 watershed study sites. Locations of paired watersheds are displayed in Figure 3 and terrain statistics for paired watersheds are displayed in Figure 4.

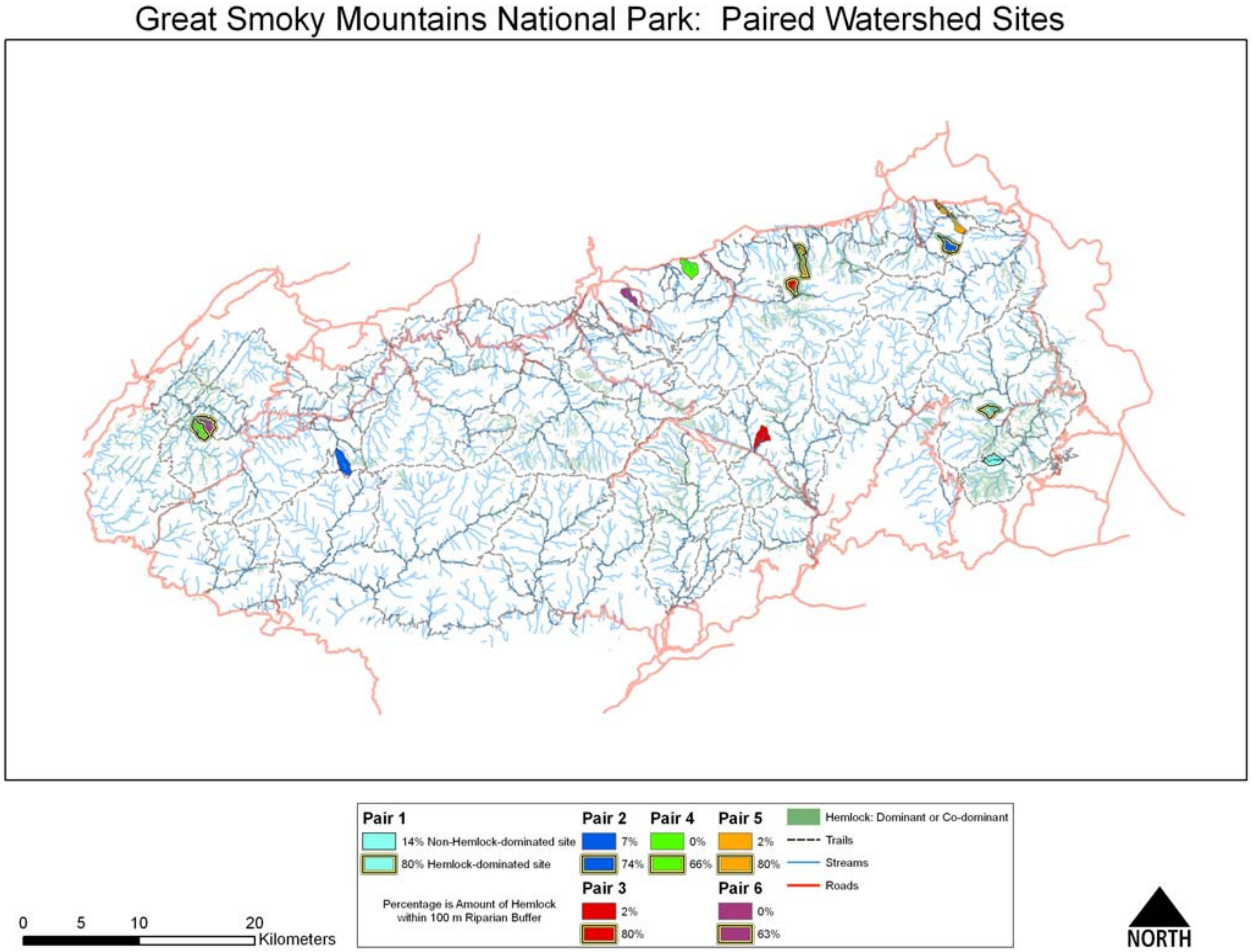


Figure 3: Paired watershed sites

Watershed Pair #	Watershed ID	Hardwood- or Hemlock-dominated	Range of elevation within riparian buffer	Mean elevation within riparian buffer	Mean slope/ aspect transformation value within riparian buffer	Mean terrain shape index value within riparian buffer	Mean solar radiation index value within riparian buffer	Slope value within riparian buffer	Mean elevation within watershed	Range of elevation of stream channel	Mean elevation of stream channel
1	116	Hardwood	185	1000	0.06	-19.58	0.38	18.33	1075	140	985
1	149	Hemlock	143	1054	-0.05	-22.21	0.4	15.55	1142	129	1047
2	161	Hardwood	490	1075	0.2	-26.55	0.36	24.38	1227	420	1055
2	10	Hemlock	396	1036	0.19	-24.16	0.41	20.99	1221	374	1018
3	137	Hardwood	410	1077	-0.18	-24.16	0.7	30.46	1221	402	1052
3	46	Hemlock	374	1051	-0.23	-27.33	0.73	26.53	1225	360	1032
4	29	Hardwood	125	496	0.09	-29.79	0.42	22.44	554	99	484
4	129	Hemlock	179	511	0.07	-31.24	0.45	29.09	575	144	495
5	5	Hardwood	618	791	0.13	-19.92	0.42	19.81	937	586	777
5	25	Hemlock	533	852	0.1	-20.69	0.42	25.53	973	521	835
6	52	Hardwood	199	626	0.14	-23.38	0.38	21.95	684	163	614
6	127	Hemlock	160	517	0.1	-23.66	0.41	26.02	575	127	502

Figure 4: Statistical table containing nine terrain variables for each paired watershed

3.2. Field Methods and Data Collection

3.2.1 Water Quality: Stream Water Temperature, Stream Nitrate Concentrations, pH

Within each of the 12 selected watersheds, I established a monitoring site approximately 20 m upstream of the watershed pourpoint (or the point of confluence at which the watershed drains into a separate downstream channel). I measured stream water temperature, pH, nitrate concentrations, and flow at this pourpoint monitoring site within each watershed. For water quality monitoring, I followed procedures outlined in the United States Geological Survey National Field Manual for the Collection of Water-Quality Data (USGS, variously dated).

I used Alpha Mach IBCod © data loggers to collect stream water temperatures measurements at hourly increments for eleven months (May 2005–March 2006). It was necessary to collect hourly water temperature measurements in order to document the daily maximum and minimum water temperatures as well as anticipated diurnal fluctuations of stream water temperature. I placed the temperature data loggers in stream riffle locations where perennial flow would be consistent. I secured and anchored each temperature data logger with plastic-coated clothesline, plastic ties, and stream rocks. I also placed stream rocks around each data logger in a manner that blocked solar radiation from directly striking data loggers and influencing stream water temperature measurements. I placed these rocks in a manner such that they would not impede the flow of water.

I visited each site every 30–60 days to download water temperature data from the data loggers and to collect additional water quality parameters. Nitrate runoff has been observed to be excessive during seasons marked by high runoff, so capturing stream conditions for all four seasons enabled observation of seasonal variability (Swanson et al. 2000). During each visit to study sites, I measured stream flow using a JDC Flowatch flow meter, pH using a Hach Sension pH meter, and I collected stream water grab samples using 60 mL polyethylene bottles. During the measurement of each stream environmental parameter, stream depth and stream flow were measured to assess the association of stream discharge fluctuations and stream condition measurements. I collected data from both the hemlock and the hardwood members of each pair either on the same day or on two consecutive days with similar weather conditions.

Temperature Data Logger Calibration

Before deploying data loggers in streams, I placed all twelve data loggers in a container of cold water. I programmed the data loggers to record temperature every minute for sixty minutes. After sixty minutes, I downloaded and examined temperature data from each data logger in order to ensure that all data loggers were calibrated to record the same water temperature (+/– 0.1 °C). On two occasions during the eleven months, I performed a validation check in the field to ensure that all temperature data loggers were still calibrated to each other. In order to accomplish this, I carried a baseline data logger to each stream site. At each site, I removed the active data logger from the stream and placed it in a container of water with the baseline data logger. I programmed both data loggers to record water temperature every minute for twenty minutes. After twenty minutes I downloaded and examined temperature data from the active data logger and the baseline data logger in order to ensure that both data loggers were calibrated to record the same water temperature. I found that all twelve active data loggers recorded the same temperature (+/– 0.1 °C) as the baseline data logger.

Laboratory Methods

I analyzed each grab water sample for stream water nitrate concentrations within 48 hours of collection using a Hach DR/2500 Spectrophotometer in the Tennessee Valley Authority Environmental Laboratory. I used a Cadmium Reduction Method for detecting nitrate, which is outlined in Hach's DR/2500 Procedure Manual (Hach Company 2004). I implemented and conducted quality control procedures based on a Hach publication for quality control in laboratories (Martin 2002). These quality control procedures included using standard solutions, sample spikes, and sample replicates in order to check the accuracy of nitrate analysis.

3.2.2. Photosynthetically Active Radiation

Changes in forest floor insolation due to declining canopy and the transition from hemlock-dominated forest to hardwood-dominated forest may have profound influences on environmental conditions in riparian hemlock forests. In order to quantify the difference in insolation on the forest floor between hemlock and hardwood forest canopies, I measured Photosynthetically Active Radiation (PAR) and canopy closure within riparian forest in three paired watersheds (3 hardwood-dominated sites and 3 hemlock-dominated sites). PAR is the range of light between 400 and 700 nanometers that is effective for photosynthesis by plants. I measured PAR using a Sunfleck Ceptometer PAR meter, which measures PAR by recording the amount of energy striking the sensors of the instrument per unit area per unit time (mmol m^{-2} s^{-1}). Canopy closure is a measure of the density of canopy cover and was measured using a spherical densiometer.

Within hardwood-dominated riparian forest sites, I identified four forest composition types: hardwood canopy with no significant understory; hardwood canopy with dense deciduous hardwood understory; hardwood canopy with hemlock understory; and hardwood canopy with dense Rhododendron understory. Within hemlock-dominated riparian forest sites I identified two forest composition types: hemlock canopy with no significant understory; and hemlock canopy with dense Rhododendron understory. In each hardwood and hemlock forest type, I established linear transects parallel to the stream channel 50 m in length. At 10 m increments along each transect, I measured PAR and canopy closure. I conducted the measurements on the forest floor at a height of 1.4 m above ground and beneath the understory canopy, if present. I then calculated an average PAR and densitometer measurement for each forest type. Methodologies for using the PAR meter are outlined in Sunfleck Ceptometer Operator’s Manual (Decagon Devices Inc. 1991); methods for using the Spherical Densiometer are in the Spherical Densiometer Instruction Sheet (Lemmon, variously dated).

PAR values vary by the angle of the sun and by atmospheric conditions at the time of measurement. Therefore, I collected PAR data on days with favorable atmospheric conditions in which there was very little cloud cover that could interfere with PAR received by the meter. With the use of a field assistant and two different PAR meters, I synchronized PAR data collection so that I measured PAR in hemlock-dominated forest types at the exact same moment as a field assistant measured PAR in hardwood-dominated forest types. This synchronization allowed for a more direct comparison of light levels in hemlock-dominated forest types and light levels in hardwood-dominated forest types without the influence of the changing angle of sun or changing atmospheric conditions. I conducted these measurements at each of the six sites once in August during leaf-on and once in January during leaf-off. Measurements in two different seasons provided an opportunity to assess differences between forest types for the leaf-on period and separately for the leaf-off period.

3.3 Statistical Analysis

I tested for significant differences of means between each pair of hardwood and hemlock-dominated sites for measured parameters including stream temperature, nitrate concentrations, discharge, and pH. I also combined all observations from the six hemlock-dominated streams together and all observations from the six hardwood-dominated streams together into two aggregate samples. I then tested for differences in mean values of each water quality and hydrology parameter for the aggregate samples.

I used the Independent Samples T-Test for determining differences in means of data that were normally distributed (nitrate concentrations, discharge and pH); and the Kolmogorov-Smirnov Z test for data that were not normally distributed (stream water temperatures). I also tested for equal variance between each pair of hemlock and hardwood sites as well as between the aggregate samples using Levene's Test for Equality of Variances. I conducted statistical analysis with SPSS statistical software.

Chapter 4. Results

4.1 Photosynthetically Active Radiation and Canopy Cover

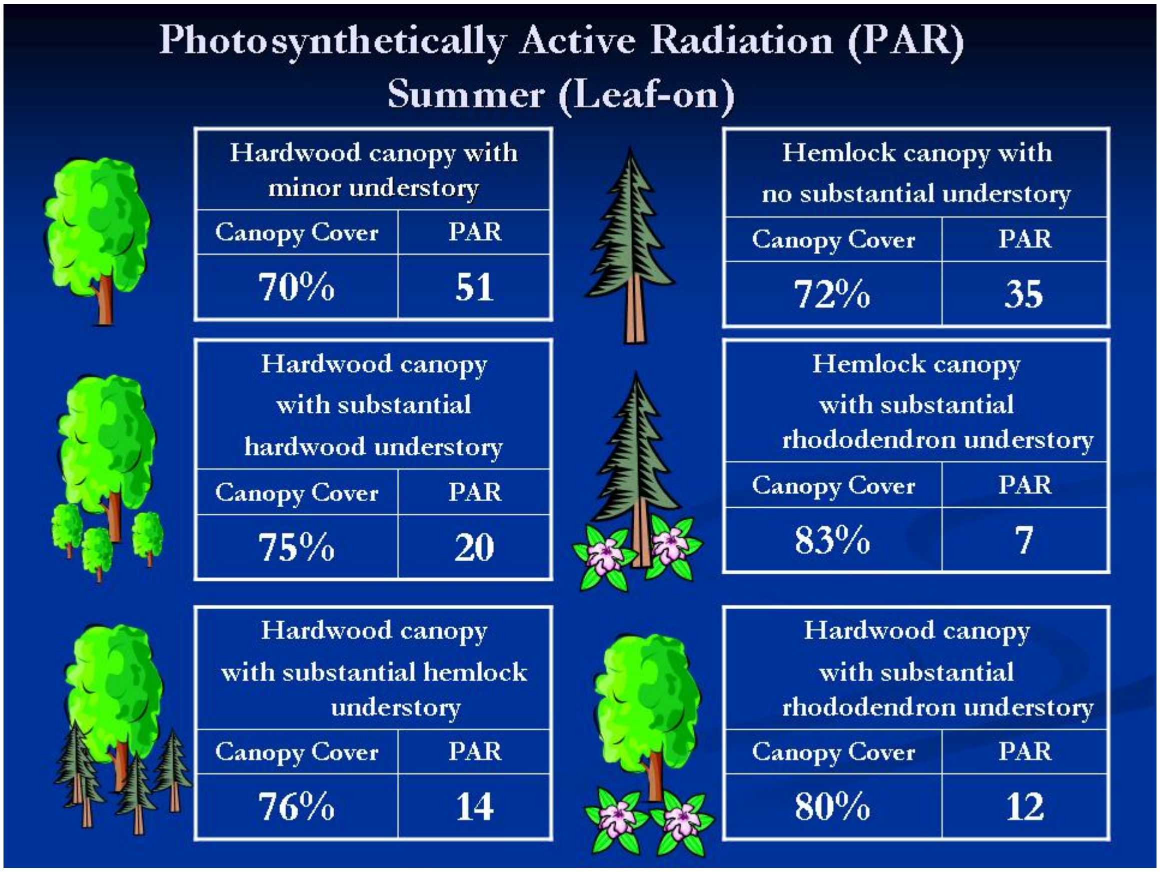
I compared PAR and canopy cover between four types of hardwood-dominated riparian forest and two types of hemlock-dominated riparian forest within three pairs of similarly structured hemlock- and hardwood-dominated watersheds (Pairs 1, 3, and 5). Measurements were conducted once in the summer with leaf-on conditions (Figure 5) and once in the winter with leaf-off conditions (Figure 6). Canopy cover and PAR measurements are shown in Figure 7.

I found that understory species composition was a strong determinant of the light conditions of the forest interior. The presence of understory species greatly influenced PAR values and canopy cover among forest types and diminished the magnitude of difference between hemlock- and hardwood-dominated canopies. Rhododendron, when present as an understory species, was a strong contributor to canopy cover and was particularly efficient at reducing PAR at the forest floor. I found that an understory composed of hemlock or Rhododendron produced the lowest PAR levels reaching the forest floor in both hardwood- and hemlock-dominated forest types. The lowest light conditions and highest canopy cover occurred in a hemlock-dominated canopy with a dense Rhododendron understory.

4.2 Water Quality and Hydrology

Stream Water Temperature

Temperature data for my 6 pairs of hemlock and hardwood-dominated streams consisted of hourly data points spanning the period from May 1, 2005 to March 1, 2006.



Photosynthetically Active Radiation (PAR)
Summer (Leaf-on)

Hardwood canopy with minor understory	
Canopy Cover	PAR
70%	51

Hemlock canopy with no substantial understory	
Canopy Cover	PAR
72%	35

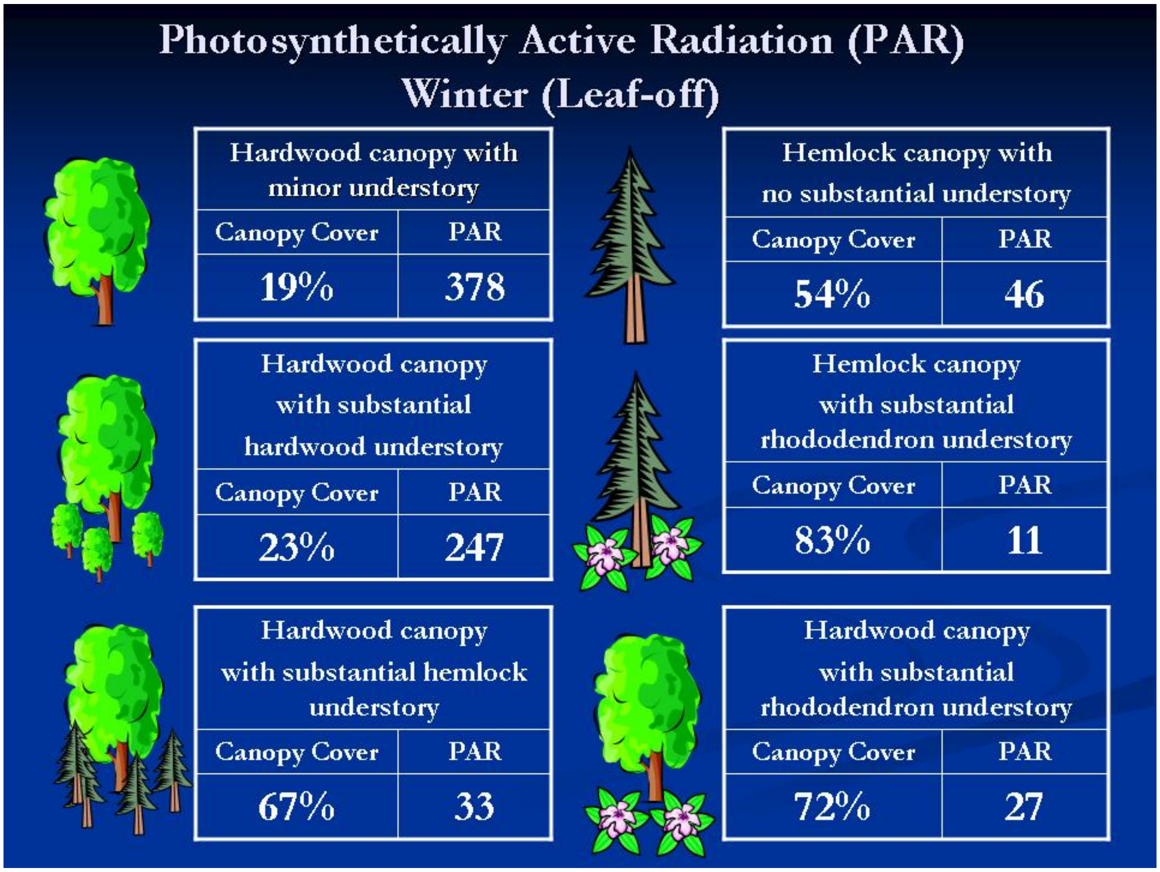
Hardwood canopy with substantial hardwood understory	
Canopy Cover	PAR
75%	20

Hemlock canopy with substantial rhododendron understory	
Canopy Cover	PAR
83%	7

Hardwood canopy with substantial hemlock understory	
Canopy Cover	PAR
76%	14

Hardwood canopy with substantial rhododendron understory	
Canopy Cover	PAR
80%	12

Figure 5: Image representing four different hardwood-dominated riparian forest types and two different hemlock-dominated riparian forest types and their associated leaf-on canopy cover measurements and PAR values. PAR values (mmol m^{-2} s^{-1}) indicate the amount of photosynthetically active radiation received at the forest floor. Canopy cover values indicate the percentage of overhead sky obscured by plant material.



Photosynthetically Active Radiation (PAR)
Winter (Leaf-off)

Hardwood canopy with minor understory

Canopy Cover	PAR
19%	378

Hemlock canopy with no substantial understory

Canopy Cover	PAR
54%	46

Hardwood canopy with substantial hardwood understory

Canopy Cover	PAR
23%	247

Hemlock canopy with substantial rhododendron understory

Canopy Cover	PAR
83%	11

Hardwood canopy with substantial hemlock understory

Canopy Cover	PAR
67%	33

Hardwood canopy with substantial rhododendron understory

Canopy Cover	PAR
72%	27

Figure 6: Image representing four different hardwood-dominated riparian forest types and two different hemlock-dominated riparian forest types and their associated leaf-off canopy cover measurements and PAR values. PAR values (mmol m^{-2} s^{-1}) indicate the amount of photosynthetically active radiation received in the forest interior. Canopy cover values indicate the percentage of overhead sky obscured by plant material.

Forest Type	Summer (leaf-on) PAR	Summer (leaf-on) Canopy Cover (%)	Winter (leaf-off) PAR	Winter (leaf-off) Canopy Cover (%)
Hardwood / minimal understory	51	69.84	378	18.88
Hardwood / Hardwood understory	20	75.04	247	23.04
Hardwood / Hemlock understory	14	76.08	33	66.72
Hardwood / Rhododendron understory	12	80.24	27	71.92
Hemlock / minimal understory	35	71.92	46	54.24
Hemlock / Rhododendron understory	7	83.36	11	83.36

Figure 7: PAR and canopy cover values for each forest type. PAR values (mmol m^{-2} s^{-1}) indicate the amount of photosynthetically active radiation received in the forest interior. Canopy cover values indicate the percentage of overhead sky obscured by plant material.

Data collection resulted in approximately 7,500 data points per stream. Comparisons of stream water temperatures between hemlock- and hardwood-dominated streams are shown in Figures 8–13. Gaps of missing data, which occurred at every stream site, can be attributed to wildlife and human disturbance of temperature data loggers. I used a motion-detection camera in order to determine the cause of temperature data logger disturbance. One of the culprits, a black bear cub, was captured on film (Figure 14). In order to avoid having the gaps of missing data influence my results, I only used temperature data for statistical analysis that I was able to retrieve for both the hardwood- and hemlock-dominated sites within a pair.

Water temperature results were mixed, with some hardwood streams having warmer mean temperatures (Pairs 1, 5, and 6), and some hemlock streams having warmer mean temperatures (Pairs 2, 3, and 4). I tested for the significance of differences between the mean recorded stream water temperatures of hardwood- and hemlock-dominated streams. The magnitude of difference in mean stream water temperatures between hemlock and hardwood ranged from to 0.02 to 0.53 °C (0.04 to 0.95 °F). Although these differences in stream water temperatures were small, I did find the differences to be statistically significant. Additionally, I found differences in stream temperatures between aggregate samples of hardwood- and hemlock-dominated streams to be statistically significant. Mean stream water temperatures among pairs are shown in Figure 15. I also examined differences in the maximum temperature and annual and diurnal ranges of temperatures between hemlock- and hardwood-dominated streams. I found no consistent pattern of maximum temperatures or ranges of temperatures occurring with forest type.

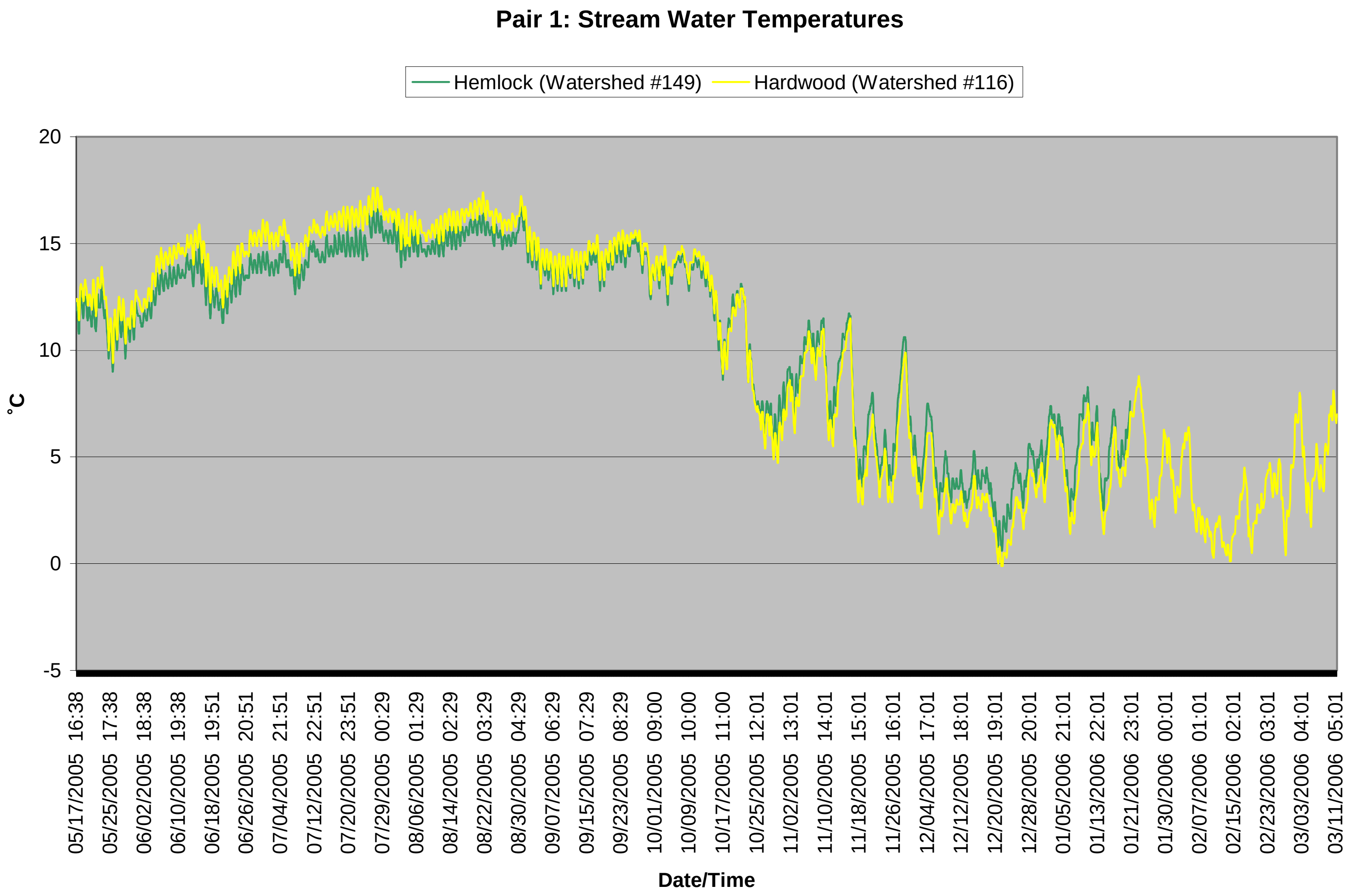


Figure 8: Stream water temperatures for pair 1

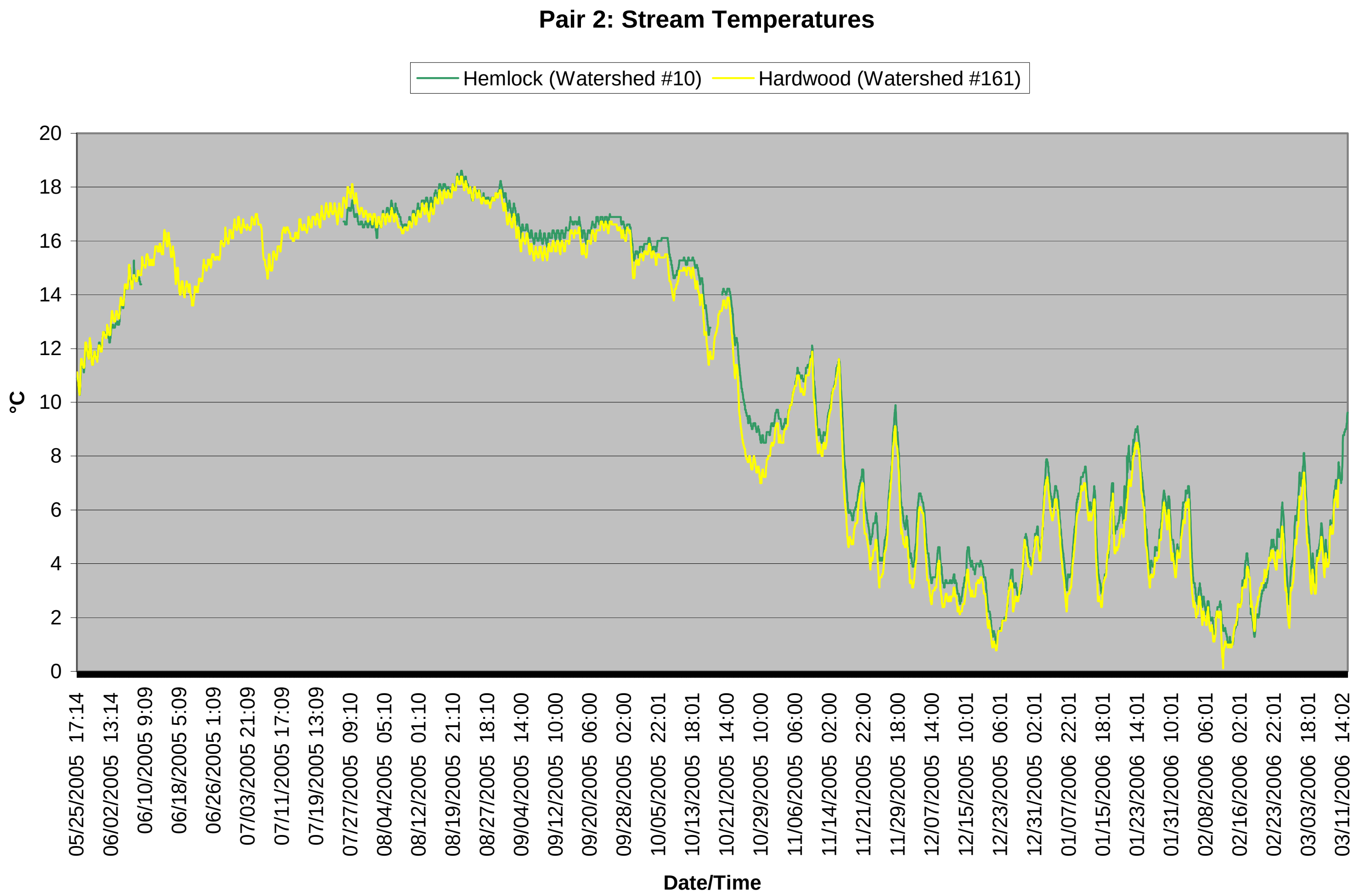


Figure 9: Stream water temperatures for pair 2

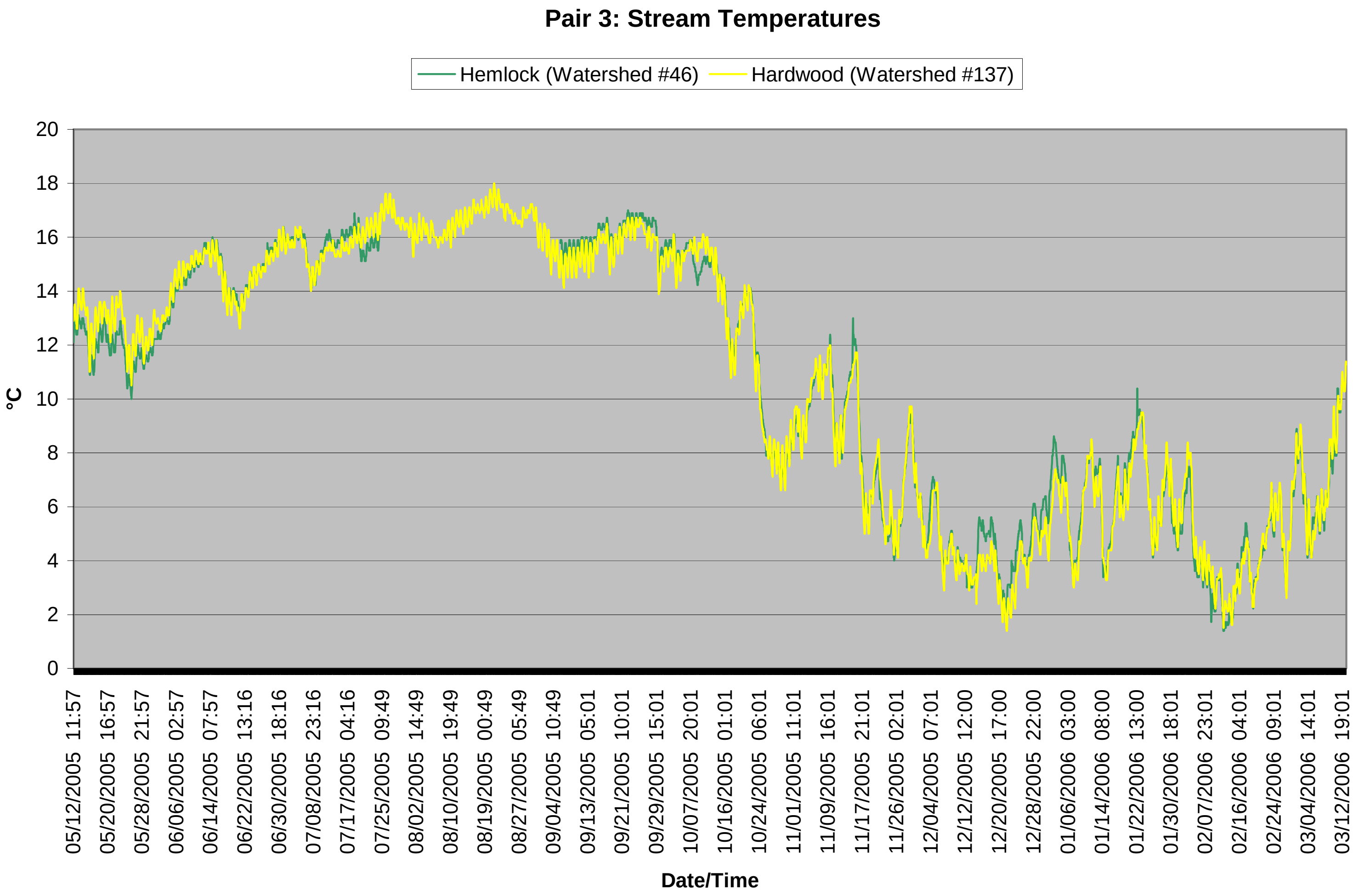


Figure 10: Stream water temperatures for pair 3

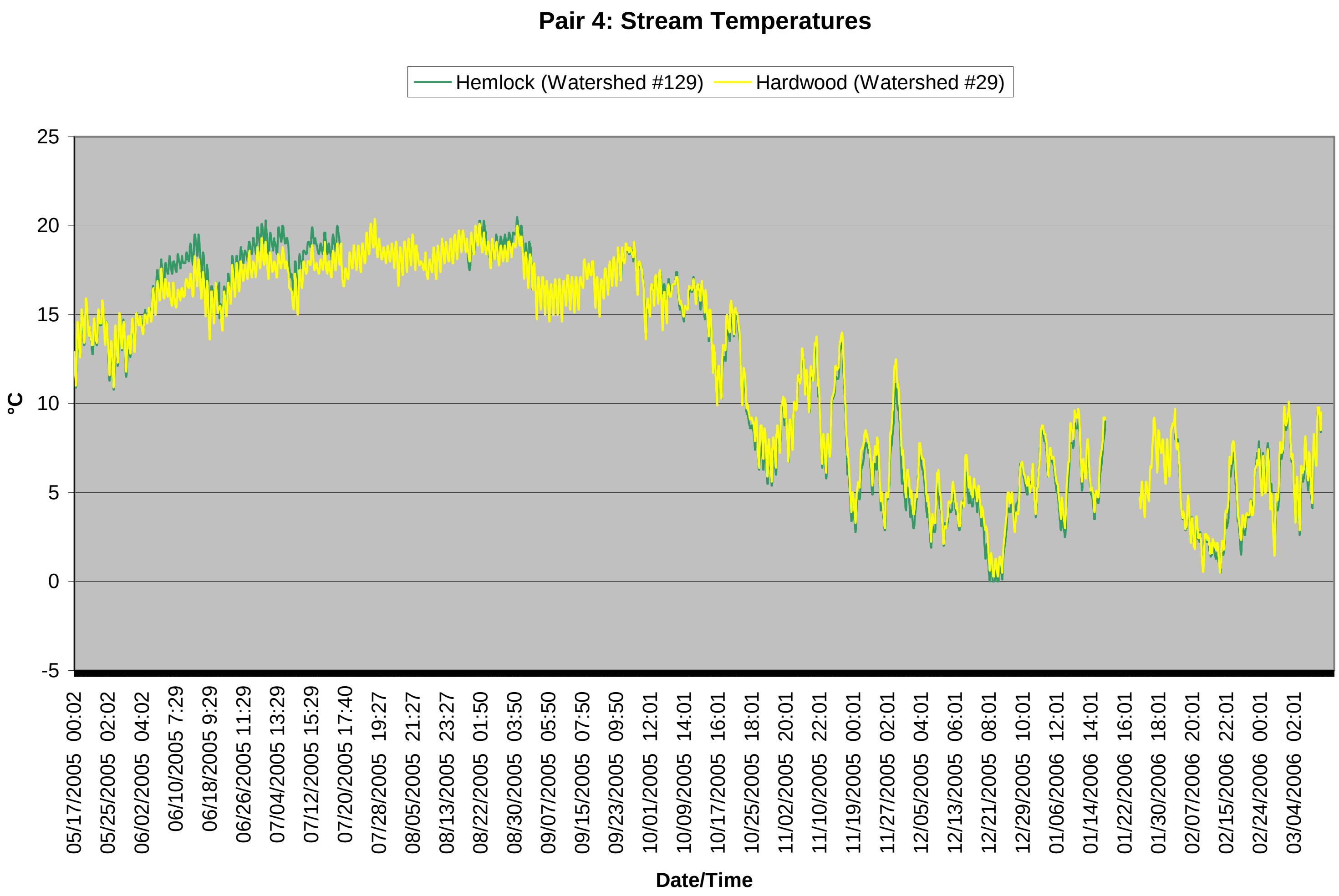


Figure 11: Stream water temperatures for pair 4

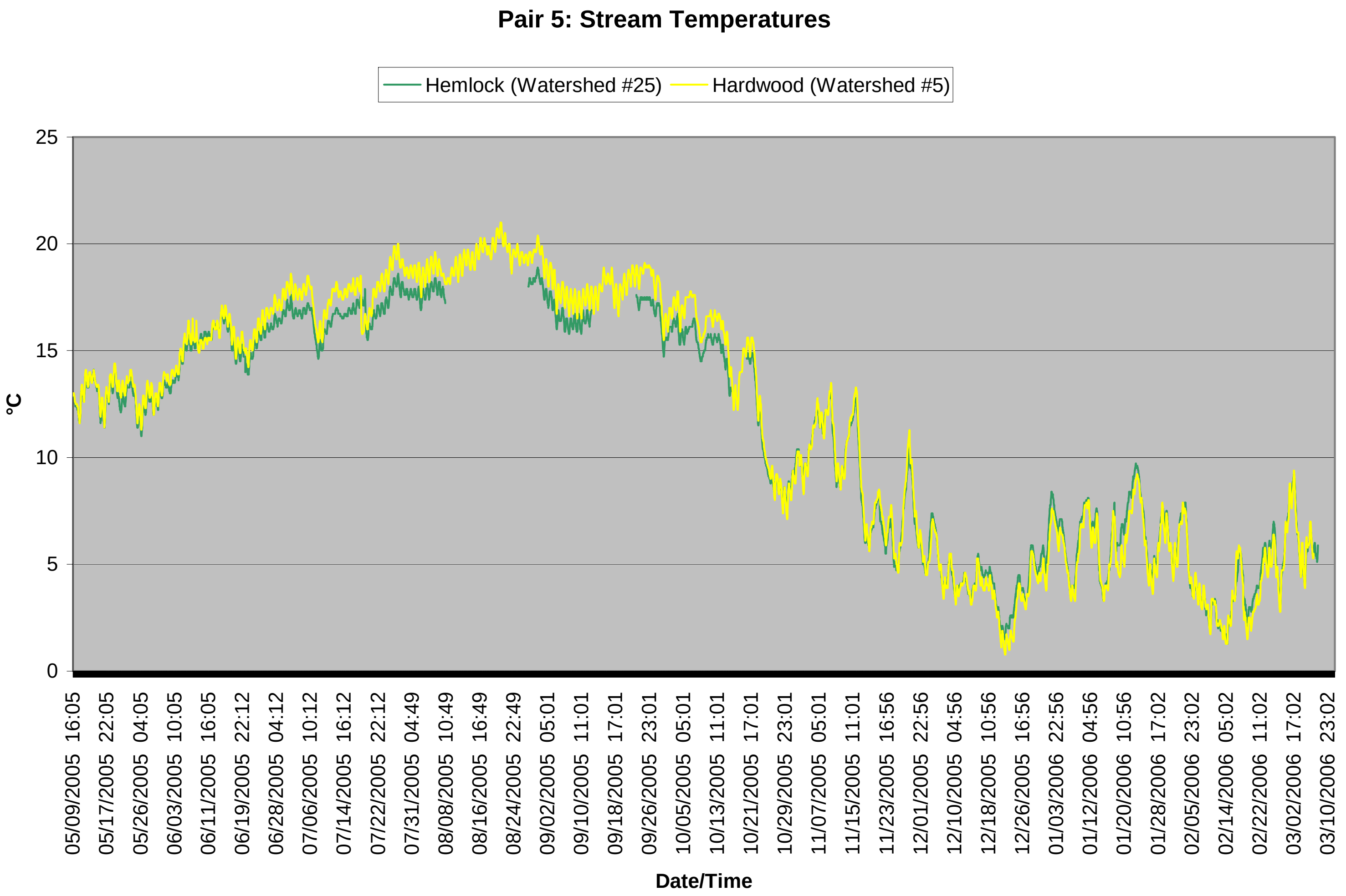


Figure 12: Stream water temperatures for pair 5

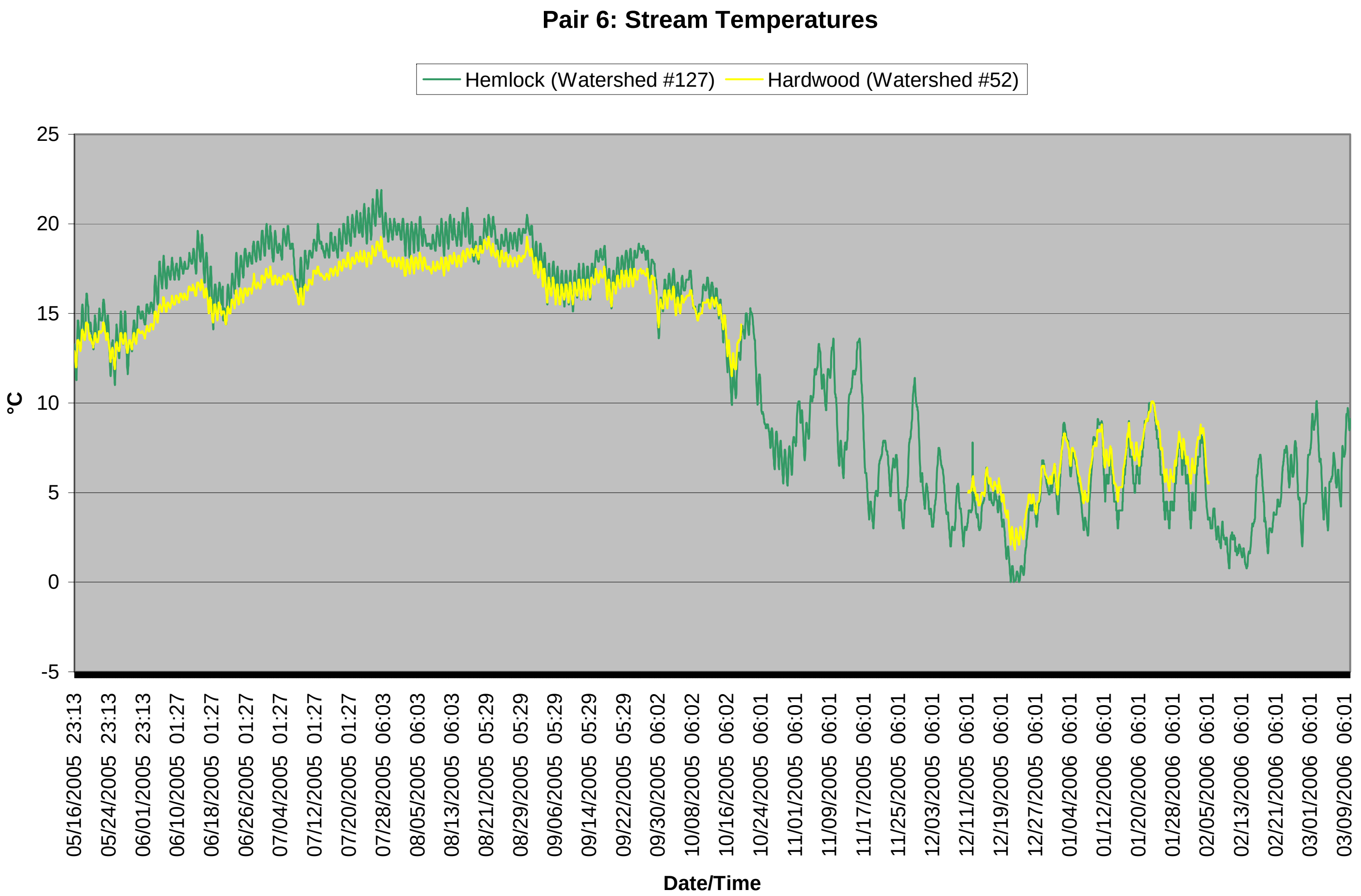


Figure 13: Stream water temperatures for pair 6

Figure 14: Photograph of black bear cub captured by a motion-detection camera. Disturbance to monitoring equipment by this bear, other wildlife, and humans caused gaps in stream water temperature data

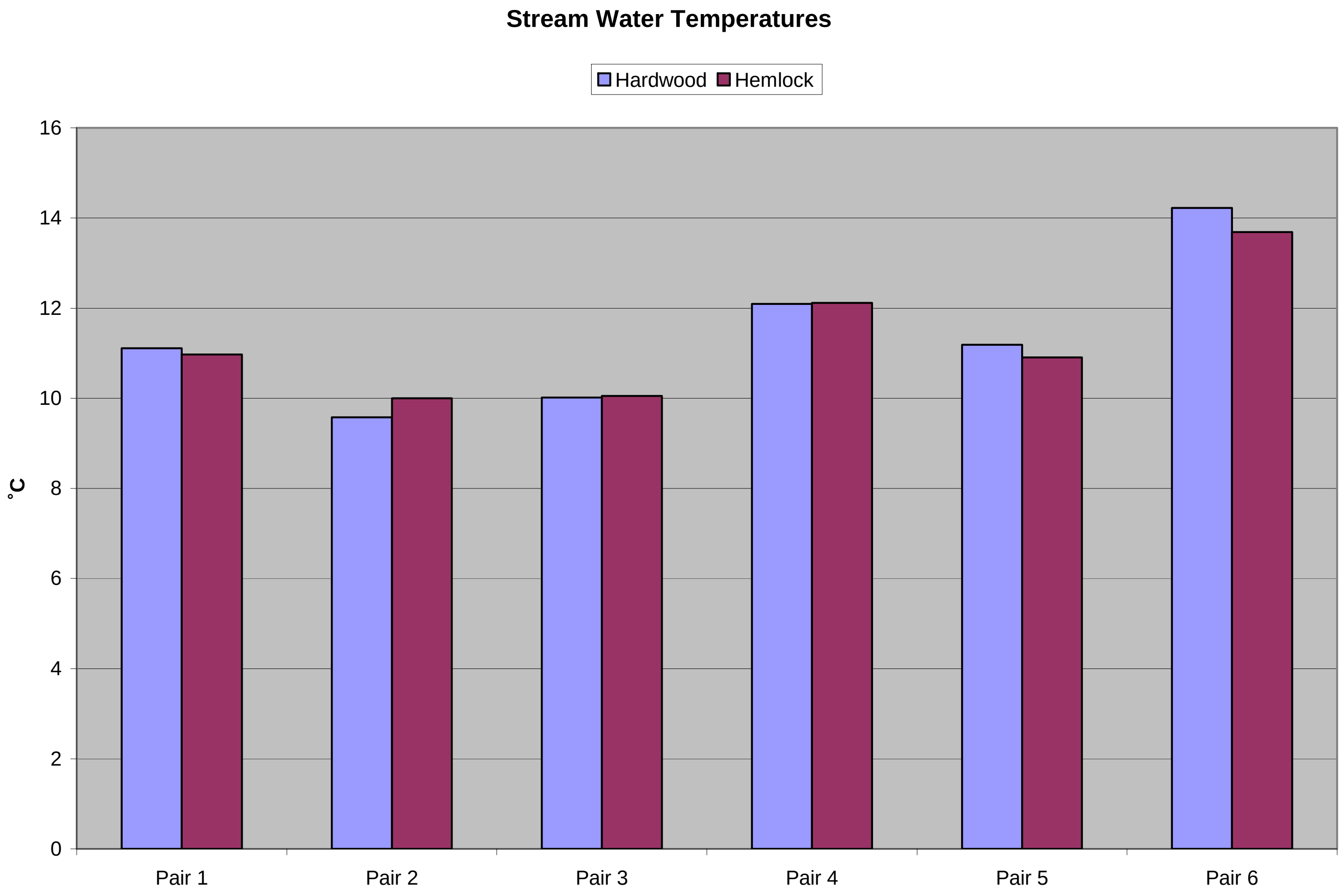


Figure 15: Mean stream water temperatures among pairs

Hemlock-dominated streams had higher maximum temperatures in pairs 2, 3, and 4 and hardwood-dominated streams had higher maximum temperatures in pairs 1, 5, and 6. In pairs 4 and 6, hemlock-dominated streams had higher ranges of stream temperatures while in pairs 1, 2, 3, and 5, hardwood-dominated streams had higher ranges of stream temperatures. Strong diurnal fluctuations were observed in both hemlock- and hardwood-dominated streams. Hemlock-dominated streams had higher diurnal ranges in pairs 1, 2, and 6 and hardwood-dominated streams had higher diurnal ranges in pairs 3, 4, and 5.

Additionally, I compared hemlock- and hardwood-dominated stream temperatures stratified by leaf-on and leaf-off conditions. For each pair I chose a single month to represent the leaf-on conditions of summer and a single month to represent the leaf-off conditions of winter. Although I generally used the month of August for summer leaf-on conditions and the month of December for leaf-off conditions, I had to choose different months for some pairs in order to avoid gaps in the data. For example, I used August and December for pairs 1, 2, and 5; late June to late July (6/24–7/23) and December for pair 3; mid June to mid July (6/19–7/18) and December; July and December for pair 5; and August and January for pair 6.

I found no consistent pattern of seasonal temperature differences occurring with forest type. In pairs 2 and 3, I found that hemlock-dominated streams had warmer mean temperatures in both leaf-on and leaf-off conditions. In pairs 4 and 6, hemlock-dominated streams had warmer mean temperatures in leaf-on conditions and cooler mean temperatures in leaf-off conditions. In pairs 1 and 5, hardwood-dominated streams had warmer mean temperatures in leaf-on conditions and cooler mean temperatures in leaf-off conditions.

Overall, I found no clear, consistent pattern of one forest type being associated with a particular thermal regime.

Stream Nitrate Concentrations

Stream nitrate concentrations were similar within pairs. For each pair, I found equal variance and no significant difference in mean nitrate concentrations between hemlock- and hardwood-dominated streams. I also found no significant difference in mean nitrate concentrations between aggregate samples of hardwood- and hemlock-dominated streams. Nitrate concentrations were low in all streams sampled, ranging from 0.01 to 0.6 mg/L with an average of 0.179 mg/L. Concentration values among pairs are shown in line graphs in Figures 16–21. Neither hemlock- nor hardwood-dominated streams had consistently higher nitrate concentrations among all pairs. Hardwood-dominated streams had higher nitrate concentrations in pairs 1 and 2, while hemlock-dominated streams had higher nitrate concentrations in pairs 3, 4, 5, and 6. Mean values of nitrate concentration among pairs are shown in Figure 22. Data for nitrate concentrations, pH, and discharge are presented in Figures 23–25.

Stream pH

Stream pH values were also similar within pairs (Figures 23-25). For each pair, I found equal variance and no significant difference in stream pH between hemlock- and

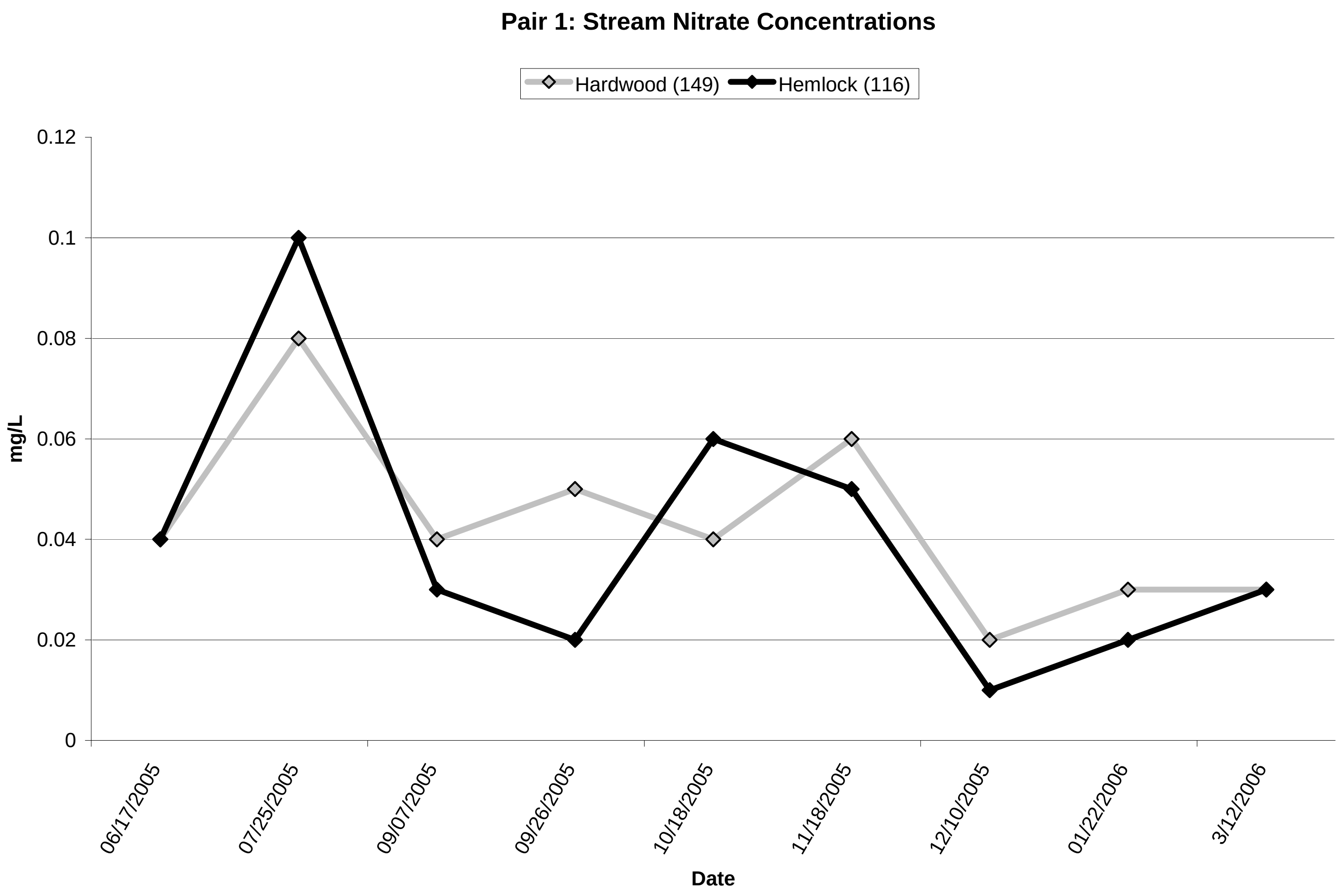


Figure 16: Stream nitrate concentrations for pair 1

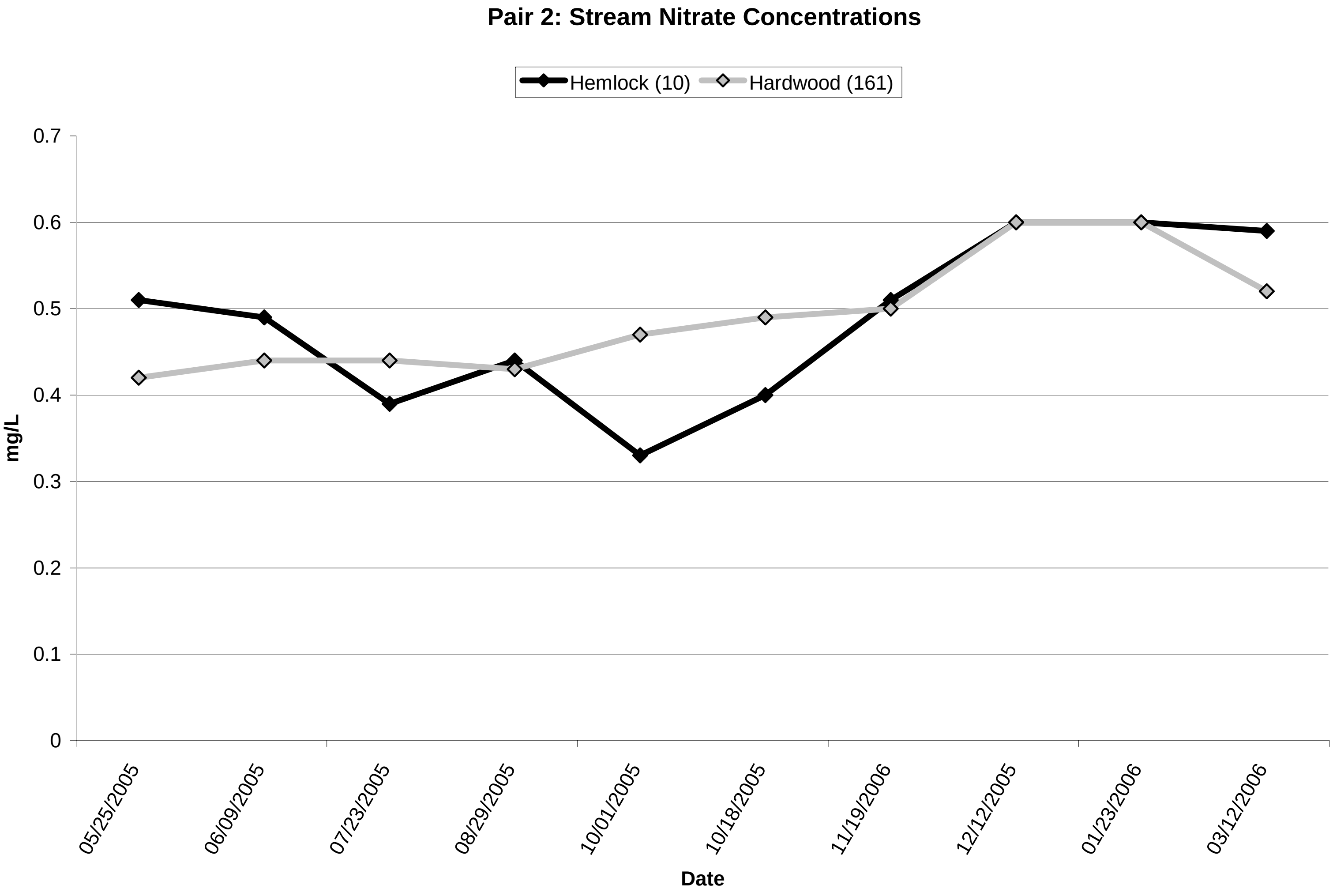


Figure 17: Stream nitrate concentrations for pair 2

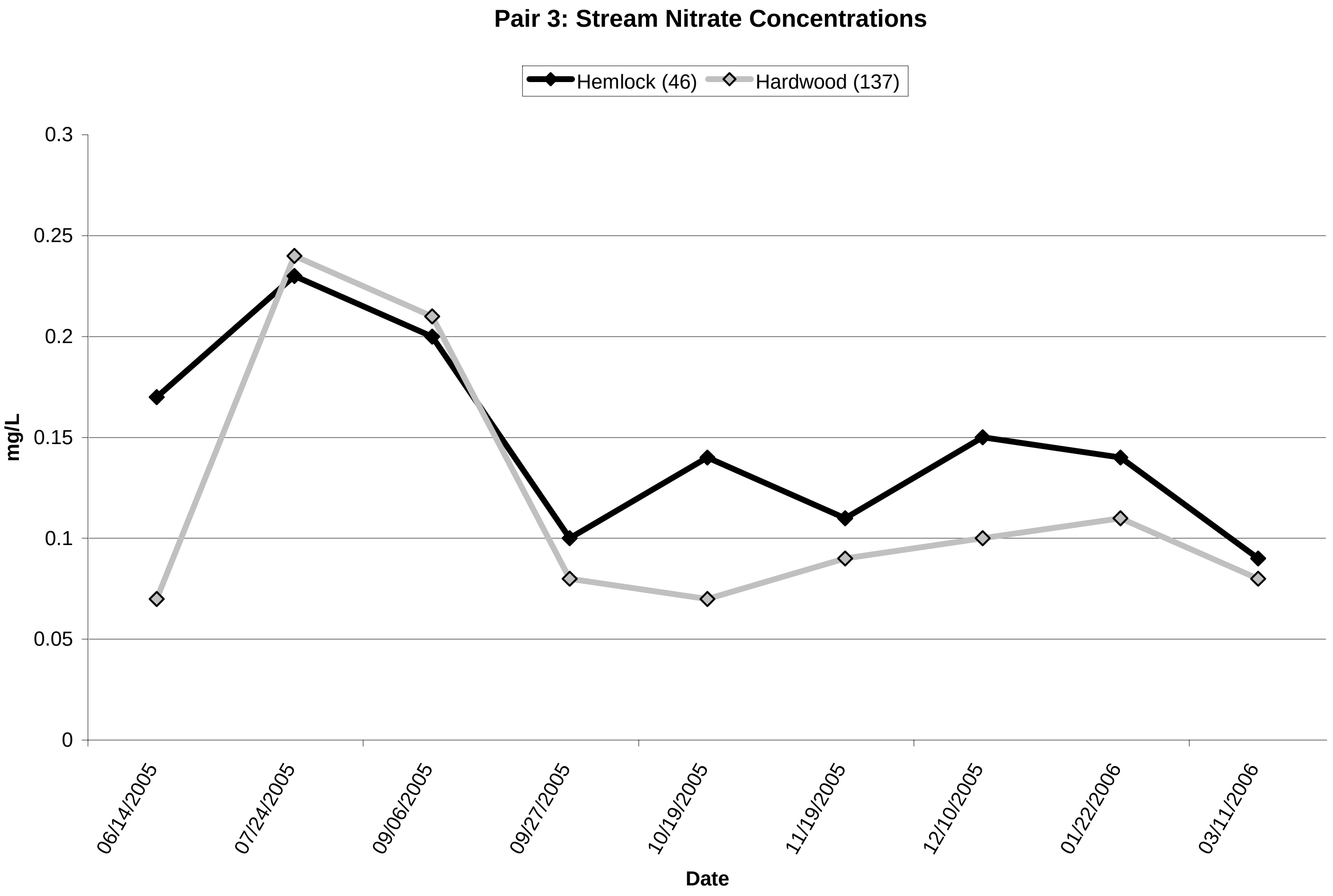


Figure 18: Stream nitrate concentrations for pair 3

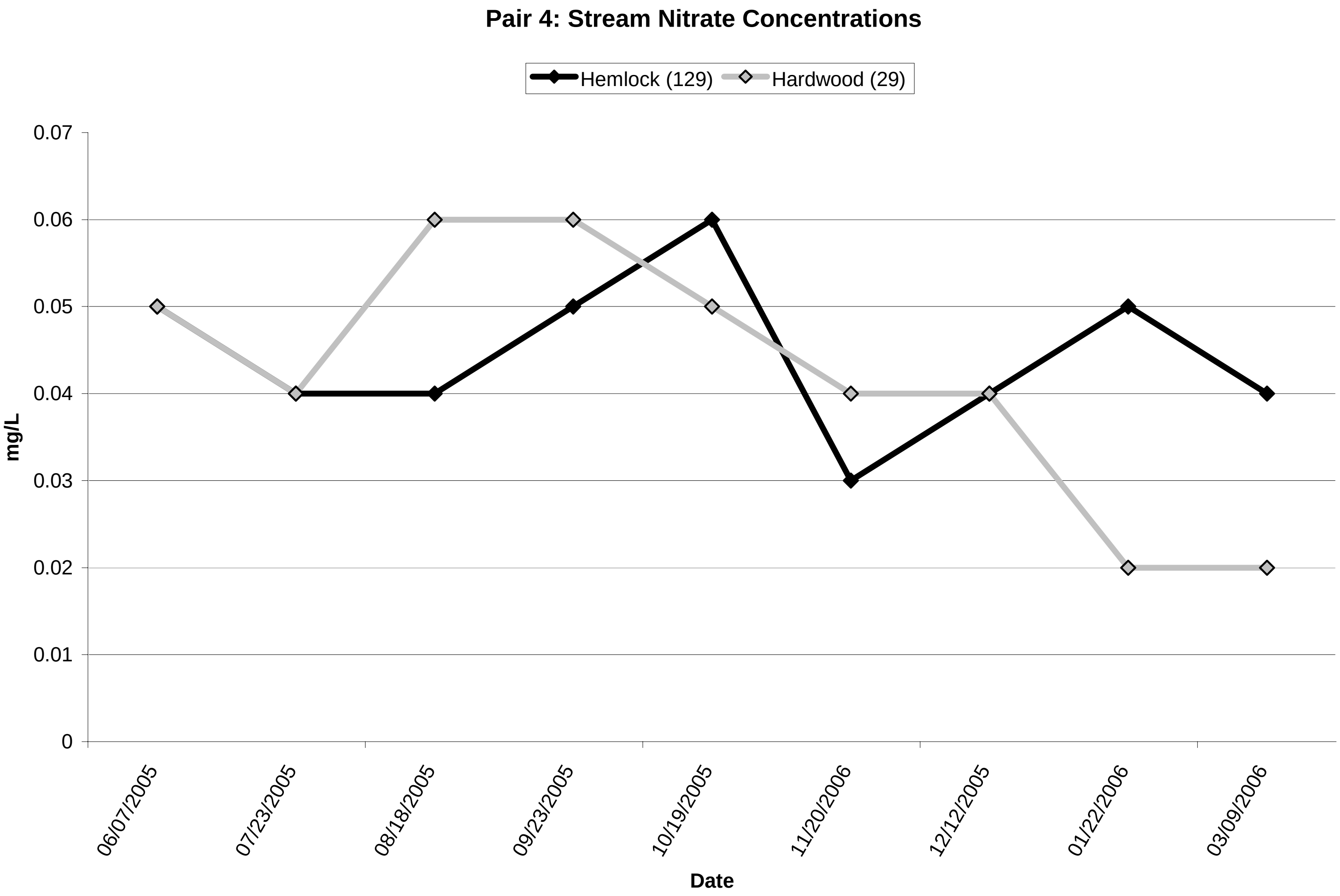


Figure 19: Stream nitrate concentrations for pair 4

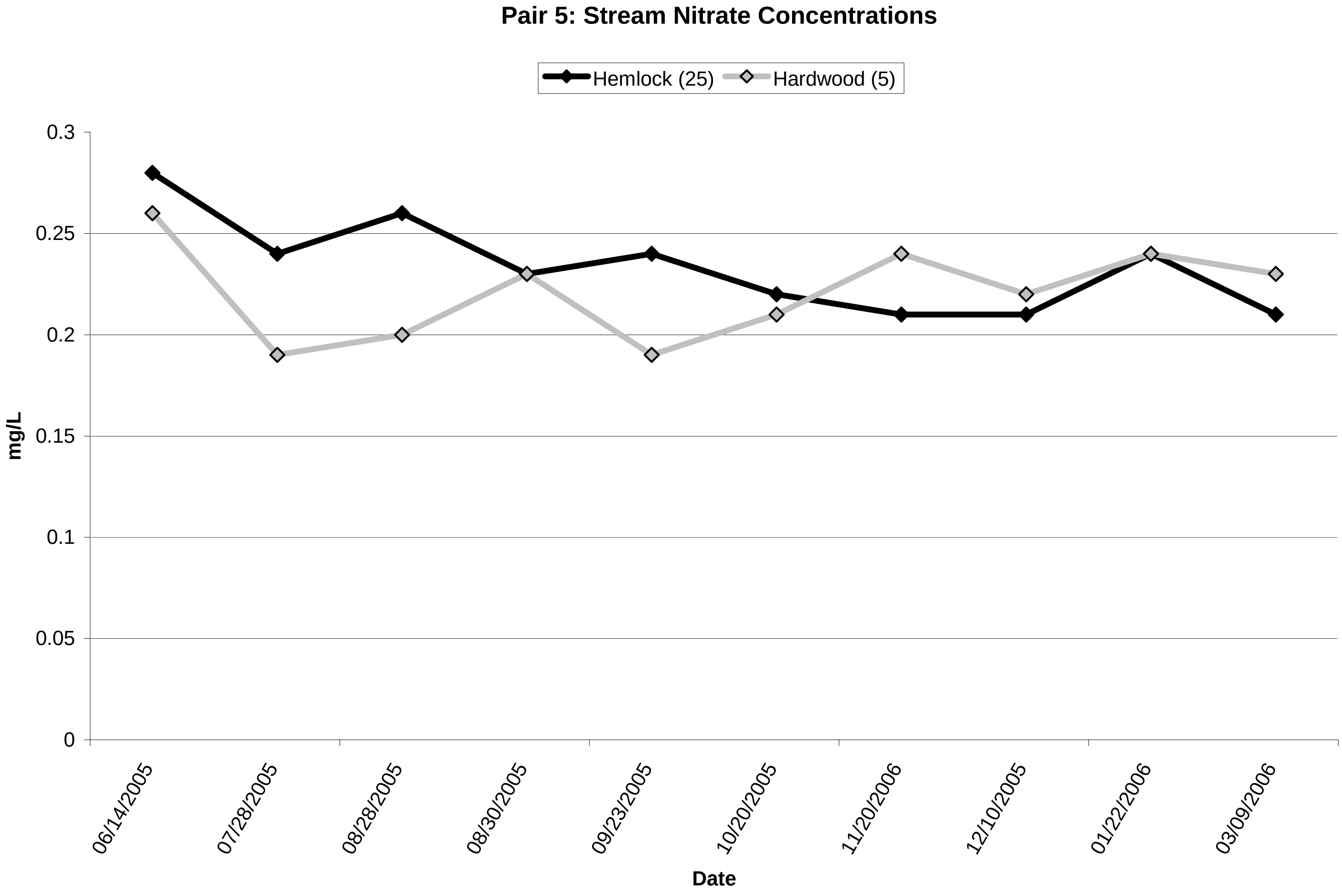


Figure 20: Stream nitrate concentrations for pair 5

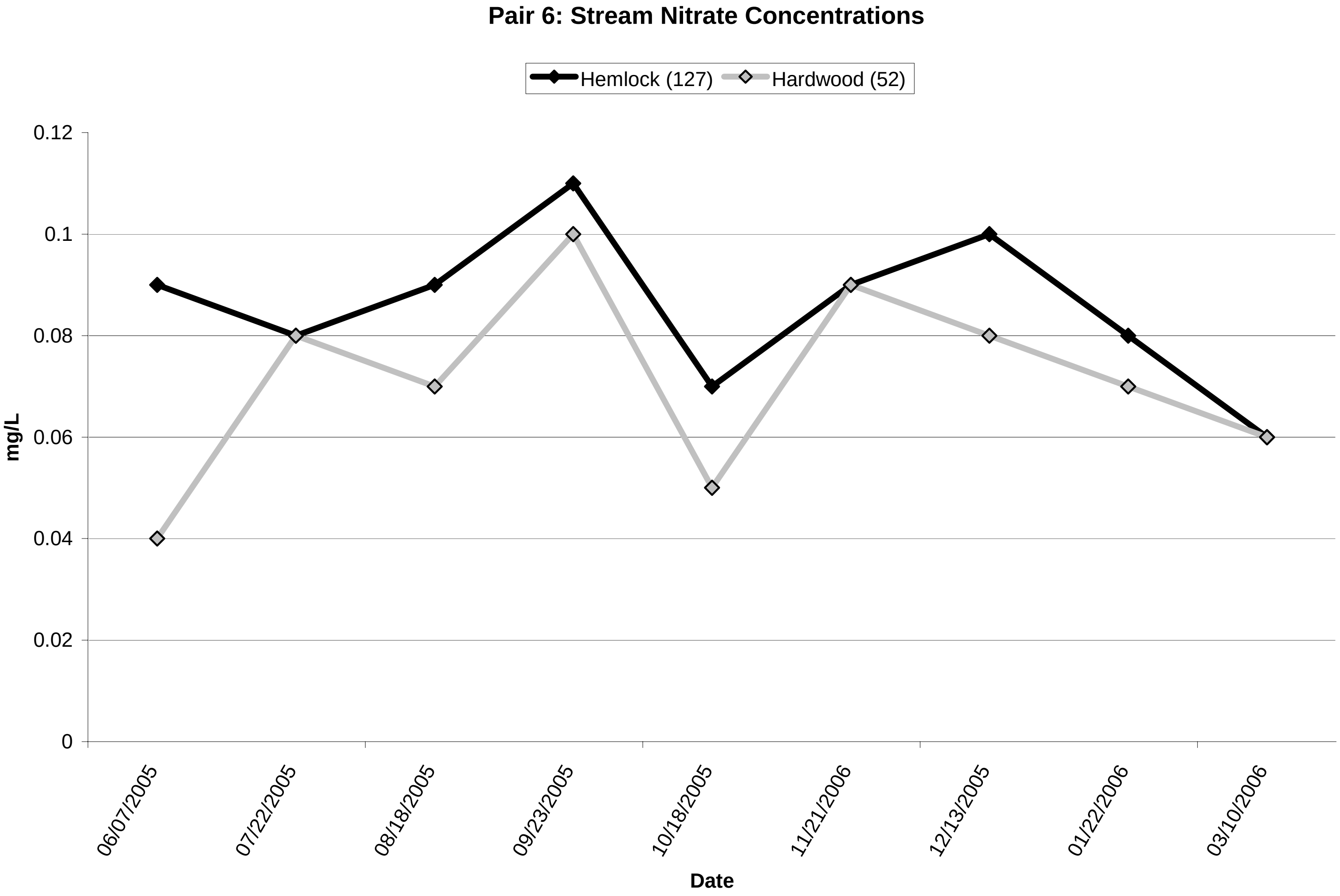


Figure 21: Stream nitrate concentrations for pair

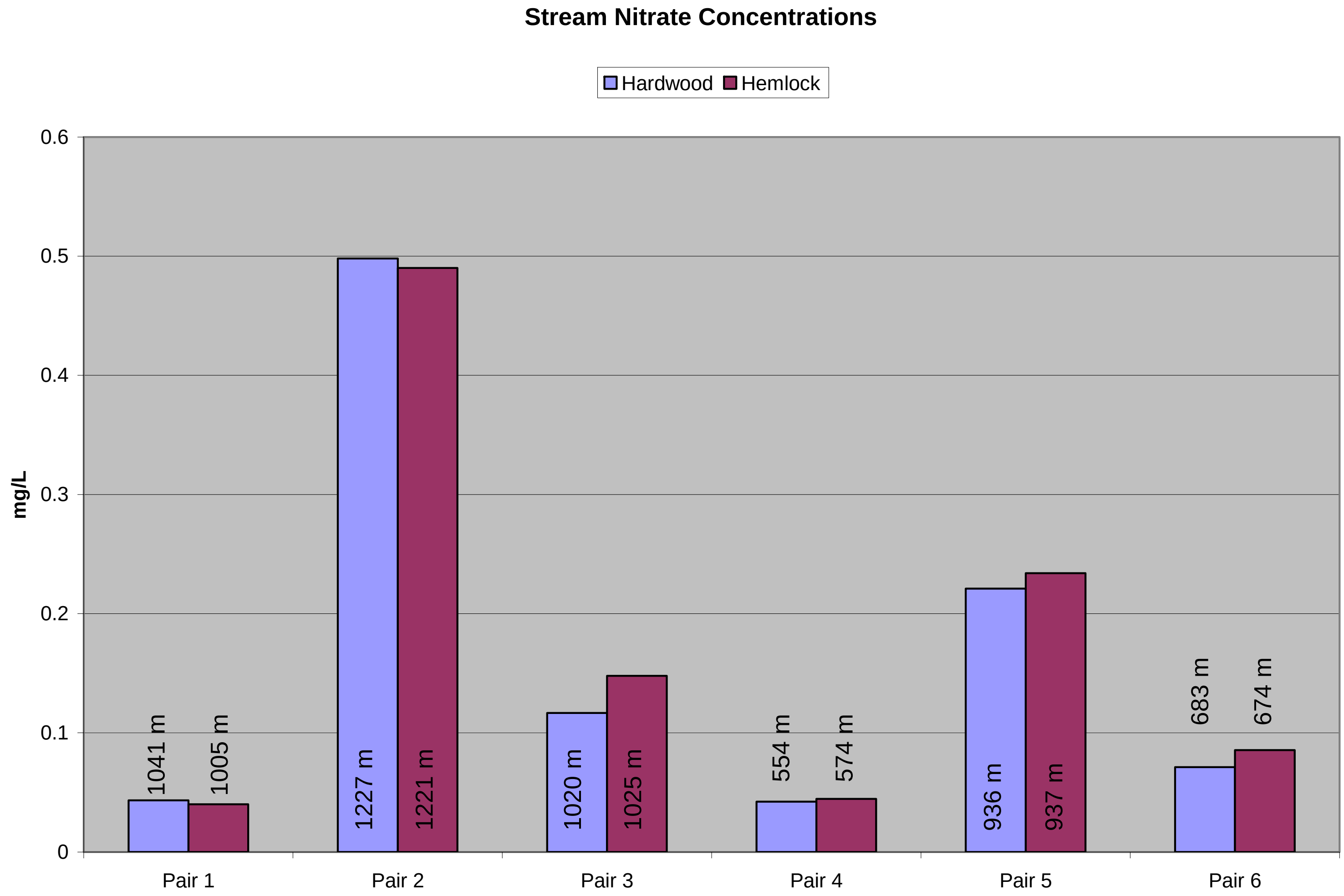


Figure 22: Mean stream nitrate concentrations among watershed pairs with mean watershed elevations in meters

Pair 1	***Stream Nitrate mg/L***		***Discharge (m3/s)***		***pH***	
Sample Date	**Hardwood** (Watershed #149)	**Hemlock** **(**Watershed #116)	**Hardwood** (Watershed #149)	**Hemlock** (Watershed #116)	**Hardwood** (Watershed #149)	**Hemlock** (Watershed #116)
06/17/2005	0.04	0.04	0.3244	0.1237	7.01	6.81
07/25/2005	0.08	0.1	0.0619	0.0565	6.84	6.79
09/07/2005	0.04	0.03	0.0525	0.0250	6.83	6.97
09/26/2005	0.05	0.02	0.0546	0.0405	6.89	6.92
10/18/2005	0.04	0.06	0.0985	0.0583	6.91	6.88
11/18/2005	0.06	0.05	0.0836	0.0760	6.89	6.87
12/10/2005	0.02	0.01	0.1658	0.0923	6.95	6.86
01/21/2005	0.03	0.02	0.0993	0.1093	6.71	6.65
03/12/2006	0.03	0.03	0.1952	0.0729	6.84	6.81
Average	0.043	0.040	0.126	0.073	6.874	6.840
Pair 2	***Stream Nitrate mg/L***		***Discharge (m3/s)***		***pH***	
Sample Date	**Hardwood** (Watershed #161)	**Hemlock** (Watershed #10)	**Hardwood** (Watershed #161)	**Hemlock** (Watershed #10)	**Hardwood** (Watershed #161)	**Hemlock** (Watershed #10)
05/25/2005	0.42	0.51	0.5549	0.2278	6.71	6.82
06/09/2005	0.44	0.49	0.1874	0.4483	6.695	6.88
07/23/2005	0.44	0.42	0.0945	0.0849	6.89	6.92
08/27/2005	0.43	0.44	0.0354	0.0327	6.83	6.87
10/01/2005	0.47	0.33	0.0292	0.0444	6.89	7.09
10/18/2005	0.49	0.43	0.0634	0.0723	6.93	6.89
11/19/2006	0.5	0.51	0.0948	0.0894	6.91	6.85
12/12/2005	0.6	0.6	0.1293	0.1911	6.97	6.89
01/23/2006	0.6	0.6	0.0791	0.0843	7.01	6.81
03/11/2006	0.59	0.57	0.0844	0.0948	6.9	6.88
Average	0.498	0.490	0.135	0.137	6.874	6.890

Figure 23: Nitrate concentrations, discharge, and pH data for Pairs 1 and 2

Pair 3	***Stream Nitrate mg/L***		***Discharge (m3/s)***		***pH***	
Sample Date	**Hardwood** (Watershed #137)	**Hemlock** (Watershed #46)	**Hardwood** (Watershed #137)	**Hemlock** (Watershed #46)	**Hardwood** (Watershed #137)	**Hemlock** (Watershed #46)
06/14/2005	0.07	0.17	0.2268	0.2153	6.742	6.81
07/25/2005	0.24	0.23	0.1374	0.1399	6.98	6.87
09/06/2005	0.21	0.2	0.0410	0.0331	6.87	6.89
09/27/2005	0.08	0.1	0.0536	0.0248	6.95	6.85
10/19/2005	0.07	0.14	0.0624	0.0579	6.99	6.93
11/19/2005	0.09	0.11	0.0742	0.0681	6.89	6.91
12/10/2005	0.1	0.15	0.0936	0.0894	7.05	7.01
01/22/2006	0.11	0.14	0.0693	0.0740	7.01	6.8
03/11/2006	0.08	0.09	0.1070	0.0959	6.78	6.81
Average	0.117	0.148	0.096	0.089	6.918	6.876
Pair 4	***Stream Nitrate mg/L***		***Discharge (m3/s)***		***pH***	
Sample Date	**Hardwood** (Watershed #29)	**Hemlock** (Watershed #129)	**Hardwood** (Watershed #29)	**Hemlock** (Watershed #129)	**Hardwood** (Watershed #29)	**Hemlock** (Watershed #129)
06/07/2005	0.05	0.05	0.1678	0.0865	6.85	6.86
07/23/2005	0.04	0.04	0.0449	0.0226	6.95	6.94
08/18/2005	0.06	0.04	0.0348	0.1181	6.88	6.87
09/23/2005	0.06	0.05	0.0159	0.0097	6.89	6.95
10/19/2005	0.05	0.06	0.0475	0.0345	6.83	6.89
11/20/2006	0.04	0.03	0.0627	0.0254	6.88	6.81
12/12/2005	0.04	0.04	0.0726	0.0432	6.78	6.98
01/22/2006	0.02	0.05	0.0256	0.0632	6.38	6.89
03/12/2006	0.02	0.04	0.0823	0.0856	6.89	6.9
Average	0.042	0.044	0.062	0.054	6.814	6.899

Figure 24: Nitrate concentrations, discharge, and pH data for Pairs 3 and 4

Pair 5	***Stream Nitrate mg/L***		***Discharge (m3/s)***		***pH***	
Sample Date	**Hardwood** (Watershed #5)	**Hemlock** (Watershed #25)	**Hardwood** (Watershed #5)	**Hemlock** (Watershed #25)	**Hardwood** (Watershed #5)	**Hemlock** (Watershed #25)
06/14/2005	0.26	0.28	0.1502	0.1159	6.79	6.83
07/28/2005	0.19	0.24	0.1487	0.2170	6.87	6.81
08/28/2005	0.2	0.26	0.0352	0.1970	6.91	6.88
08/30/2005	0.23	0.23	0.0473	0.0512	6.93	6.84
09/23/2005	0.19	0.24	0.0813	0.0226	7.01	6.82
10/20/2005	0.21	0.22	0.0362	0.0315	6.89	6.79
11/20/2006	0.24	0.21	0.0617	0.0579	6.84	6.81
12/10/2005	0.22	0.21	0.0315	0.0328	6.81	6.91
01/22/2006	0.24	0.24	0.0402	0.0378	6.86	6.83
03/10/2006	0.23	0.21	0.1050	0.0830	6.85	6.91
Average	0.221	0.234	0.074	0.085	6.876	6.843
Pair 6	***Stream Nitrate mg/L***		***Discharge (m3/s)***		***pH***	
Sample Date	**Hardwood** (Watershed #52)	**Hemlock** (Watershed #127)	**Hardwood** (Watershed #52)	**Hemlock** (Watershed #127)	**Hardwood** (Watershed #52)	**Hemlock** (Watershed #127)
06/07/2005	0.04	0.09	0.0923	0.0691	6.87	6.91
07/22/2005	0.08	0.08	0.0526	0.0236	6.85	6.84
08/18/2005	0.07	0.09	0.1651	0.0310	6.94	6.9
09/23/2005	0.1	0.11	0.0091	0.0089	6.98	6.85
10/18/2005	0.05	0.07	0.0317	0.0259	7.01	6.87
11/21/2006	0.09	0.09	0.0375	0.0473	6.81	6.79
12/13/2005	0.08	0.1	0.0476	0.0357	6.88	6.93
01/22/2006	0.07	0.08	0.0692	0.0542	6.4	6.81
03/12/2006	0.06	0.06	0.0729	0.0913	6.94	7.01
Average	0.071	0.086	0.064	0.043	6.853	6.879

Figure 25: Nitrate concentrations, discharge, and pH data for Pairs 5 and 6

hardwood-dominated streams. I also found no significant difference in stream pH between aggregate samples of hardwood- and hemlock-dominated streams. Concentrations were all very close to neutral (7.0); values ranged from 6.4 to 7.0, with an average of 6.87 for all streams. Neither hemlock- nor hardwood-dominated streams had consistently higher pH among all pairs. Hardwood-dominated streams had a higher pH in pairs 1, 3, and 5, while hemlock-dominated streams had a higher pH in pairs 2, 4, and 6. Mean pH values among pairs are shown in Figure 26.

Stream Discharge

Stream discharge was also similar within pairs (Figures 23-25). I found equal variance and no significant difference in mean stream discharge between hemlock and hardwood-dominated streams. Additionally, I found no significant difference in stream discharge between aggregate samples of hardwood- and hemlock-dominated streams. Mean discharge for all streams was less than one m^3/s, ranging from 0.0008 to 0.958 $m^3 s^{-1}$. Neither hemlock nor hardwood-dominated streams had consistently higher discharge among all pairs. Hardwood-dominated streams had a higher discharge in pairs 1, 3, 4, and 6, while hemlock-dominated streams had a higher discharge in pairs 2 and 5. Mean discharge values among pairs are shown in Figure 27.

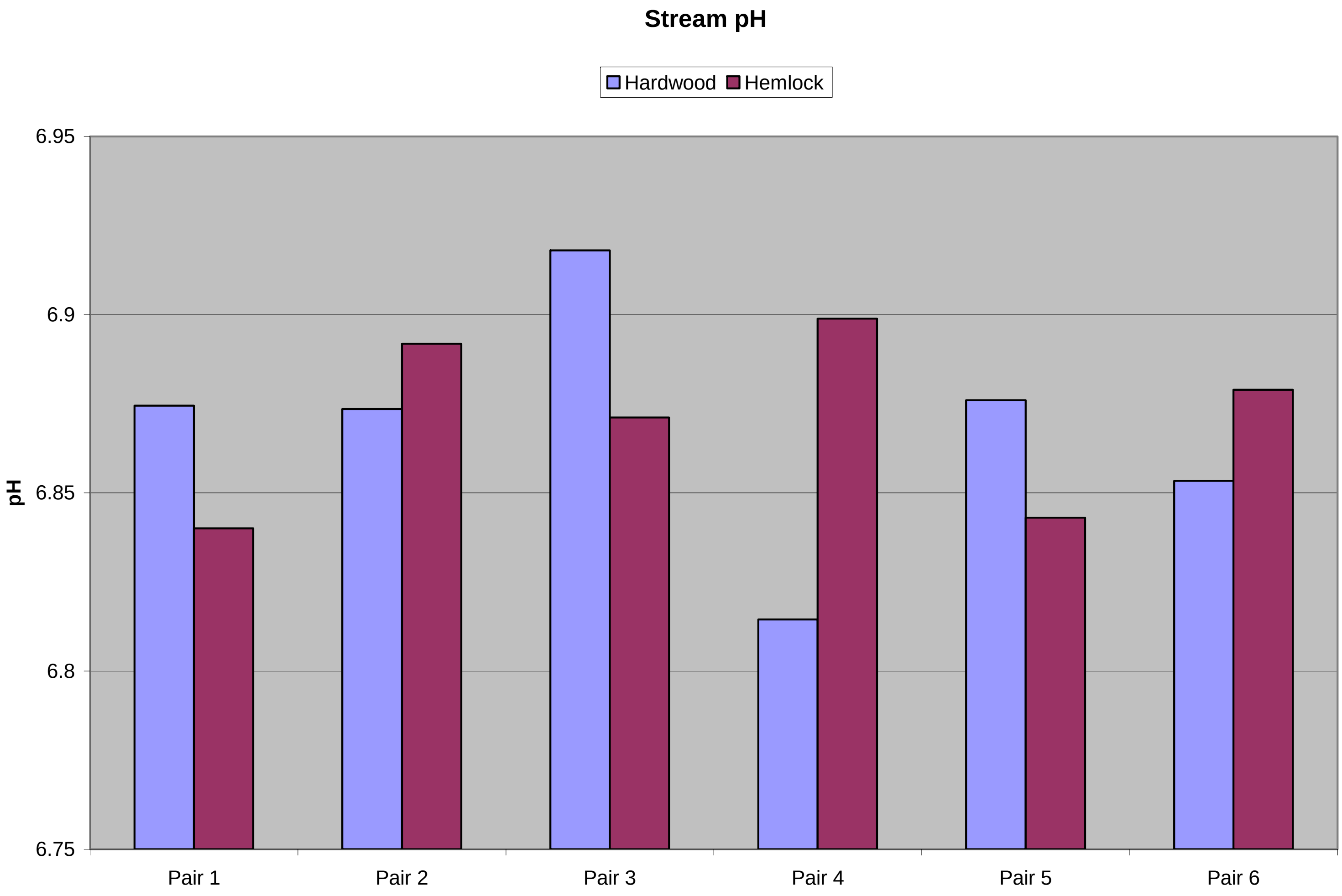


Figure 26: Mean stream pH among watershed pairs

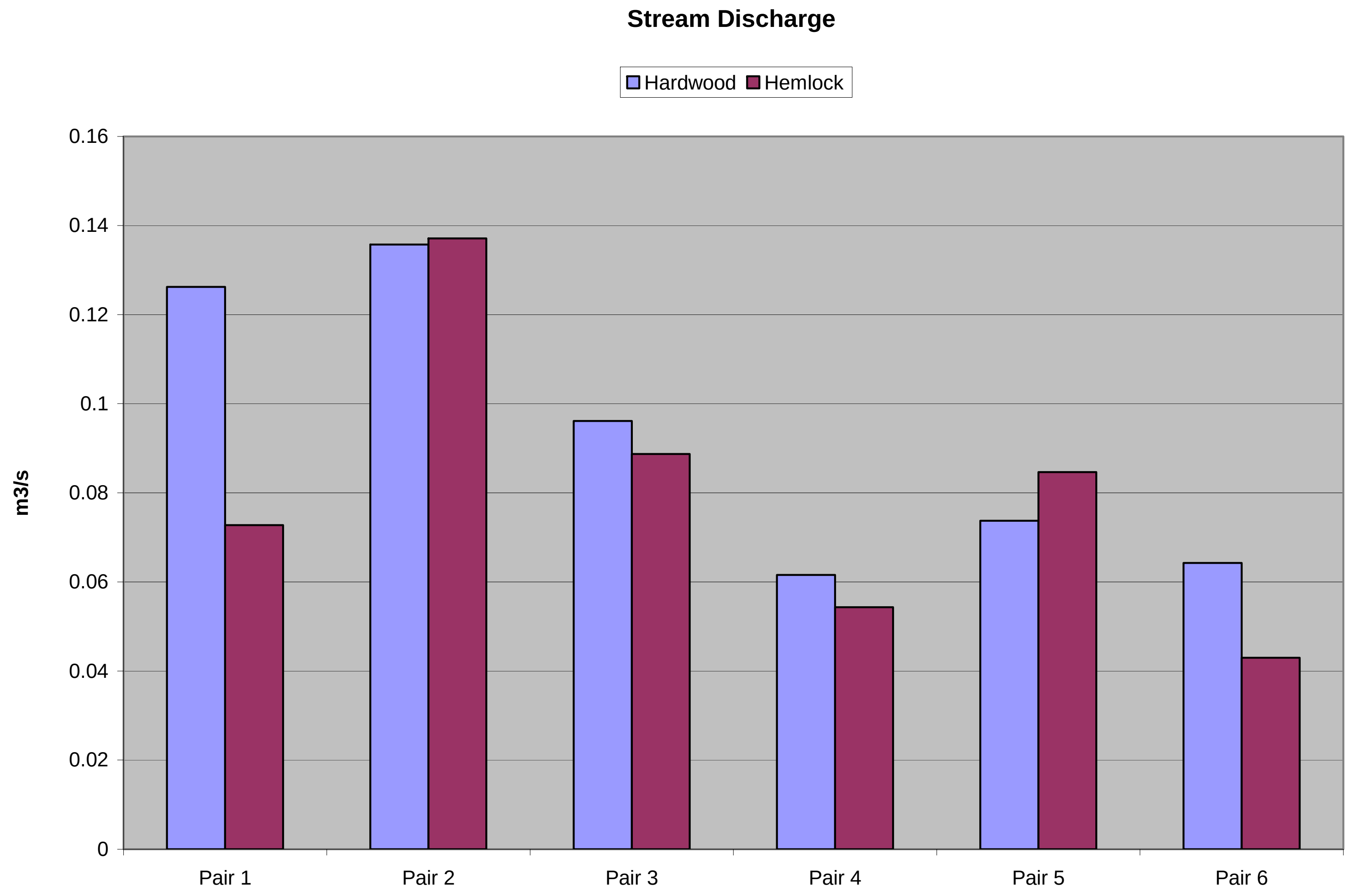


Figure 27: Mean stream discharge among watershed pairs

Chapter 5. Discussion

5.1 Introduction to Discussion

The results of this research indicate that stream nitrate concentrations, pH, hydrology, and water temperatures are similar between hemlock and hardwood-dominated streams in GSMNP. I did not find a significant influence of riparian hemlock stands on stream conditions, which implies that differences in hemlock and hardwood riparian forest types do not result in significant differences in stream water conditions of temperature, nitrate concentrations, discharge, and pH. Additionally, the results of this research indicate that if a riparian hemlock forest is eventually able to successfully make the transition to an intact riparian hardwood forest, there will be no significant difference in stream nitrate concentrations, water temperatures, pH, or discharge. Thus, if a formerly hemlock-dominated riparian forest is able to fully recover from hemlock mortality and hardwood replacement species are able to function as an intact ecosystem, long term impacts of hemlock mortality on watershed-scale stream nitrate concentrations, pH, stream temperature, and discharge will be minimal in GSMNP. These findings are discussed in detail in the following text.

5.2 Water Chemistry and Hydrology

The presence of hemlock-dominated riparian forest does not seem to have a significant influence on stream nitrate concentrations, pH, and discharge in GSMNP. I found no significant statistical difference in nitrate concentrations, pH, or discharge between streams draining hemlock-dominated forest and streams draining hardwood-dominated forest. The lack of significant difference in water chemistry between hemlock- and hardwood-dominated streams is consistent with other research (Snyder et al. 2002).

Although pH values were not statistically significantly different between hemlock and hardwood streams, I did find four of six hemlock streams to be slightly more acidic than their paired hardwood-dominated streams. I found stream water discharge to be similar between hemlock and hardwood-dominated streams. I believe the similarities of measured stream water parameters within pairs are an indication that watershed pairing was effective and successful.

Although I did not find significant differences in nitrate concentrations within pairs, I did find differences in nitrate concentrations between pairs of watersheds occurring at different elevational ranges. For example, pair 2 is located at higher elevations than pair 4 and also has greater nitrate concentrations than pair 4 (Figure 22). This elevational gradient in stream nitrate concentrations in GSMNP can be explained by the greater levels of atmospheric deposition of nitrogen that occur at higher elevations in GSMNP. These results are consistent with observations from other research, confirming that there is an elevational gradient in stream nitrate concentrations in GSMNP (Flum and Nodvin 1995).

I also found stream nitrate concentrations to be low, an average of 0.18 mg/L for all 14 study sites, which suggests that these low- to mid-elevation watersheds may not be particularly susceptible to nitrogen saturation as a result of hemlock mortality. The nitrate concentrations observed in this study are significantly lower than observed nitrate concentrations from other studies on headwater streams in GSMNP. Stream nitrate concentrations in Noland Divide, a high elevation stream in GSMNP, have been documented to be in excess of 1.5 mg/L (Flum and Nodvin 1995). The difference in observed concentrations can be attributed to elevation. The Noland Divide watershed is located at an elevation of 1676–1920 m (5498–6299 ft) and thus has higher nitrate concentrations than the watersheds chosen for this study, which are located at elevations under 1400 m (4593 ft). Because hemlock-dominated riparian forest only occurs at low to middle elevation watersheds, I chose to focus this study on low to middle elevation watersheds instead of high elevation watersheds. I chose to use low to middle elevation watersheds for study sites, instead of high elevation watersheds, since the focus of the study was on hemlock-dominated riparian forest, which only occurs at low to middle elevations in GSMNP. The results of this research suggest that because substantial hemlock-dominated riparian forest only occurs at lower elevations where nitrogen saturation is not yet occurring, long-term impacts of hemlock mortality will not cause increased nitrogen saturation. However, short term impacts could include increased stream nitrate concentrations as riparian hemlock stands decline.

5.3 Water Temperature

Although stream water temperatures were similar between hemlock- and hardwood-dominated streams, I did find averaged differences to be statistically significant. However, differences in stream water temperature are not consistent for either hemlock or hardwood riparian forest. For example, hardwood water temperatures in Pairs 1, 5, and 6 were warmer than paired hemlock water temperatures on average. In contrast, hemlock water temperatures in pairs 2, 3, and 4 were warmer than hardwood water temperatures on average. I also examined differences in mean temperatures between hemlock- and hardwood-dominated streams stratified by leaf-on summer conditions and leaf-off winter conditions. I found no clear consistent pattern of the seasonality of stream temperatures for either hemlock or hardwood riparian forest.

Additionally, the magnitude and timing of maximum stream water temperatures are also not consistent with one particular forest type. Hemlock streams reached higher maximum temperatures in pairs 2, 3, and 4 while hardwood streams reached higher maximum temperatures in pairs 1, 5, and 6. Maximum temperatures were reached in August for pairs 2, 3, and 5, while maximum temperatures were reached in July for pairs 1, 4, and 6. Thus, the presence of hemlock or hardwood riparian forest does not appear to exert a strong, consistent signal on thermal regimes of headwater streams in GSMNP. The inconsistency in the results suggests that other landscape variables, such as the influence of groundwater or understory species, may exert more control on stream temperatures than differences between hemlock and hardwood forest types.

Although statistical analysis indicates that differences between hemlock and hardwood stream temperatures are significant, it is important to put these data into perspective and ask if the differences are “significant” in a practical sense. In this case, stream ecology is the subject of concern. So the question should be whether these differences are ecologically “significant.”

The average difference in stream water temperatures between all hardwood- and hemlock-dominate streams in this study is 0.43 °C (0.78 °F). Additional research is needed on multiple aquatic species native to headwater streams in GSMNP to determine whether an annually-averaged difference in mean water temperature of 0.43 °C (0.78 °F) is ecologically important.

For native brook trout populations in GSMNP, elevated stream water temperatures in the summer pose the greatest threat. Green (1950) reports that the ideal water temperature for eastern brook trout is 18.9 °C (66 °F) and the maximum limit is 23.9 °C (75 °F). Of the twelve headwater streams monitored for water temperature in this study, not one reached the maximum temperature limit for brook trout (23.9 °C). With this in mind, hemlock and hardwood-dominated headwater streams should both provide adequate habitat for brook trout when considering maximum stream temperatures alone. Therefore, I suggest that if hardwood species are able to replace hemlock in GSMNP and streams are able to recover from short-term impacts, long-term impacts to brook trout habitat will be minimal. However, as hemlocks decline in GSMNP, I hypothesize that short-term elevated stream temperatures could occur. Additional research is needed to determine whether brook trout populations could sustain temporary unfavorable water temperatures until formerly hemlock-dominated streams return to stable conditions.

5.4 Conflict in Literature

Some results presented in this thesis differ from the results of a similar study assessing differences between hemlock- and hardwood-dominated streams in Delaware Water Gap National Recreation Area (DWGNRA), New Jersey. Snyder and others found that in DWGNRA, hemlock-dominated streams had more stable thermal and hydrologic regimes than hardwood-dominated streams. In GSMNP, I found thermal and hydrologic regimes to be similar between hemlock and hardwood-dominated streams, with no clear consistent pattern of differences between the two forest types.

The contrasting results can likely be attributed to differences in terrain and forest species composition between GSMNP and DWGNRA study sites. DWGNRA is a linear Park straddling the Delaware River with an elevational range of 84 to 490 m (275–1607 ft) (Snyder et al. 2002). GSMNP is larger in size and has more complex terrain and drainage patterns and has an elevation of 256 to 2024 m (840–6643 ft). Due to latitudinal and elevational differences between the two sites, the distribution of hemlock forest also differs. Synder and others report that when hemlock occurs in DWGNRA, it comprises as much as 77% of the basal area in stands (Synder et al. 2002). In GSMNP, riparian hemlock is patchy in distribution and often is mixed with northern and cove hardwood species. Riparian hemlock in GSMNP is not continuous along the entire course of a stream, but occurs in sheltered patches intermixed with hardwood species. In a current, ongoing study, Kincaid found that in GSMNP riparian hemlock stands are more species rich than hemlock stands that occur on upper elevational slopes (Kincaid 2005, personal communication).

Additionally Kincaid found that riparian hemlock stands in GSMNP have dense understories of Rhododendron (*Rhododendron maximum*) thickets that are not as common in hemlock stands located on upper elevational slopes. The presence of dense Rhododendron thickets in the understory of riparian hemlock and hardwood forests in GSMNP may have a significant impact on thermal and hydrologic regimes of headwater streams in GSMNP. Rhododendron are able to extend out and over stream surfaces, sometimes creating a continuous canopy above the stream water surface. Rhododendron, like hemlock, is evergreen and provides perennial shade for stream water. In this research, I found that the lowest levels of photosynthetically active radiation and the highest canopy closure measurements were recorded beneath Rhododendron. These observations suggest that riparian Rhododendron may play a more important role in moderating stream temperature than riparian hemlock. Additionally, Rhododendron often is associated with deep, slowly decomposing litter on the forest floor, similar to the litter beneath hemlocks (Romancier 1971). Furthermore, since Rhododendron shares somewhat similar attributes with hemlock and it occurs almost ubiquitously as a dense understory species in both hemlock and hardwood-dominated riparian forests, I hypothesize that Rhododendron dampens the otherwise unique influences of hemlock and hardwood forest types on riparian environmental conditions.

I hypothesize that the small size, patchy distribution, and mixed-species composition of riparian hemlock stands, along with the presence of understory Rhododendron, limit riparian hemlock's effects on the thermal and hydrologic regimes of headwater streams in GSMNP. These same limitations may not occur in DWGNRA and other more northerly locations and would thus explain the discrepancy of findings between DWGNRA and GSMNP.

5.5 Immediate Short-Term and Eventual Long-Term Impacts of Hemlock Mortality

Immediate, short-term impacts of hemlock mortality

The response of stream conditions to hemlock mortality will occur at two different temporal scales: 1) The short-term, immediate response of stream conditions to hemlock decline and mortality; and 2) The long-term, eventual response of stream conditions to the replacement of hemlock-dominated forest with hardwood-dominated forest. Although the type and extent of immediate impacts to water quality caused by riparian hemlock decline and mortality in GSMNP is unknown, I believe that immediate impacts to stream conditions caused by hemlock decline and mortality will be similar to the documented impacts to stream conditions from other types of disturbances to riparian forest that are discussed in the literature survey in this thesis. The severity of these initial impacts will depend on the size of and composition of riparian hemlock stands and the rate and timing of decline and mortality. If hemlock mortality occurs in a large pure species riparian hemlock stand, I believe the defoliation of canopy cover and the cessation of nutrients and water being taken up by formerly healthy hemlocks will cause somewhat severe localized impacts to water quality. I believe short-term impacts to hemlock decline and mortality will include increased solar radiation reaching the forest floor, increased nutrient export to stream water, increased sediment delivery, and alterations to hydrology and stream temperatures.

Many riparian hemlock stands in GSMNP are composed of mixed hardwood species with an understory of Rhododendron. The presence of remaining hardwood species and Rhododendron species in formerly hemlock-dominated riparian stands will help to minimize the effects of increased solar radiation from the loss of hemlock canopy. Living hardwood and Rhododendron species can also continue to take up nutrients and water from the soil so that nutrient and water export from soils as a result of hemlock mortality will not be as severe. On the other hand, Yeakley and others (2003) investigated the effects of removing riparian Rhododendron compared to the natural blowdown of canopy trees on nutrient export to streams. They found that removing Rhododendron resulted in significantly less nutrient export to streams than the natural blowdown of canopy trees. Yeakley and others (2003) suggest, in conclusion, that rhododendron only plays a minor role in controlling nutrient export to headwater streams that canopy trees are essential in order to control nutrient loss from soils.

The timing and the manner in which hemlock decline and mortality occur will be a strong determinant of the severity of hemlock mortality's immediate impacts to stream conditions. If hemlock mortality occurs throughout a watershed at the same time, the immediate impacts to stream conditions could be much more severe. Broad-scale hemlock mortality could cause dead standing hemlocks to be more susceptible to windthrows. If a storm felled and uprooted numerous dead standing hemlocks, disruption of the roots and soil could result in significant increases in soil and stream water nitrate concentrations (Yeakley 2003). Additionally, the felling and uprooting of dead standing hemlocks over a broad area could result in significant sediment input to streams and woody debris accumulation, which could cause significant changes to the fluvial geomorphology and aquatic habitat of affected stream channels.

Although the potential immediate impacts of hemlock mortality have been compared to impacts from riparian forest harvesting, I believe the impacts will not be as great in magnitude because hemlock loss is unlikely to occur at the same time throughout a watershed. Furthermore, it is unlikely that hemlock mortality will result in a complete lack of vegetation, like some riparian forest harvesting practices.

The severity of hemlock mortality's immediate impacts will ultimately depend on the rate at which undisturbed species and replacement species can develop and fill the empty niche left by declining and standing dead hemlocks. It has been documented that elevated stream nitrate concentrations returned to pre-disturbance levels five to ten years after forest harvesting had occurred (Bormann and Likens 1979; Townsend et al. 2004).

Robinson and others (2002) found that elevated soil and stream nitrate concentrations decreased significantly as regenerating Fraser firs began to replace standing dead mature Fraser firs in GSMNP. The elevated soil and stream nitrate concentrations had been caused by the decline and mortality of mature Fraser fir trees that were killed by the balsam woolly adelgid. Further watershed-specific investigations are needed in GSMNP in order to determine the length of time that formerly hemlock-dominated watersheds will require before riparian conditions will be able to return to pre-disturbance environmental conditions.

Eventual, long-term impacts from hemlock mortality

The results of this research indicate that the long-term, eventual response of stream conditions to the replacement of hemlock-dominated forest with hardwood-dominated forest should be minimal if these systems can recover from the immediate impacts of hemlock decline and mortality. However, these results refer to watershed-scale impacts and may not address the more localized impacts that may occur at the stream-feature scale as a result of the replacement of riparian hemlock with hardwood species. For example, some headwater streams have deep pools that are currently located beneath dense hemlock canopy and may be a refuge for biota seeking the shaded cooler water during warm summer months. Although this research suggests that long-term impacts from hemlock mortality may be insignificant on a watershed-wide scale, this conclusion does not take into account more localized impacts that may occur on a smaller stream-feature scale, such as in refuge pools.

The long-term, eventual response of stream conditions to the replacement of hemlock-dominated forest with non-hemlock-dominated forest will be determined by the ability of riparian forest to recover from the immediate impacts from hemlock mortality and return to a functional intact system. The research presented in this thesis makes an assumption, based on the current literature, that riparian hemlock forest in GSMNP will be replaced by riparian hardwood forest. However, the observations of hardwood replacing hemlock in the current literature are based on research conducted in the mid-Atlantic and northeastern United States. Environmental conditions in the southern Appalachians and GSMNP differ from conditions in the mid-Atlantic and northeast. Therefore, there are some factors in GSMNP which may restrict the recovery from short-term impacts and limit the ability of the formerly hemlock-dominated riparian forest from developing into a functional intact hardwood-dominated riparian ecosystem.

For example, although none of the 14 watersheds chosen for this study exhibited signs of nitrogen saturation, other watersheds in GSMNP are experiencing nitrogen saturation (Flum and Nodvin 1995), which may prevent a fully functional intact hardwood-dominated riparian forest from replacing the formerly hemlock-dominated riparian forest. In this case, net loss of nitrate from the soil to stream water may continue and significant long term impacts to stream and soil nitrate concentrations from hemlock mortality may occur.

Additionally, forest species compositions in the southern Appalachians differ from species compositions of the mid-Atlantic and northeastern United States. For example, Rhododendron, which is a ubiquitous riparian species in the southern Appalachians, is present but not as common in the mid-Atlantic, and is not found at all in the Northeast. Rhododendron has two significant attributes that could prevent an intact hardwood-dominated riparian forest from replacing the formerly hemlock-dominated riparian forest in the southern Appalachians: 1) Rhododendron grows vigorously in disturbed areas and 2) Rhododendron limits the regeneration of hardwood tree species. Dobbs and Parker (2004) found significant expansion of the distribution of Rhododendron in riparian environments in the Southern Appalachians as a result of forest disturbance and fire suppression. Researchers have suggested that many Rhododendron thickets formed in the southern Appalachians as a result of the opening of the forest canopy in the 1930s after the blight-induced decline of the American chestnut (*Castanea dentata*) (McGinty 1972; Clinton et al. 1994). Other researchers attribute the establishment of Rhododendron thickets to forest disturbances such as logging (McGee and Smith 1967; Vandermast 2002). In locations where the mortality of overstory tree species has occurred, Rhododendron has been found to develop into a dense thicket forming a continuous sub-canopy with an absence of canopy tree species (Baker and Van Lear 1998; Vandermast et al. 2002). With the opening of the forest canopy caused by hemlock decline and mortality, I hypothesize that dense thickets of Rhododendron could expand along formerly hemlock-dominated riparian corridors and prevent the recruitment and colonization of hardwood canopy tree seedlings. Research has shown that Rhododendron has the ability to cause mortality and suppress the growth of hardwood seedlings (Nilsen et al. 2000; Lei et al. 2002; Hille Ris Lambers and Clark 2003). Rhododendron has been found to reduce the availability of resources both above (light, precipitation) and below ground (water, nutrients) for canopy tree seedlings (Nilsen et al. 2000).   Other research has suggested that Rhododendron may have some allelopathic characteristics (Nilsen et al. 1999; Nilsen et al. 2001). In these southern Appalachian Rhododendron thickets, hemlock has been found to be the only riparian tree species that has been able to regenerate and attain overstory status (Vandermast et al. 2002). Once hemlock is absent from riparian forests in the southern Appalachians, I hypothesize that it may be possible that no other tree species will be able to regenerate and attain overstory status in dense Rhododendron thickets without the aid of gap-phase disturbance. I also hypothesize that with the loss of riparian hemlock, dense Rhododendron thickets without overstory tree species may become more prevalent in the riparian forest of the southern Appalachians. If this occurs, the long-term impacts to riparian environmental conditions in the southern Appalachians caused by hemlock mortality will be much more severe than if intact hardwood-dominated riparian forest was able to replace formerly hemlock-dominated riparian forest. Additional research on the influence of Rhododendron thickets on riparian environmental conditions would contribute to a better understanding of the potential future of currently hemlock-dominated riparian forest in GSMNP.

5.6 Baseline Data

The riparian environmental parameters measured in this study will serve as baseline data, characterizing the conditions of low to middle elevation headwater streams before the onset of HWA induced hemlock mortality in GSMNP. Baseline data can be used in the future to track the magnitude of change in riparian environmental conditions that occur with hemlock decline and hemlock mortality.

5.7 Forest Management Implications

As a result of this research, I believe that riparian hemlock stands should be considered as priority sites for the implementation of HWA control strategies in order to help minimize potential short-term impacts to riparian environmental conditions. I would suggest that management of hemlock mortality focus on minimizing the immediate effects of hemlock decline and mortality. Efforts should be focused on large pure-species riparian hemlock stands that will have the greatest immediate impact to stream conditions. There is evidence that the uprooting of trees result in more significant losses of nitrate to stream water than if trees remain standing (Yeakley et al. 2003). Therefore, management agencies should attempt to prevent declining and dead standing hemlocks from being uprooted in locations where they are in close proximity to streams.

Management agencies should also investigate opportunities to encourage the establishment of hardwood canopy species in locations where hemlock mortality has occurred. Vandermast and Van Lear (2002) suggest introducing periodic fire into riparian forests in the Southern Appalachians in order to control Rhododendron expansion and to help encourage hardwood canopy tree regeneration. While fire introduction may suppress the establishment of Rhododendron thickets, it may also lead to further increases in nutrient export to stream water and therefore should be used with caution. The mechanical removal of Rhododendron has proved somewhat unsuccessful and should also be used with caution. Clinton and Vose (2000) document the development of extremely high densities of Rhododendron after only a few years following mechanical removal.

Management agencies should strive to establish an intact riparian vegetative cover in order to minimize the impacts to riparian environmental conditions from hemlock decline and mortality. An intact riparian vegetative cover will intercept solar radiation, reducing energy input to stream water surfaces, and will take up nutrients, reducing the levels of nitrate that will enter stream water.

5.8 Conclusions

The current peer-reviewed literature documents that short-term impacts to stream conditions from hemlock mortality and other forest disturbances can be severe. However, research also indicates that stream conditions return to pre-disturbance levels within five to ten years. In GSMNP, the return to pre-disturbance levels after hemlock mortality will depend on the type of replacement species and how quickly the replacement species can establish in disturbed sites. There is evidence that deciduous hardwood species are most likely to replace hemlock. The results of this study suggest that hemlock and hardwood stream conditions (temperature, nitrate concentrations, pH, and discharge) are similar in GSMNP. Therefore, if deciduous hardwood species are able to replace hemlock in GSMNP and formerly-hemlock ecosystems are able to recover from short-term impacts, the long-term impacts from hemlock mortality on stream conditions will be minimal. However, the presence of Rhododendron in the understory of riparian hemlock forests in GSMNP may prevent deciduous hardwood species from replacing hemlock, which could result in significant long term impacts to species composition and stream conditions in formerly hemlock-dominated sites.

This paper specifically addresses impacts to stream conditions from hemlock mortality and suggests that long-term impacts to stream conditions in GSMNP will be minimal. However, it is important to note that this paper does not address impacts from hemlock mortality to aesthetics, recreation, or wildlife, all of which could be substantially impacted by the loss of hemlock from eastern forests. Additional studies investigating impacts from hemlock mortality on specific wildlife species are needed.

The results and inferences from this research are limited by the duration of the study and the sample size. Although I found no clear, consistent pattern of hemlock or hardwood riparian forest being associated with particular stream conditions, it is possible that a pattern could emerge from a larger sample size monitored over a longer period of time. However, it is my opinion that the results presented in this thesis are a good representation of the stream conditions that occur with hardwood- and hemlock-dominated headwater streams in GSMNP and that a larger sample size would yield similar results. Furthermore, I believe that the results and discussion in this paper further our understanding of the differences in stream conditions between hemlock- and hardwood-dominated headwater steams and the potential short-term and long-term impacts that can be expected from hemlock mortality in GSMNP.

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Vita

Scott Wesley Roberts was born and raised in Birmingham, Alabama. He attended middle school and high school in Louisville, Kentucky where he graduated from Ballard High School in 1998. In 2002, he received a B.S. in Geography from Appalachian State University in Boone, North Carolina. The following year he embarked on a hike of the entire Appalachian Trail over the course of five consecutive months. In 2004, Scott began working on a graduate degree in the department of Geography at the University of Tennessee where he received a research assistantship with the Tennessee Valley Authority. He received a M.S. in Geography in May of 2006.