James Madison University
JMU Scholarly Commons

Spring 2010

Changes in light levels with loss of eastern hemlock (*Tsuga canadensis*) at a Southern Appalachian Headwater Stream: Implications for brook trout (*Salvelinus fontinalis*)

Leigh A. Siderhurst
James Madison University

Follow this and additional works at: https://commons.lib.jmu.edu/master201019

Part of the Biology Commons

Changes in Light Levels with Loss of Eastern Hemlock (*Tsuga canadensis*) at a

Southern Appalachian Headwater Stream:

Implications for Brook Trout (*Salvelinus fontinalis*)

Leigh A. Siderhurst

A thesis submitted to the Graduate Faculty of

JAMES MADISON UNIVERSITY

In

Partial Fulfillment of the Requirements

for the degree of

Master of Science

Department of Biology

May 2010

Acknowledgements

Table of Contents

List of Tables

List of Figures

Abstract

The exotic invasive insect, hemlock woolly adelgid (*Adelges tsugae* Annand), is causing mortality in eastern hemlocks (*Tsuga canadensis* [L.] Carr.) throughout the eastern U. S. Hemlocks are being replaced by hardwood species that cannot fill the structural and functional role of hemlock in forest ecosystems. Because hemlocks produce dense shade, their loss may increase understory light levels. In the southern Appalachians, increases in understory light could cause changes in stream ecosystems because riparian hemlocks may help maintain cool stream temperatures for cold water species such as brook trout (*Salvelinus fontinalis*). I studied changes in light levels with eastern hemlock decline at a southern Appalachian brook trout stream using hemispherical photography and multi-temporal satellite images (ASTER). My results indicate that stream light levels have increased significantly with adelgid infestation. Leaf-on light levels are currently significantly higher ($P < 0.02$) in plots containing high basal areas of hemlock (mean global site factor (GSF)(SE) = 0.267(0.01)) compared with plots containing no hemlock (mean GSF(SE) = 0.261(0.01)), suggesting that increases in light have occurred with hemlock decline. The Normalized Difference Vegetation Index (NDVI) decreased with hemlock decline from 2001 to 2008. In 2001, NDVI showed no relationship ($R^2 = 0.003$; $F = 0.14$; $P = 0.71$) with hemlock basal area, but by 2008, there was a significant negative relationship ($R^2 = 0.352$; $F = 19.55$; $P < 0.001$) between NDVI and hemlock basal area. I also conducted a gap experiment that showed that light levels may increase by up to 64.7% more (mean increase in GSF = 27.5%) as hemlocks fall, creating gaps in the canopy. However, by comparing light levels between plots containing hemlock and those containing only hardwood species, I found that if hemlocks

are replaced by hardwood species, light levels under an all-hardwood canopy (mean GSF(SE) = 0.240(0.005)) are unlikely to be higher than they are under the current forest (mean GSF(SE) = 0.254(0.007)). These results suggest that loss of hemlock along southern Appalachian streams could have short-term impacts on light levels and stream temperatures, potentially threatening brook trout populations, but that long-term changes in light levels may be unlikely.

Introduction

The hemlock woolly adelgid (*Adelges tsugae*) is an exotic insect pest of eastern hemlock (*Tsuga canadensis*) and Carolina hemlock (*Tsuga caroliniana*) that was accidentally introduced from Asia to the eastern U.S. in the 1950s. Since then it has spread to 18 states, causing the widespread decline and mortality of eastern hemlock (hereafter referred to simply as hemlock) (McClure and Cheah, 1999; Morin et al., 2009; Trotter and Shields, 2009). Once infested, hemlocks can die within 4-15 years and so far no controls for hemlock woolly adelgid that are both economical and practical for use on a large scale have been found (Orwig et al., 2002; Trotter and Shields, 2009). Loss of hemlock is predicted to have major impacts on forest ecosystems throughout the eastern U.S. because hemlock is considered to be a foundation species – one that strongly influences the structure and function of ecosystems in which it is found (Ellison et al., 2005; Rohr et al., 2009).

Hemlocks influence the forest environment through a unique combination of species traits. They are long-lived, extremely shade-tolerant evergreens that form self-regenerating climax forests. They cast deep shade year-round, creating cool damp microclimates, produce acidic litter that is slow to decompose, and are associated with low rates of nitrogen cycling (Hadley, 2000; Orwig, 2002; Lovett et al., 2006). In addition, hemlock stands often occur along streams, where they help to maintain cooler water temperatures in summer, warmer temperatures in winter, and more stable flow regimes year round compared with hardwood streams (Snyder et al., 2002; Hadley et al., 2008). Because of their unique combination of species traits, hemlock forests also provide habitat for unique communities of insect, bird, and fish species (Snyder et al.,

2002; Ross et al., 2003; Ross et al., 2004; Dilling et al., 2007; Allen et al., 2009; Rohr et al., 2009).

Following adelgid-induced mortality, hemlock will most likely be replaced by various hardwood species (Sullivan and Ellison, 2006; Spaulding and Rieske, 2010). Studies in New England show that hemlocks are being replaced by less shade-tolerant species such as black birch (*Betula lenta*), oaks (*Quercus* spp.), and maples (*Acer* spp.) (Orwig and Foster, 1998; Orwig et al., 2002; Stadler et al., 2005). These species gain an advantage over hemlock when light levels increase with hemlock decline (Catovsky and Bazzaz, 2000). Hardwood species, however, cannot fill the structural and functional role of hemlocks in forest ecosystems, so changes in microclimate, nutrient cycling, hydrology, and species composition are likely to occur (Jenkins et al., 1999; Ellison et al., 2005; Stadler et al., 2005; Ford and Vose, 2007; Nuckolls et al., 2009).

Because hemlock is often found along streams, loss of this species and its replacement by hardwoods could cause particularly pronounced changes in stream ecosystems. Studies have shown that hemlock streams differ from hardwood streams in important ways. For example, Rowell and Sobczak (2008) found that streams with hemlock riparian zones received less solar radiation and had less algal biomass than streams with hardwood riparian zones; Snyder et al. (2002) and Willacker et al. (2009) found that different aquatic invertebrate communities were associated with hemlock streams compared with nearby hardwood streams; and at the Delaware Water Gap National Recreation Area in Pennsylvania and New Jersey, brook trout (*Salvelinus fontinalis*) have been shown to occur more frequently in hemlock streams than in hardwood streams (Ross et al., 2003). Therefore, with loss of hemlock, major shifts in

stream communities could occur, accompanied by regional losses in diversity as hemlock streams become more like nearby hardwood streams.

One of the most important changes governing the influence that loss of hemlock will have on stream communities is the potential for increases in light levels. As hemlocks are defoliated by hemlock woolly adelgid and die, they create gaps in the riparian canopy that could increase the amount of solar radiation reaching the stream. In addition, if hemlocks are eventually replaced by hardwood species in the riparian canopy, light levels in the understory could remain higher because hemlock canopies, and those of evergreen species in general, allow less light to penetrate into the understory than hardwood canopies (Gower and Norman, 1991; Canham et al., 1994; Catovsky et al., 2002). Increases in light levels could in turn cause increases in stream water temperatures, because solar radiation tends to be the primary controlling factor for stream temperature in small streams (Danehy et al., 2005; Caissie, 2006), and riparian forests have a strong influence on water temperature (Moore et al., 2005). Forested streams tend to have lower water temperatures than unforested reaches (Malcolm et al., 2008); for example, Johnson (2004) found that shading 200 meters of stream, such that photosynthetically active radiation (PAR) was reduced by 99.9%, caused maximum water temperatures to drop by 1°C. Other studies have shown that decreasing shade to streams through forest harvesting causes increases in stream water temperatures (Moore et al., 2005; Kreutzweiser et al., 2009; Pollock et al., 2009). For example, Gravelle and Link (2007) found that removal of 50% of the canopy increased temperatures by up to 3.6°C directly downstream of harvested sites. These effects can also be fairly long-lasting as

stream temperatures have been shown to take anywhere from 5-15 years to recover after harvesting (Moore et al., 2005).

The potential for increases in stream water temperatures caused by increased solar input following loss of hemlock is of particular concern in the southern Appalachian region where riparian hemlock forests are thought to play an important role in keeping stream temperatures cool enough for the survival of brook trout (Ross et al., 2003; Johnson, 2004; Ward et al., 2004; Whitledge et al., 2006). Brook trout, an important ecological indicator species whose populations have already been greatly reduced, is in the southernmost part of its range in this area (Hudy et al., 2008). The optimal stream temperature for brook trout is about 18.9°C and the maximum temperature is about 23.9°C (Eaton et al., 1995). In the south, where summer maximum water temperatures are already near these thermal limits, any increases in water temperature due to increases in light level could adversely affect brook trout populations (Eaton et al., 1995). In addition, the effects of climate change might exacerbate adelgid-induced changes, posing a significant threat to already reduced populations (Caissie, 2006; Webb et al., 2008). Natural resource managers in the southeastern U.S. need to be able to predict changes in light levels and stream temperatures at adelgid-impacted sites in order to evaluate the potential effects on stream ecosystems and make appropriate management decisions.

Predicting the degree to which loss of hemlock will change stream conditions, however, is challenging. This is because the degree to which solar input to streams increases, and in turn the level of impact on stream temperatures, depends heavily on local site-related factors such as latitude, topography, and riparian vegetation. For example, studies have shown that light levels increase with decreasing latitude, from

north to south. Canham et al. (1990) found that light levels differed significantly between sites based on latitude in old-growth Douglas-fir-hemlock forests. Those at more southern latitudes received more light. In addition, Canham (1988) found that differences in gap light levels were greater between northern and southern latitudes than between different topographic positions (i.e. north-facing vs. south-facing slopes) at the same latitude. However, the amount of light reaching a stream is also dependent upon the topography (slope and aspect) of its banks in that, (in the northern hemisphere) south-facing banks receive more radiation per unit area than north-facing banks (Barnes et al., 1998) and banks with steep slopes can significantly decrease solar input to streams (Moore et al., 2005). In addition, riparian vegetation shades streams to different degrees depending not only upon the species involved (e.g., hemlock vs. hardwood), but also on the height of the vegetation (Kelley and Krueger, 2005), its density, and its volume (Drever and Lertzman, 2003). The amount of light penetrating the canopy decreases with increasing height, density and volume of riparian vegetation, and studies have shown that the reverse is true as well: as tree basal area decreases, light levels in the understory increase (Drever and Lertzman, 2003). One study found that decreasing basal area by more than 60% caused significant increases in transmitted light; however, lesser decreases in basal area did not result in significant changes in light level (Heithecker and Halpern, 2006). Therefore, changes in solar input to streams in adelgid-impacted hemlock forests will be strongly influenced not only by latitude but also by riparian topography and the percentage of hemlock in the riparian canopy, all of which are highly site-specific factors.

Nearly all research to date, however, on changes in light levels due to hemlock decline and mortality has been carried out in the northeastern U.S. These studies have

noted dramatic increases in light levels in declining hemlock stands (Orwig and Foster, 1998; Jenkins et al., 1999; Eschtruth et al., 2006; Cleavitt et al., 2008). For example, Eschtruth et al. (2006) found that transmitted light had increased significantly (p<0.001) from an average of 5% in 1994 to an average of 11.7% in 2003 in declining hemlock stands in northern Pennsylvania. In addition, Rowell and Sobczak (2008) found that hemlock streams in Massachusetts and Connecticut received significantly less light (34% less) than hardwood streams. However, few studies have examined the impacts of hemlock mortality on understory light or stream temperatures in southeastern forests.

Southern riparian hemlock forests tend to differ from those in the northeast, not only in latitude, but also in that they tend to occur as mixed hemlock-hardwood stands, often with a rhododendron understory, while in northeastern forests, hemlocks tend to form very dense, almost pure stands with little or no understory (Ellison et al., 2005; Kincaid, 2007; Roberts et al., 2009). Because of these differences in hemlock forest composition and structure, changes in light levels observed in northeastern forests may not be applicable to hemlock forests of the southern Appalachians (Kincaid and Parker, 2008). If hemlock plays a primary role in influencing solar input to southern streams, its loss could cause even more dramatic changes in light levels than those seen in the northeast due to the effects of latitude on solar radiation. Alternatively, other factors such as slope and aspect of stream banks, degree of hardwood cover, and/or understory vegetation may be more important than hemlocks in regulating stream light levels in this region. More studies are needed to improve our understanding of how loss of hemlock will impact southeastern forests and streams.

I studied changes in light level with hemlock decline and mortality at a southern Appalachian mixed hemlock-hardwood brook trout stream using hemispherical photography and remote sensing techniques. Hemispherical photos have been used successfully in many studies to quantify light levels in the understory of forests and forested streams (Bellingham et al., 1996; Ringold, 2003; Cleavitt et al., 2008). They have also been shown to produce more consistent results and require smaller samples sizes when compared with other instruments commonly used to measure shade, such as the densiometer and the clinometer (Kelley and Krueger, 2005).

Remote sensing techniques have also become important tools for monitoring forest health, especially where on-the-ground data are not available (Maingi and Luhn, 2005; Quinonez-Pinon et al., 2007; Gross et al., 2009) and they have been used in previous studies to monitor damage to hemlock forests caused by hemlock woolly adelgid (Royle and Lathrop, 1997; Bonneau et al., 1999; Royle and Lathrop, 2002). I used the Normalized Difference Vegetation Index (NDVI) derived from satellite images (ASTER: NASA Land Processes Distributed Active Archive Center) to examine hemlock decline. The NDVI has been widely used in studies monitoring changes in vegetation and forest health because it can detect declines in canopy cover (Pettorelli et al., 2005) and, more specifically, it has been useful in monitoring insect defoliation in forests over time (de Beurs and Townsend, 2008; Dennison et al., 2009; Eklundh et al., 2009). For example, Dennison et al. (2009) successfully used ASTER-derived NDVI data to detect tamarisk stand defoliation over time along a narrow strip of riparian area.

The objectives of this study were (1) to determine whether light levels have increased with hemlock decline and mortality at a southern Appalachian brook trout

stream in Virginia, (2) to predict how solar input to this stream is likely to change in the future as more hemlocks die, create gaps in the canopy, and are replaced by hardwood species, and (3) to assess the potential for changes in light level to cause increases in water temperature and therefore impact the brook trout population at the study site. I hypothesized that: (1) despite adelgid infestation, current stream light levels would be lower in areas containing greater basal areas of hemlock than those containing less hemlock or only hardwood species, but that (2) adelgid infestation has caused stream light levels to increase in areas containing hemlock, (3) light levels are likely to continue to increase in the near future as more hemlocks die and fall, (4) light levels will likely remain higher in the long-run if hemlocks are completely replaced by hardwood species, and (5) increases in light level are likely to cause increases in stream temperature. In testing these hypotheses, I predicted that: (1) plots with a greater proportion of the canopy composed of hemlock would have lower light levels under both leaf-on and leaf-off conditions, (2) that NDVI has declined from 2001 to 2008 because of adelgid infestation, (3) that removing up to 70% of the riparian basal area by felling dying hemlocks would increase understory light levels, (4) that light levels in a simulated future all-hardwood forest would be higher than those in the mixed hemlock-hardwood forest, and (5) that there would be a positive relationship between light levels and stream temperatures.

Materials and methods

Site description

This study was conducted at Fridley Run, which is located at Fridley Gap (38.49 °N; 78.70 °W) in George Washington National Forest, near Harrisonburg, Virginia (Figure 1a). Hemlocks occur in the riparian area in a mixed hemlock-hardwood forest along approximately 1.7 km of the stream. Fridley Run is also the headwaters for the Smith Creek watershed, which is part of the Eastern Brook Trout Joint Venture's program to restore and protect brook trout habitat (Eastern Brook Trout Joint Venture, 2010) (Figure 1b).

a)   b)
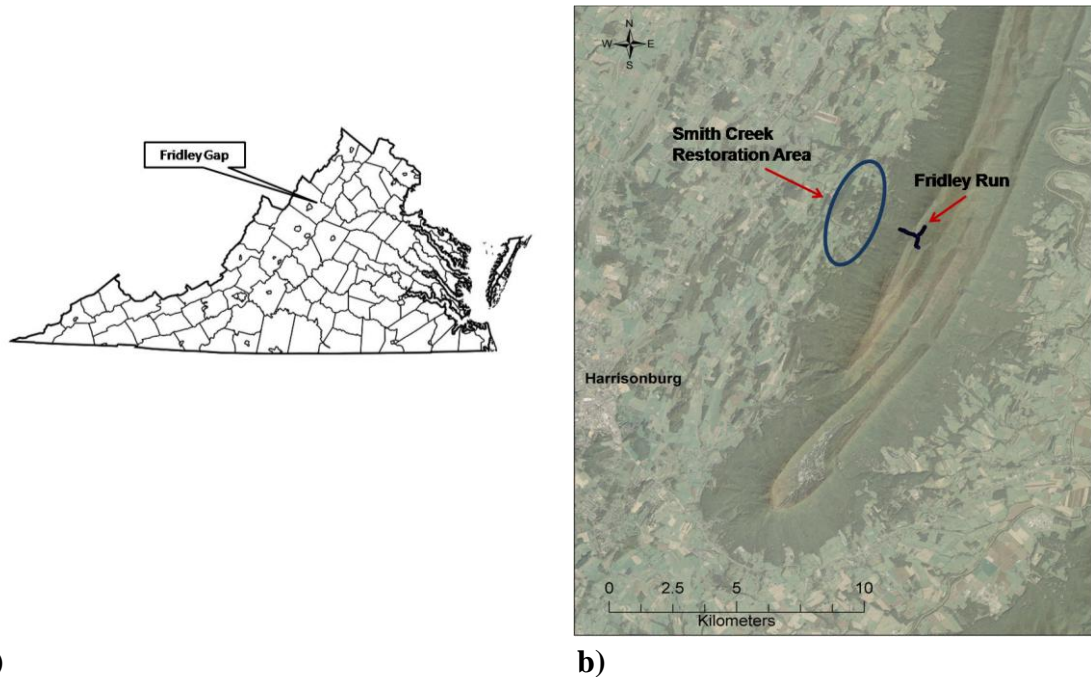
Figure 1. a) Map of Virginia showing the location of the study site at Fridley Gap in George Washington National Forest, Rockingham County; **b)** 2008 aerial photograph (USDA National Agricultural Imaging Program (NAIP)) showing the location of the headwater study site, Fridley Run, in relation to the Eastern Brook Trout Joint Venture's restoration site downstream at Smith Creek.

The stream was limed in 1993 to counteract the effects of years of acid deposition that had decimated brook trout populations (Hudy et al., 2000). Following liming, a population of brook trout was successfully restored at Fridley Run and the population has been subsequently monitored. Forest restoration efforts are also underway downstream at Smith Creek to encourage the brook trout population to extend its range and increase in numbers. However, riparian hemlock trees at Fridley Gap were found to be infested with hemlock woolly adelgid as of 2000-2001, so the loss of hemlocks along the stream poses a potential new threat to this already small population of brook trout.

Detecting changes in light levels and stream temperatures from early adelgid infestation to the present

Monitoring hemlock defoliation and changes in stream temperature

Hemlock defoliation was monitored at Fridley Gap in 2002, 2004, 2006, and 2007 by measuring the crown densities of all hemlock trees in the riparian area that were over 10 cm in diameter at breast height (DBH). Crown density was measured using the U.S. Department of Agriculture (USDA) Forest Service crown density classification method as described in Schomaker et al. (2007).

In 2008, I established 38 study plots (20 m in radius) at 50 m intervals along the stream, starting at a randomly selected point in the hardwood forest portion of the stream, downstream from the section containing hemlocks (Figure 2). GPS coordinates were recorded in the field for each plot center using a Trimble® GeoXM™ handheld GPS unit, and all data were post-processed using differential correction in GPS Pathfinder® software version 3.1 (Trimble Navigation Ltd.) to an accuracy level of approximately 1-5

m. I then imported GPS plot coordinates into ArcMap (Arc GIS 9.2 ESRI 2006) for use

in mapping and data analysis.

I placed HOBO® water temperature data loggers (Onset: HOBO® Pro v2) in the

center of the stream at every other plot (Figure 2). Loggers were set to record

temperatures continuously at half hour intervals. I secured each water temperature logger

to a brick attached to a cable that was tied to the nearest sturdy root extending from the

stream bank, and covered loggers with flat, heavy stones in order to insure that they

would remain on the stream bed and not be exposed to direct sunlight.
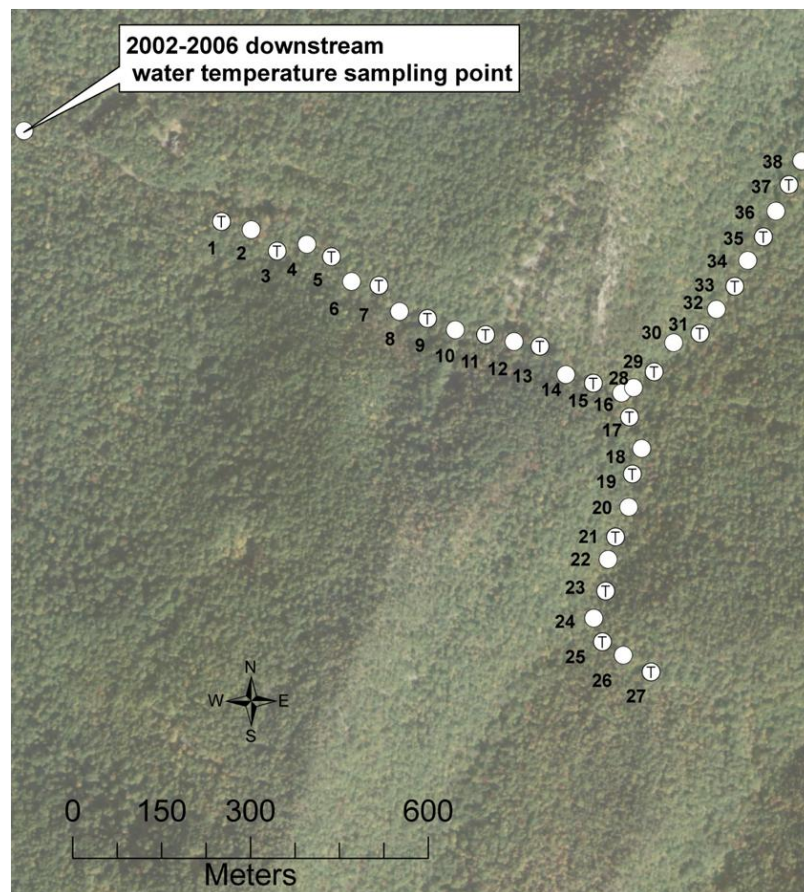
Figure 2. NAIP 2008 aerial photograph of the study site showing locations of 38
permanent study plots established in 2008. Plots containing water temperature loggers are
marked with a 'T'. The location of the earlier 2002-2006 water temperature sampling
point is also shown.

Water temperatures had also been monitored from 2002-2006 at a location about 300 m downstream of plot 1 (Figure 2), so I was able to look for changes in mean daily maximum stream temperatures (for the July to September critical period for brook trout) from 2002-2009 using this data and data from plot 1 (for 2008-2009). I used the paired samples Wilcoxon Signed-Ranks test to compare mean daily maximum temperatures among years.

Forest composition and hemlock basal area

I carried out an inventory (in 2008) of the riparian forest in 38 inventory plots located at 50 m intervals along the stream in order to characterize the composition of the riparian forest at Fridley Gap. Inventory plot centers were located in the riparian forest 15 m from the stream plot centers on either the left or right bank. Left and right bank plots were chosen randomly. GPS coordinates were recorded for each plot center. All trees over 10 cm DBH within a 20 m radius of each plot center were included in the inventory. I identified each tree to species and measured its DBH. I calculated the basal area of each species and determined the relative basal area of hemlock in each plot and at the study site as a whole.

Current light levels

To determine current stream light levels, I measured leaf-off (late February 2008) and leaf-on (early October 2008) light levels in the study plots using hemispherical photography. Light was measured in odd numbered stream plots for leaf-off data, and in all 38 stream plots for leaf-on data. Photographs were taken at 1 m above stream center under uniform cloud cover conditions in which no shadows were being cast (with a NIKON FC-E8 fish-eye converter lens fitted to a Nikon Coolpix 950 digital camera) and

subsequently analyzed using HemiView 2.1 SR1 software (Delta T Devices Ltd., 2000).

Hemispherical photographs create a permanent record of an area of the canopy

approximately 20 m in radius with the center being the point at which the camera is

located on the ground (my stream plot centers). After taking photographs in each plot, I

used HemiView software to calculate the global site factor (GSF) for each plot. The GSF

is the total amount of diffuse and direct light penetrating the canopy relative to the

amount that would be received given no canopy interception (values range from zero to

one) (Anderson, 1964). Topography, altitude and geographic location are taken into

account by the software in the calculation of the GSF.

To assess the relationship between current light levels and hemlock at the site, I

examined the relationship between GSF and the percentage of the basal area composed of

hemlock in the study plots using linear regression analyses. I also looked for differences

in mean light level among plots grouped by the percentage of the basal area composed of

hemlock (0% hemlock, <20% hemlock, and >20% hemlock). I used the non-parametric

Mann-Whitney *U* test to compare means between basal area groups.

Changes in NDVI

No data on stream light levels had been collected at the study site prior to adelgid

infestation. Therefore, in order to assess changes in light levels that have occurred since

hemlock woolly adelgid was first noticed at the site (around 2001), I conducted a remote

sensing analysis using satellite images acquired by the Advanced Spaceborne Thermal

Emission and Reflection Radiometer (ASTER). ASTER is a multispectral sensor that

records data from different spectral ranges using three sensors. For this study, I used data

from bands 2 and 3, red and near infrared (NIR) respectively, acquired by ASTER's

VNIR sensor, which has a 15 m resolution. ASTER images of the study site for August 9, 2001 and August 12, 2008 were ordered from the NASA Land Processes Distributed Active Archive Center (LPDAAC) as expedited L1A reconstructed unprocessed instrument data. I used images acquired during the summer (leaf-on period) in our analysis because July through September is the critical period for brook trout, when water temperatures may exceed their thermal limits. Both images were acquired on clear days with no cloud cover over the study site.

From the ASTER images, I calculated the Normalized Difference Vegetation Index (NDVI) for each of our study plots. NDVI was calculated as follows from the digital number (DN) values in the red and NIR extracted from the ASTER images using plot GPS coordinates:

$$NDVI = (DN_{NIR} - DN_{red}) / (DN_{NIR} + DN_{red})$$

The NDVI is a measure of the density of green leaves and tends to show a direct linear correlation with the amount of photosynthetically active radiation (PAR) absorbed by the canopy (Nagler et al., 2004; Glenn et al., 2008). Higher NDVI values mean that more light is being absorbed and reflected by leaves in the canopy and therefore that less light is being transmitted to the understory.

Because of differences in raw NDVI values from the 2001 and 2008 images created by atmospheric and illumination differences, I could not directly examine absolute differences in NDVI values between 2001 and 2008. Instead, I looked at the relationship between hemlock basal area and NDVI for both 2001 and 2008 using linear regression analyses. This allowed me to assess the relative impact of hemlock canopy cover on NDVI for each year and to look for any changes in the relationship between

NDVI and hemlock basal area between 2001, around the beginning of adelgid infestation, and 2008 when the current study was conducted. For example, I would expect to see a more negative relationship between NDVI and hemlock basal area in 2008 compared with 2001 if there had been a decrease in hemlock canopy cover. In addition, I used linear regression to examine the relationship between hemlock basal area and the difference in NDVI (NDVI 2008 – NDVI 2001) for each plot to see whether NDVI decreased more from 2001 to 2008 for plots that contained more hemlock compared with plots containing little or no hemlock.

Effects of topography on light level

To examine any effects that riparian topography might have on light levels at Fridley Run, I calculated the potential solar radiation for each of our 38 stream plots using ArcGIS Spatial Analyst extension's area solar radiation tool (ArcGIS 9.2, ESRI 2006). ArcGIS's potential solar radiation model calculates the total radiation (sum of direct and diffuse radiation) in Watt Hours per square meter (WH/m^2) based on a digital elevation model (DEM). The solar radiation model takes into account the effects of topography, atmospheric effects, elevation, daily and seasonal changes in sun angle, and latitude in its calculations (Ruiz-Arias et al., 2009; Gastli and Charabi, 2010). It does not take vegetative cover into account and therefore measures only the amount of light a given location would receive in the absence of any vegetation. Such models have been shown to provide reliable estimates of daily solar radiation for DEMs with resolutions of 100 m or less (Pons and Ninyerola, 2008; Ruiz-Arias et al., 2009). For my analysis, I used a U.S. Geological Survey (USGS) 30 m resolution DEM (National Elevation Dataset: Gesch, 2007) of the study area acquired from the University of Virginia library

(Geospatial and Statistical Data, Alderman Library) and calculated the total potential

solar radiation from March to September for the study area. In ArcGIS, I created a 20 m

radius buffer around each plot center and then used ArcGIS's zonal statistics tool to

calculate the mean potential solar radiation for each study plots from the potential solar

radiation model. I then examined the relationship between potential solar radiation (i.e.,

the degree of shading due to effects of topography) and hemlock basal area at the study

site using linear regression, and I compared mean potential solar radiation between

hardwood and hemlock plots using the Mann-Whitney U test.

**Predicting future light level changes and potential for changes in stream
temperature**

Gap experiment

Because most of the live and dead hemlocks at Fridley Gap are still standing,

though heavily defoliated, I conducted a gap experiment to determine how much light

levels are likely to change in the near future as hemlocks continue to die and fall, creating

gaps in the canopy. For this experiment, I divided the study plots into two groups: those

with low hemlock basal area (<15% hemlock by basal area) and those with high hemlock

basal area (>20% hemlock by basal area). These groupings allowed for a clear distinction

between basal area groups and an approximately equal number of plots in each group

from which to sample. Four non-adjacent plots were randomly chosen from each group

and, in each selected plot, all hemlocks over 20 cm DBH within a 20 m radius of the

stream center were felled in March 2009 (Figure 3).
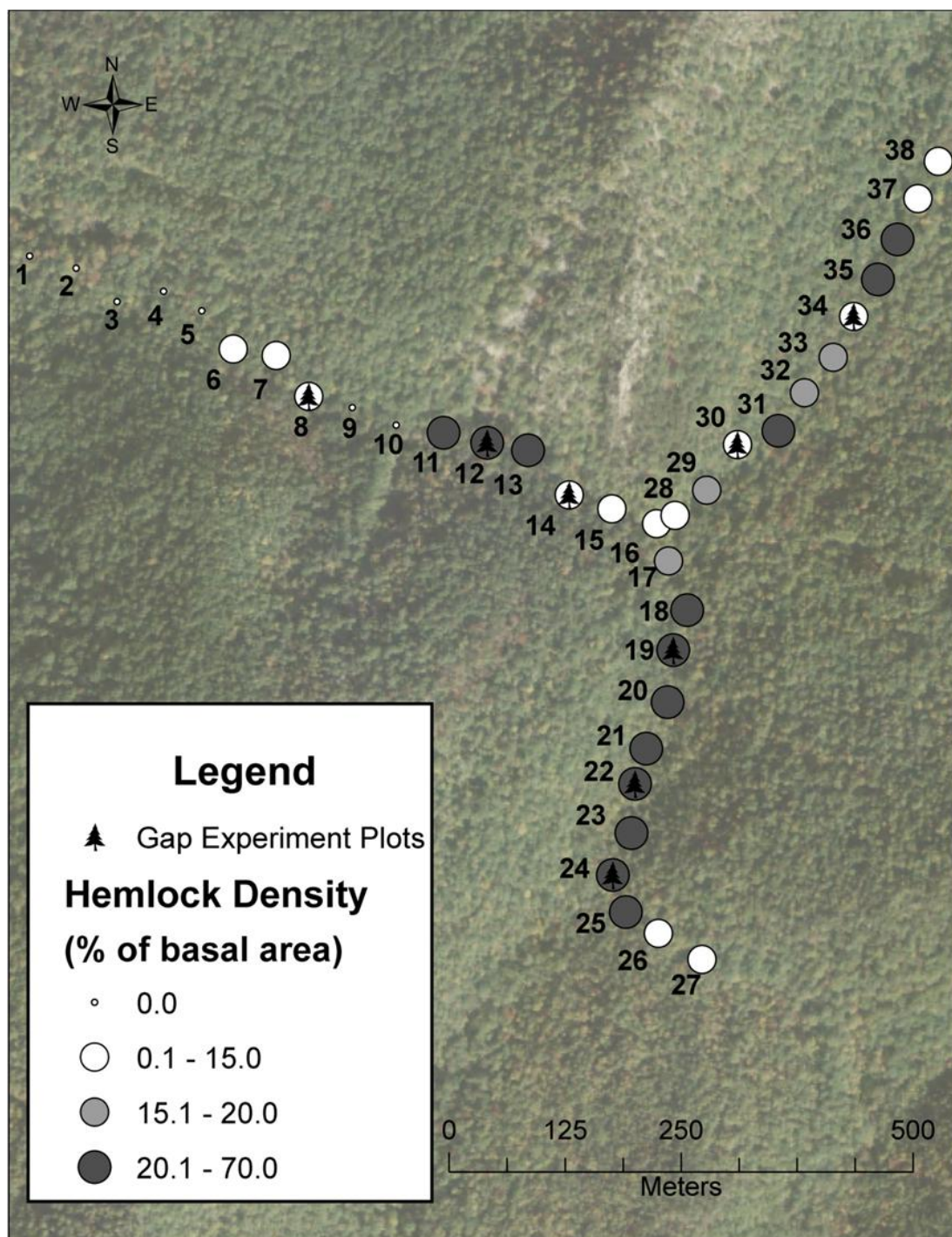
Figure 3. NAIP 2008 aerial photograph showing hemlock basal area (as % of the total basal area) in each study plot at Fridley Run. Plots randomly selected for the gap experiment are identified by an evergreen tree symbol.

Trees were cut such that they fell into the stream in order to allow the downed wood to provide additional trout habitat. This study employed a paired sampling design in that I measured light levels in the same plots both before and after selective removal. This allowed me to examine the influence of hemlock on light levels apart from other potentially confounding variables, such as topography, since no major changes other than the loss of hemlocks over 20 cm in DBH occurred in these plots between sampling dates. I measured light levels in the gap experiment plots using hemispherical photography in early October of 2008, prior to selective removal, and again in early October of 2009, post-hemlock removal. Since light level (GSF) data for gap experiment plots showed some deviation from normality, I analyzed changes in light level using the paired samples Wilcoxon Signed-Ranks test. I also looked for a difference in mean change in light level between high and low hemlock basal area experimental groups using the Mann-Whitney *U* test for independent samples.

Light levels in a simulated future all-hardwood forest

I matched hemlock plots with non-hemlock plots based on their potential solar radiation values in order to predict what stream light levels might be like in a future all-hardwood riparian forest. I then assigned each hemlock plot the 2008 leaf-on light level (GSF) of the hardwood plot most closely matching it in potential solar radiation (potential solar radiation values ranged from 869,007.31 WH/m^2 to 981,937.25 WH/m^2 and the difference in potential solar radiation for each pair of matched plots was no greater than 26,000.00 WH/m^2) (Table 1). This allowed me to account for the effects of topography on light level while simulating leaf-on light levels under a potential future all-hardwood riparian forest. I then compared the mean light level for the simulated all-

hardwood forest with the mean light level of the current forest using the paired Wilcoxon

Signed-Ranks test.

Table 1. Potential solar radiation, hemlock basal area, and light level (GSF) values listed
for each study plot in order from lowest to highest potential solar radiation value. Each
plot containing hemlock was paired with the plot containing no hemlock (shown in bold)
that had the closest potential solar radiation value and then assigned the current leaf-on
GSF for that plot in order to obtain a rough estimate of what light levels might be like
under a future all-hardwood riparian canopy.

Plot #	Potential solar radiation (WH/m^2)	Hemlock basal area (% of total)	Current leaf-on GSF (2008)	Predicted all-hardwood GSF
23	869007.31	50.00	0.267	0.212
17	881650.19	15.83	0.282	0.212
3	**893627.13**	**0.00**	**0.212**	**0.212**
22	897054.50	39.40	0.319	0.212
29	912316.56	17.09	0.280	0.212
18	912571.88	22.00	0.196	0.212
19	912699.00	43.12	0.287	0.212
31	916149.38	25.99	0.241	0.212
32	918327.06	16.88	0.249	0.212
20	919383.25	42.58	0.309	0.212
24	921596.00	34.30	0.222	0.259
35	924276.50	26.67	0.299	0.259
21	929119.69	59.49	0.271	0.259
34	930732.31	3.35	0.257	0.259
11	933717.81	69.97	0.286	0.259
33	935247.69	16.83	0.282	0.259
37	945028.31	11.13	0.260	0.259
25	945458.88	44.37	0.226	0.259
10	**946475.63**	**0.00**	**0.259**	**0.259**
12	946690.44	30.72	0.280	0.259
38	947529.81	9.61	0.241	0.259
13	947937.13	35.38	0.306	0.259
6	948606.31	8.36	0.292	0.259
26	951871.88	3.50	0.168	0.259
36	952324.88	20.79	0.228	0.259
16	953161.81	12.64	0.262	0.259
28	955045.00	10.73	0.263	0.259
30	964317.56	9.25	0.216	0.232
2	**964645.75**	**0.00**	**0.232**	**0.232**
8	964844.50	1.54	0.296	0.232
27	965179.60	0.31	0.144	0.232
14	970306.44	5.57	0.273	0.179
1	**970466.94**	**0.00**	**0.179**	**0.179**
4	**973331.38**	**0.00**	**0.242**	**0.242**
15	976813.81	1.16	0.310	0.242
7	979445.75	0.73	0.321	0.321
9	**981937.25**	**0.00**	**0.206**	**0.206**
		Mean GSF =	**0.254**	**0.240**

Relationship between hemlock basal area, light levels, and stream temperature

Finally, to assess the impact that observed and predicted changes in light levels

might have on the brook trout population at Fridley Gap, I looked at the relationship

between mean daily maximum stream temperatures, recorded between July and

September of 2008 and 2009, and hemlock basal area for all study plots using linear regression analyses. I also used regression analyses to examine the relationships among stream temperatures and GSF, potential solar radiation, and relative distance from the primary ground water inputs (located just upstream of plots 27 and 38).

All statistical analyses in this study were conducted using either SPSS 15.0 or PASW Statistics 17.0, and all determinations of significance were made at the $P < 0.05$ level.

Current status of the forest and changes in light levels and stream temperatures from early adelgid infestation to the present

Forest composition, hemlock defoliation, and changes in stream temperature

Forest inventory results showed that at Fridley Gap hemlock is co-dominant with chestnut oak (*Quercus prinus*) in a mixed hemlock-hardwood riparian forest (Table 2).

Table 2. Composition of the riparian forest in 38 study plots at Fridley Gap. The top seven species by basal area are listed. Hemlock is co-dominant with chestnut oak.

Tree Type	% of the total site basal area
Basswood (*Tilia americana*)	5.7
Birch (*Betula* spp.)	7.3
Red Maple (*Acer rubrum*)	8.0
Tulip Poplar (*Liriodendron tulipifera*)	10.3
Red Oak (*Quercus rubra*)	11.2
Hemlock (*Tsuga canadensis*)	18.0
Chestnut Oak (*Quercus prinus*)	19.1

I observed extensive patches of mountain laurel (*Kalmia latifolia*) in the understory, but unlike some other southeastern hemlock forests, Fridley Gap does not possess a dense rhododendron understory. I found that hemlocks make up 18% of the total basal area at the study site with the hemlock component of the basal area ranging from 0% to 70% in individual plots (Figure 3; Table 3).

Other species that make up a significant proportion of the riparian forest are chestnut oak (*Quercus prinus*), red oak (*Quercus rubra*), tulip poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), birch (*Betula* spp.), and basswood (*Tilia americana*) (Table 2).

Table 3. Hemlock % of the total basal area in each of 38 study plots at Fridley Gap.

Plot #	Hemlock % Basal Area	Plot #	Hemlock % Basal Area
1	0.0	20	42.6
2	0.0	21	59.5
3	0.0	22	39.4
4	0.0	23	50.0
5	0.0	24	34.3
6	8.4	25	44.4
7	0.7	26	3.5
8	1.5	27	0.3
9	0.0	28	10.7
10	0.0	29	17.1
11	70.0	30	9.3
12	30.7	31	26.0
13	35.4	32	16.9
14	5.6	33	16.8
15	1.2	34	3.4
16	12.6	35	26.7
17	15.8	36	20.8
18	22.0	37	11.1
19	43.1	38	9.6

Hemlocks at the site have become heavily defoliated since the adelgid infestation began, but most trees are still standing. Average crown density declined from 42.1% in early 2002 to 8.2% by the end of 2007 (Figure 4), but there was no increase in mean daily maximum water temperature at the downstream sampling points (plot 1 and earlier 2002-2006 sampling points), which had the warmest temperatures out of all the plots across all years (Figure 5).
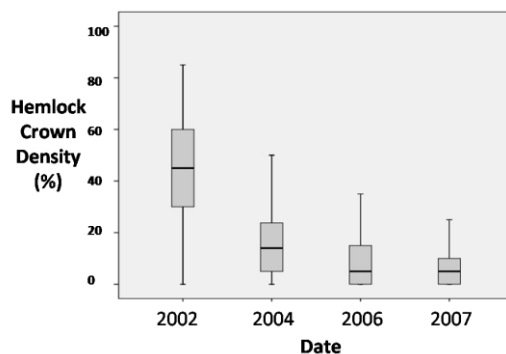
Figure 4. Box plot of crown densities (as percentage of full crown cover) of riparian hemlock trees over 10 cm in diameter at breast height at Fridley Gap measured using the USDA, Forest Service crown density classification method. Thick horizontal bars represent median crown density for each year.
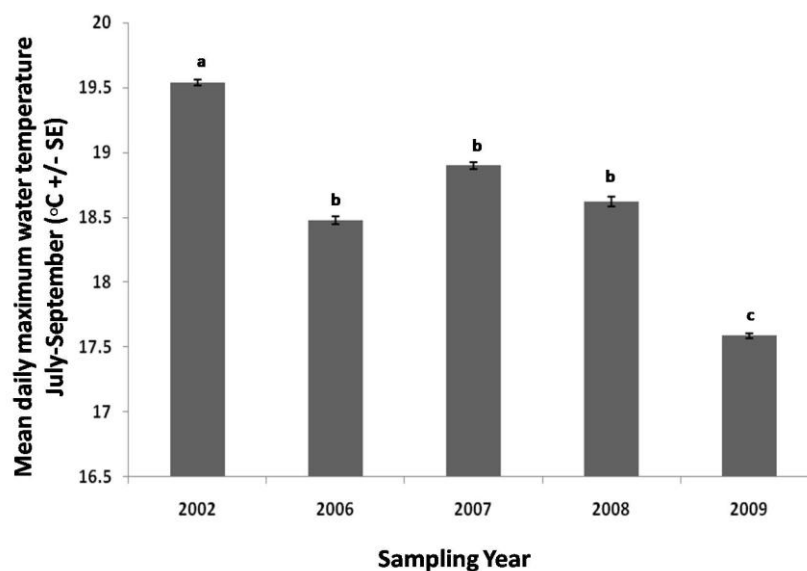
Figure 5. Mean daily maximum water temperatures (˚C +/- SE) recorded at the downstream sampling point (years 2002-2007) and plot 1 (2008 and 2009). Bars with different letters indicate significant differences in mean daily maximum water temperature among years (Wilcoxon Z test: $P < 0.05$).

Mean daily maximum temperature was significantly higher in 2002 compared with later years (Wilcoxon $Z \geq$ -2.29 and $P \leq 0.02$ for all pairwise combinations) and lower in 2009 compared with previous years (Wilcoxon $Z \geq$ - 3.61 and $P \leq 0.001$ for all pairwise comparisons). Mean daily maximum temperature at the downstream sampling point was 19.5 ˚C in 2002 (which is below the thermal limit of 23.9 ˚C for brook trout, but over the optimal temperature of 18.9 ˚C) and 17.6 ˚C in 2009, with intermediate temperatures in intervening years. This indicates that water temperature varies to some extent at this site from year to year due to variations in factors unrelated to canopy cover, such as air temperature or precipitation.

Current light levels

Leaf-off light levels (GSF) measured in 2008 showed a negative relationship (R^2 = 0.40; F = 10.80; P = 0.005) with hemlock basal area (Figure 6a). In addition, plots

containing no hemlock had significantly higher light levels (mean GSF(SE) = 0.600(0.032)) than plots with high basal areas of hemlock (mean GSF(SE) = 0.480(0.023)) (Mann-Whitney U, Z = -2.04; P = 0.042). Light levels in plots with low basal areas of hemlock had intermediate light levels (mean GSF(SE) = 0.527(0.022)) which were not significantly different from plots with no hemlock (Mann-Whitney U, Z = -1.71; P = 0.088) or plots with high basal areas of hemlock (Mann-Whitney U, Z = -1.23; P = 0.220). These results are not unexpected considering that hemlocks are known to produce more shade than hardwood species, especially in winter when hardwood species have lost their leaves. In contrast however, leaf-on light levels showed a positive relationship (R^2 = 0.13; F = 5.355; P = 0.026) with hemlock basal area (Figure 6b), though less strongly so due to substantial variation in light level among plots containing little or no hemlock. This indicates that during the summer plots containing more hemlock tend to receive more light than those containing less hemlock or only hardwoods. These results were supported by comparison of mean light levels among hemlock basal area groups, which showed that plots containing no hemlock had significantly lower light levels (mean GSF(SE) = 0.216(0.011)) than either plots with low hemlock basal areas (mean GSF(SE) = 0.254(0.014)) (Mann-Whitney U, Z = -2.45; P = 0.013) or plots with high hemlock basal areas (mean GSF(SE) = 0.267(0.010)) (Mann-Whitney U, Z = -2.46; P = 0.012). Light levels in plots with low basal areas of hemlock did not show significantly different light levels from plots with high basal areas of hemlock (Mann-Whitney U, Z = -0.44; P = 0.681).
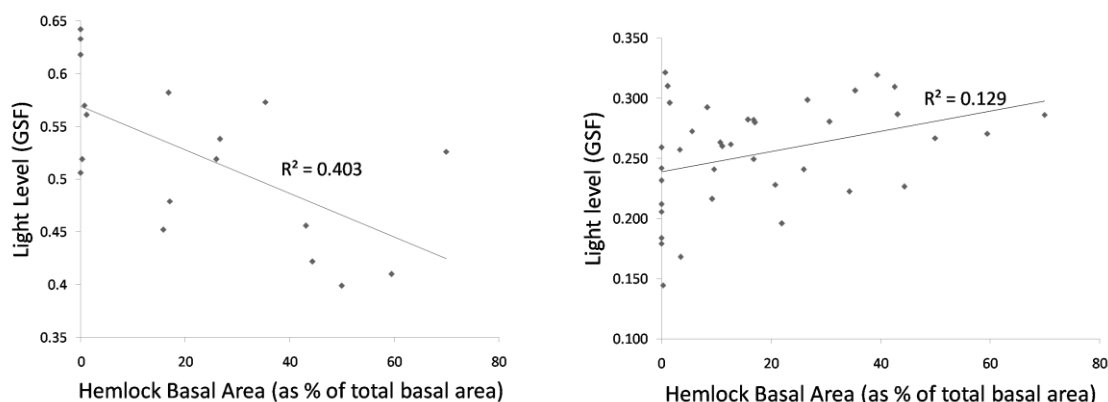
a)   **b)**

Figure 6. a) The relationship between hemlock basal area in each plot and light level (measured as the global site factor (GSF)) under leaf-off (winter) conditions in 2008. Linear regression showed a significant negative relationship between hemlock basal area and light level ($R^2 = 0.403$; $F = 10.80$; $P = 0.005$); **b)** The relationship between hemlock basal area and light level (measured as the global site factor (GSF)) under leaf-on (summer) conditions in 2008. Linear regression showed a significant positive relationship between hemlock basal area and light level ($R^2 = 0.129$; $F = 5.355$; $P = 0.026$).

These results suggest that heavy defoliation of hemlocks by woolly adelgid has probably caused increases in understory light levels, since under healthy hemlock conditions we would expect plots with more hemlock to have lower light levels in both winter and summer. However, the winter light level results indicate that, though heavily defoliated, standing hemlocks are still contributing substantially to shading the stream at Fridley Gap. There was no significant relationship ($R^2 < 0.05$; $P > 0.05$) between total plot basal area and light levels under either leaf-off or leaf-on conditions, so the observed patterns could not have been due merely to differences in total basal area among plots.

Changes in NDVI

On-the-ground light level data were supported by the results of my remote sensing analysis. In 2001 (prior to hemlock decline), there was no significant relationship ($R^2 =$

0.003; $F = 0.14$; $P = 0.71$) between the Normalized Difference Vegetation Index (NDVI)

and hemlock basal area in the study plots (Figure 7a); however, by 2008 (post hemlock

decline), there was a significant negative relationship ($R^2 = 0.352$; $F = 19.55$; $P < 0.001$)

(Figure 7b).
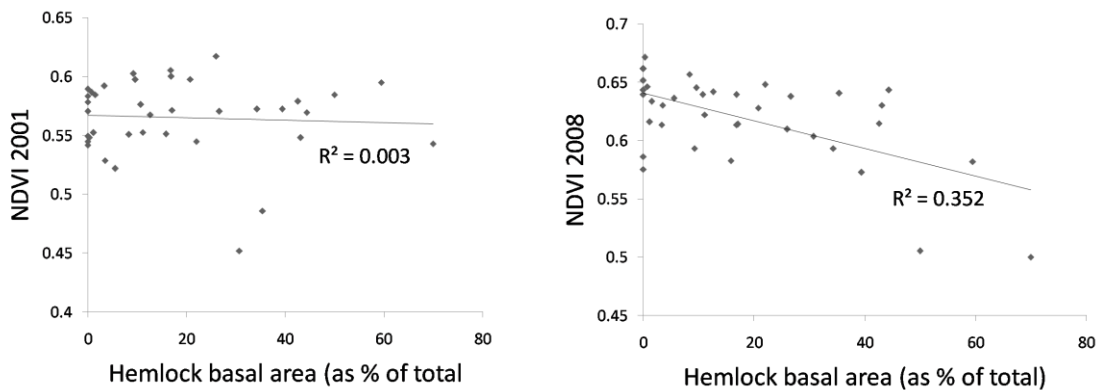
a)   b)

Figure 7. a) The relationship between hemlock basal area (as % of the total basal area in
each plot) and NDVI in 2001 (prior to hemlock decline) ($R^2 = 0.003$; $F = 0.14$; $P = 0.71$);
b) The relationship between hemlock basal area (as % of the total basal area in each plot)
and NDVI in 2008 (post-hemlock decline) ($R^2 = 0.352$; $F = 19.55$; $P < 0.001$).

Similarly, there was a significant negative relationship ($R^2 = 0.15$; $F = 6.35$; $P = 0.016$)

between the difference in NDVI between 2008 and 2001 and hemlock basal area in the

study plots (Figure 8). This indicates that NDVI decreased more from 2001 to 2008 in

plots containing more hemlock and therefore that canopy cover decreased more in those

plots.
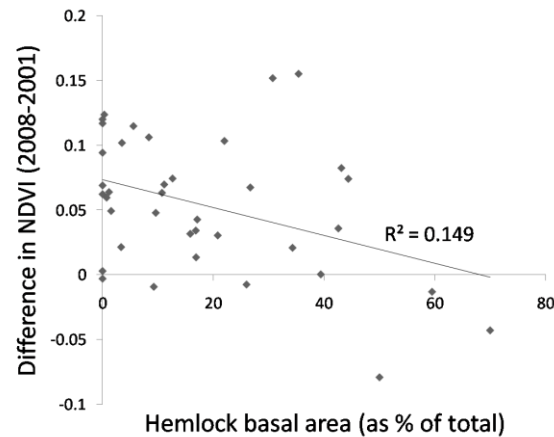
Figure 8. The relationship between hemlock basal area and the difference in NDVI between 2008 and 2001.

These decreases in canopy cover were most likely due to hemlock decline at the site because I did not observe signs of other types of damage to trees at the site. Plots containing hemlock did not contain large numbers of snags or downed trees and, according to USDA Forest Service maps, there has been no gypsy moth damage recorded in the area since 1995 (U.S. Department of Agriculture, Forest Service 2008).

Effects of topography on light level

Regression analysis showed a significant negative relationship ($R^2 = 0.30$; $F = 15.09$; $P < 0.001$) between hemlock basal area and potential solar radiation at the study site (Figure 9). Plots containing hemlock had significantly lower (Mann-Whitney U, $Z = -2.02$, $P = 0.044$) potential solar radiation values (mean = 936,400.4 WH/m^2) than plots containing no hemlock (mean = 956,786.4 WH/m^2).
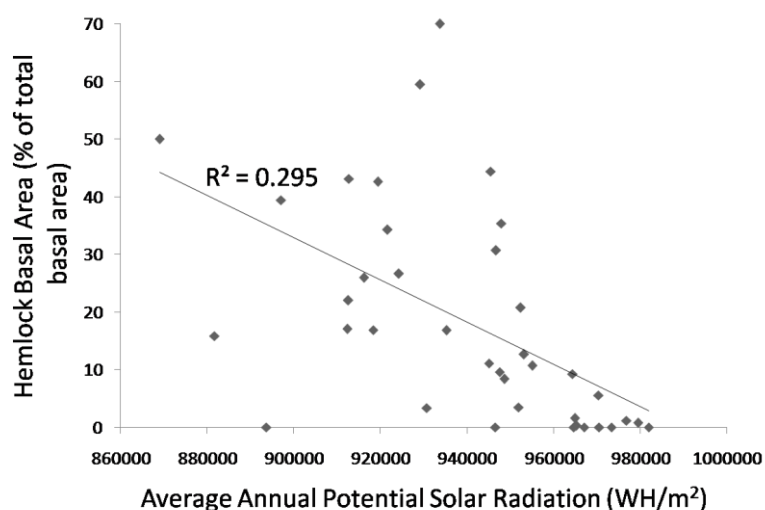
Figure 9. The relationship between average annual potential solar radiation (WH/m^2) and hemlock basal area (as % of the total basal area in each plot). Linear regression showed a significant negative relationship ($R^2 = 0.295$; $F = 15.09$; $P = 0.000$).

These data indicate that hemlock tend to dominate the riparian forest in locations that are shadier due to topographic features. This is to be expected since they are known to be a highly shade-tolerant species. However, the fact that I observed higher summer light levels (as measured by hemispherical photography) in plots with more hemlock (Figure 6b), in spite of the shadier topography of these plots, is a strong indication that hemlock decline in these plots has indeed caused light levels to increase.

Future light level changes and the potential for increases in stream temperature

Gap experiment

Gap experiment results showed significant increases (Wilcoxon $Z = -2.52$, $P = 0.012$) in light levels (GSF) with hemlock loss across all experimental plots. Increases in light level ranged from 6.1% to 64.7% in individual plots and the mean increase in GSF

was 27.5% (Figure 10). There was no significant difference (Mann-Whitney U, $Z =$ 0.000, $P = 1.000$) in mean change in light level between high hemlock basal area (mean increase in GSF = 25%) and low hemlock basal area (mean increase in GSF = 30%) treatment groups. These results show that in the near future, as dying hemlocks fall, solar input to the stream at Fridley Gap will increase significantly regardless of the proportion of hemlock in the canopy.
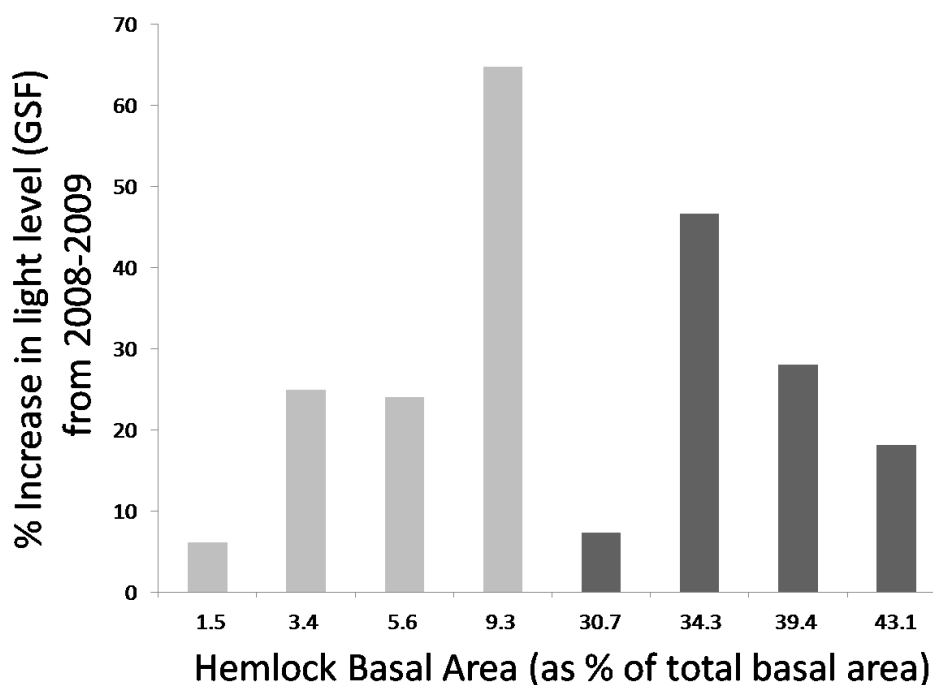
Figure 10. The percentage increase in light level (GSF) in each gap experiment plot. Low basal area treatment plots are represented by light grey bars and high basal area treatment plots by dark grey bars. Mean percent increase in GSF was 27.5%. There was a significant increase in light level for all experimental plots combined (Wilcoxon $Z = -2.52$, $P = 0.012$), but no significant difference in increase between high and low basal area treatment groups (Mann-Whitney U, $Z = 0.000$, $P = 1.000$).

Light levels in a simulated future all-hardwood forest

By matching current leaf-on GSF data from hardwood plots to hemlock plots paired based on topographic similarity, I found that the predicted future all-hardwood forest would have significantly lower (Wilcoxon $Z = -2.31$, $P = 0.021$) light levels (mean GSF(SE) = 0.240(0.005)) than the current mixed hemlock-hardwood forest (mean GSF(SE) = 0.254(0.007)) (Figure 11).
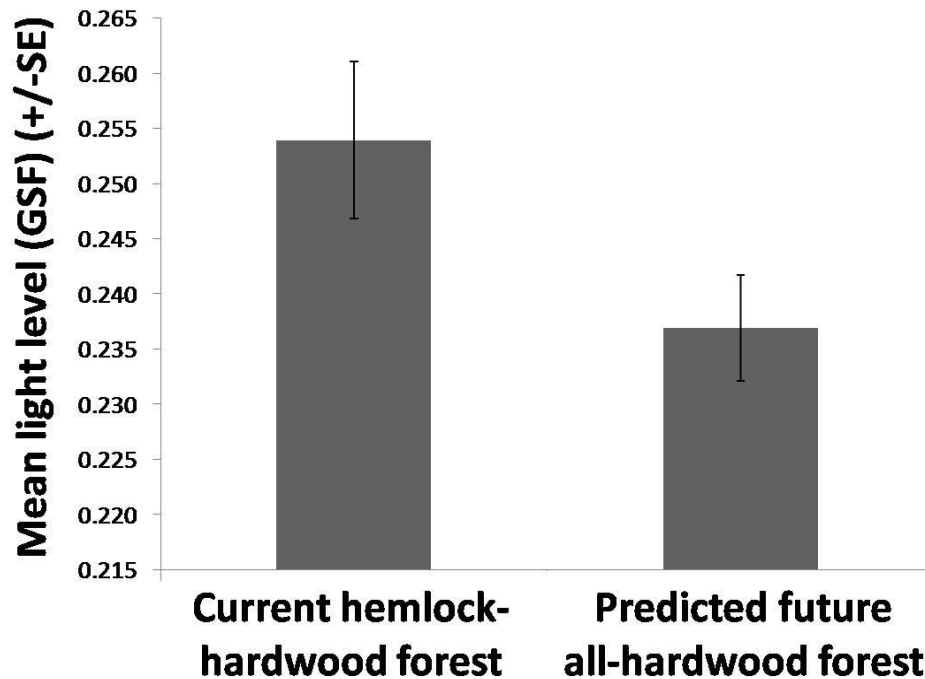
Figure 11. Bar graph showing mean summer light level (GSF +/- SE) for all plots (n=38) for the current mixed hemlock-hardwood forest and a potential future all-hardwood forest. Mean light level is significantly higher under the current riparian forest than under the predicted all-hardwood forest (Wilcoxon $Z = -2.31$, $P = 0.021$).

Although this is only a rough estimate of how light levels may change if hemlocks are completely replaced by hardwood species at this site, it is more likely to be an

overestimate of predicted light levels than an underestimate because the plots that

currently contain hemlock tend to be shadier than plots currently containing only

hardwoods based on topography alone (as measured by potential solar radiation).

Therefore, once hemlocks are replaced by hardwood species in the riparian canopy, light

levels will probably be lower than they are currently. In addition, light levels may not be

significantly different from what they were under the former healthy mixed hemlock-

hardwood canopy because NDVI results for 2001 (described above) did not show

differences in NDVI between hemlock-containing versus hardwood-only plots.

Relationship between hemlock basal area, light levels, and stream temperature

Because mean daily maximum water temperatures downstream of the hemlock

riparian area have not increased and actually appear to have decreased somewhat since

adelgid infestation began (Figure 5), it seems unlikely that water temperatures would be

higher under a potential future all-hardwood forest. However, I did find a negative

relationship between hemlock basal area and current summer mean daily maximum water

temperatures at the site (for 2008: $R^2 = 0.23$; $F = 4.8$; $P = 0.04$ (data not shown), and for

2009: $R^2 = 0.253$; $F = 5.4$; $P = 0.03$), suggesting that hemlocks might be influencing

stream temperatures to some degree (Figure 12a). However, because plots with greater

hemlock basal areas currently have higher summer light levels, this relationship is likely

due to proximity to ground water inputs rather than to the proportion of hemlock in the

canopy. This conclusion is supported by the observation that mean daily maximum

temperatures also showed a negative relationship with proximity to the stream's primary

ground water input points located just upstream of plots 38 and 27 (for 2008: $R^2 = 0.315$;

$F = 7.36$; $P = 0.015$ (data not shown), and for 2009: $R^2 = 0.451$; $F = 13.14$; $P = 0.002$)

(Figure 12b). In addition, linear regression analysis showed no significant relationship ($R^2 < 0.03$; $P > 0.05$) between stream temperature and potential solar radiation or between stream temperature and GSF.

a)   **b)**
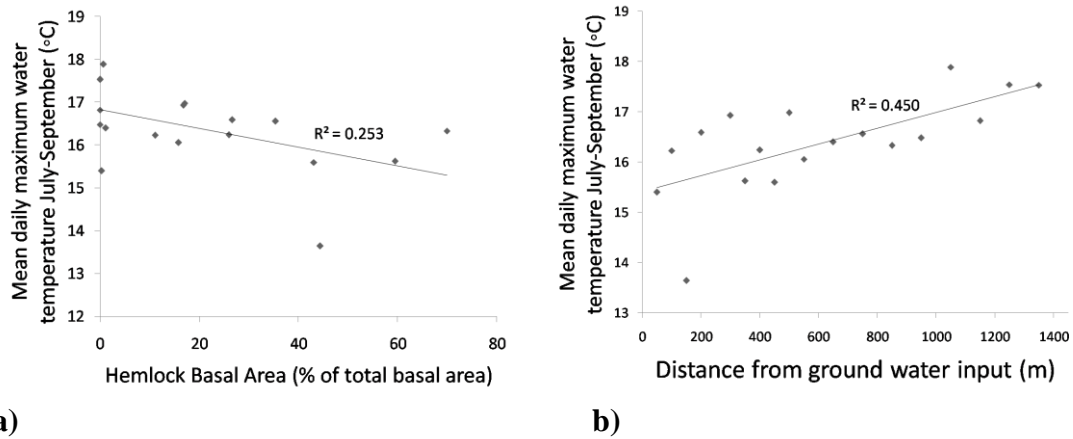
Figure 12. a) Hemlock basal area (as % of total basal area) versus mean daily maximum water temperature from July to September (in $^{\circ}$C) at Fridley Run for 2009 ($R^2 = 0.253$; $F = 5.40$; $P = 0.03$); **b)** Distance from ground water input versus mean daily maximum water temperature from July to September (in $^{\circ}$C) for 2009 ($R^2 = 0.451$; $F = 13.14$; $P = 0.002$).

Discussion

Natural resource managers need to be able to predict changes in light levels at adelgid-impacted sites in order to assess potential impacts to stream ecosystems. In the southern Appalachians in particular, managers need to know whether the loss of hemlock is likely to increase solar input to streams sufficiently to increase stream temperatures such that brook trout populations would be adversely affected (Ross et al., 2003; Ward et al., 2004; Whitledge et al., 2006; Roberts et al., 2009). Because light levels are such an important factor in regulating water temperatures in small streams (Danehy et al., 2005; Caissie, 2006), understanding the influence that loss of hemlock along southern Appalachian streams is having on understory light levels will help us to determine whether or not water temperatures are likely to increase and therefore help to inform management decisions.

The results of my study suggest that, in the short-term, stream light levels are likely to increase with hemlock decline and mortality in the southern Appalachians, even where hemlocks occur at low densities in mixed hemlock-hardwood forests. At Fridley Run, where hemlocks have become severely defoliated by woolly adelgid, I found that plots containing hemlock had significantly higher ($P < 0.02$) light levels in summer than plots with no hemlock and that there was a positive relationship ($R^2 = 0.13$; $P = 0.026$) between hemlock basal area (which ranged from 0-70% of the total basal area) and stream light level. Because this is the opposite of what we would expect under healthy hemlock conditions, in which hemlocks produce more shade than hardwood species (Canham et al., 1994; Catovsky et al., 2002), these data suggest that light levels have already increased at this site with adelgid-induced hemlock decline. These results were

supported by results obtained from ASTER satellite images, which showed that, while there was no relationship ($R^2 = 0.003$; $P = 0.71$) between the Normalized Difference Vegetation Index (NDVI) and hemlock basal area in 2001, by 2008 there was a significant negative relationship ($R^2 = 0.352$; $P < 0.001$) between NDVI and hemlock basal area, where plots with more hemlock had lower NDVI values. Similarly, there was a significant negative relationship ($R^2 = 0.15$; $P = 0.016$) between hemlock basal area and the difference in NDVI between years indicating that NDVI decreased more between 2001 and 2008 in plots that contained more hemlock. These data suggest that canopy cover has been reduced at Fridley Gap since the beginnings of woolly adelgid infestation, probably due to hemlock decline.

In addition, the results of the gap experiment showed that as dying hemlocks continue to fall, light levels will increase even more. I found that when hemlocks were removed from the canopy in both low basal area (0-15% of the total basal area) and high basal area (20-70% of the total basal area) treatments, stream light levels increased significantly ($P = 0.012$). There was no significant difference ($P = 1.000$) in increase in light level between high and low hemlock basal area treatments, but this may have been due to a combination of small sample size and the fact that there was an exceptionally high increase in light level in one of the low basal area plots (originally 9.3% hemlock by basal area) and a surprisingly small increase in light level in one of the high basal area plots (originally 30.7% hemlock by basal area). The high increase in light level in the low basal area plot might have been due to a large amount of understory vegetation that had been shading the stream but was removed by cut hemlocks that fell on top of it. The relatively small increase in light observed in the high basal area plot was probably due to

the steep, high nature of the stream banks at this location, which allows for extra shading of the stream regardless of canopy cover.

Overall, I observed a mean increase in light level (GSF) of 27.5% in gap experiment plots. With this degree of increase in light level, it is possible that stream temperatures could also increase. Previous studies have found stream temperatures to increase anywhere from 0.5 ˚C to 6 ˚C with increases in light resulting from the removal of canopy cover ranging from 15% to completely clear-cut (Bourque and Pomeroy, 2001; Johnson, 2004; Gravelle and Link, 2007). At Fridley Gap, hemlock makes up approximately 18% of the total basal area (ranging from 0-70% in any given plot) over 1.7 km of the stream. According to a water temperature model developed downstream of Fridley Run at Smith Creek, a mean increase of 25-30% in light level over 1.7 km of stream is predicted to increase mean daily maximum water temperatures by approximately 2 ˚C (Fink, 2008). This would put mean daily maximum water temperatures at Fridley Run right at about the maximum (23.9 ˚C) that brook trout can withstand (Eaton et al., 1995) and would likely place considerable stress on the population. However, it is difficult to predict exactly how much water temperatures might increase because of the influence of other factors such as ground water inputs and flow rates. At Fridley Run, the proximity of ground water inputs, due to its status as a headwater stream fed by ground water, would likely have some mitigating effect on potential increases in temperature caused by increases in solar radiation. This is supported by my data, which showed a negative relationship between water temperatures and proximity to primary ground water inputs. Further studies would be needed to more accurately predict water temperature increases with hemlock loss at Fridley Gap.

Though increases in light levels and stream temperatures may occur in the short-term with hemlock decline and mortality at southern Appalachian streams, this study suggests that in the long-run, if hemlocks are replaced by hardwood species in the riparian canopy, light levels and therefore stream temperatures are unlikely to be higher. When I matched plots currently containing no hemlock with plots containing hemlock based on similarities in potential solar radiation (i.e., topography), mean light level in the predicted future all-hardwood riparian forest at Fridley Gap was actually significantly lower ($P = 0.021$) than current light levels. This is probably due to the current state of hemlock decline. Hemlocks at this site have already become heavily defoliated by hemlock woolly adelgid and are therefore allowing more light to pass through the canopy into the understory than are the hardwood species. I found that plots currently containing only hardwood species tend to have lower light levels, in spite of having sunnier topographies, than plots currently containing hemlock, which have shadier topographies. This is a strong indication that light levels have increased in areas of the forest where hemlock is present and that the hardwood species are now producing more shade than the declining hemlocks. Therefore, replacement of hemlocks by hardwoods in these plots would actually lower light levels relative to current conditions. In addition, NDVI results from ASTER images indicate that in 2001, before hemlocks were severely damaged by adelgid, there was no relationship between canopy cover and the proportion of the basal area composed of hemlock at this site. Therefore, even if hemlocks are completely replaced by hardwood species, light levels are unlikely to be higher than they were under the original healthy mixed hemlock-hardwood canopy. These results concur with those of Roberts et al. (2009) who found that southern Appalachian mixed hemlock-hardwood

riparian forests do not necessarily have lower understory light levels than hardwood riparian forests.

Because stream temperatures at Fridley Run are not currently higher than they were at the beginnings of adelgid infestation, it is also unlikely that stream temperatures would be too warm to support brook trout under a possible future all-hardwood riparian canopy. Daily maximum summer water temperatures currently do not exceed the thermal limits for brook trout, and my results suggest that after hardwoods have replaced hemlocks in the canopy, solar input to the stream will be lower than it is currently. Therefore, water temperatures should not be any higher under a future all-hardwood canopy than they are currently. These results are similar to those of the Roberts et al. (2009) study on light levels and stream temperatures in southern Appalachian hemlock forests. Roberts et al. compared light levels and stream temperatures at hemlock-dominated streams paired topographically with hardwood-dominated streams in Great Smoky Mountains National Park (on the border of Tennessee and North Carolina). They found that hemlock riparian forests generally had lower light levels than hardwood forests, but that light levels were strongly influenced by *Rhododendron* species in the understory (i.e., hardwood forests with rhododendron had lower light levels than hemlock forests with no rhododendron). In addition, Roberts et al. found no significant differences in stream temperatures between hemlock and hardwood sites in Great Smoky Mountains National Park. Their study, like mine, suggests that long-term changes in light levels and stream temperatures resulting from loss of hemlock in southern riparian forests may not be as great as those observed in the northeastern U.S.

However, the lag time that will occur between loss of hemlock and regeneration of an all-hardwood canopy could be a problem. Given the degree of increase in light levels predicted by my gap experiment, it is possible that water temperatures in some streams could increase during this lag time. This may stress and possibly even eliminate trout populations. In addition, regeneration could be slowed by factors such as deer herbivory (Eschtruth and Battles, 2008) and lack of appropriate mycorrhizal fungi in the soil (Lewis et al., 2008), making for an even longer transition time between loss of hemlock and replacement by hardwood canopy cover. Also, in the southern Appalachians in particular, regeneration of hardwood seedlings could be delayed by *Rhododendron* species or mountain laurel (*Kalmia latifolia*) in the understory (Beier et al., 2005), especially during the initial stages of regeneration (Waterman et al., 1995). This could be a problem at Fridley Gap, where there are some extensive patches of mountain laurel in the riparian understory as well as a sizeable deer population.

Therefore, under circumstances such as those observed in my study, particularly at sites with streams whose temperatures are not strongly influenced by cooling ground water inputs, some form of active management might be necessary in order to preserve brook trout populations during the transition from hemlock to hardwood forest; for example, enrichment plantings of other evergreen species that could mimic hemlock in structure and function (Hoover et al., 2009; Weston and Harper, 2009). Restoration with other evergreen species could also help to preserve some of the characteristics of the former hemlock forest and therefore maintain regional diversity by mitigating the effects of loss of hemlock on microclimate, hydrology, and plant and animal communities (Ward et al. 2004).

This study demonstrates the need for more site-specific assessment of the impacts of hemlock woolly adelgid on light and stream temperatures in southern Appalachian riparian forests. Though substantial hardwood cover mixed with hemlock at these sites could minimize long term effects on light levels and water temperatures, management efforts may be needed in order to maintain brook trout populations in these streams during the transition from hemlock to hardwood canopy.

References

Allen, M.C., Sheehan Jr., J., Master, T.L., Mulvihill, R.S., 2009. Responses of acadian flycatchers (*Empidonax virescens*) to hemlock woolly adelgid (*Adelges tsugae*) infestation in appalachian riparian forests. Auk 3, 543-553.

Anderson, M.C., 1964. Studies of the woodland light climate. I. The photographic computation of light conditions. Journal of Ecology, 27-41.

Barnes, B.V., Zak, D.R., Denton, S.R., Spurr, S.H., 1998. Forest Ecology. John Wiley & Sons Inc., New York.

Beier, C.M., Horton, J.L., Walker, J.F., Clinton, B.D., Nilsen, E.T., 2005. Carbon limitation leads to suppression of first year oak seedlings beneath evergreen understory shrubs in Southern Appalachian hardwood forests. Plant Ecology 1, 131-142.

Bellingham, P.J., Tanner, E.V.J., Rich, P.M., Goodland, T.C.R., 1996. Changes in light below the canopy of a Jamaican montane rainforest after a hurricane. Journal of Tropical Ecology 5, 699-722.

Bonneau, L.R., Shields, K.S., Civco, D.L., 1999. Using satellite images to classify and analyze the health of hemlock forests infested by the hemlock woolly adelgid. Biological Invasions 2-3, 255-267.

Bourque, C.P., Pomeroy, J.H., 2001. Effects of forest harvesting on summer stream temperatures in New Brunswick, Canada: An inter-catchment, multiple-year comparison. Hydrology and Earth System Sciences 4, 599-613.

Caissie, D., 2006. The thermal regime of rivers: A review. Freshwater Biology 8, 1389-1406.

Canham, C.D., Denslow, J.S., Platt, W.J., Runkle, J.R., Spies, T.A., White, P.S., 1990. Light regimes beneath closed canopies and tree-fall gaps in temperate and tropical forests. Canadian Journal of Forest Research 5, 620-631.

Canham, C.D., Finzi, A.C., Pacala, S.W., Burbank, D.H., 1994. Causes and consequences of resource heterogeneity in forests: Interspecific variation in light transmission by canopy trees. Canadian Journal of Forest Research 2, 337-349.

Canham, C.D., 1988. An Index for Understory Light Levels in and Around Canopy Gaps. Ecology 5, 1634-1638.

Catovsky, S., Bazzaz, F.A., 2000. The role of resource interactions and seedling regeneration in maintaining a positive feedback in hemlock stands. Journal of Ecology 1, 100-112.

Catovsky, S., Holbrook, N.M., Bazzaz, F.A., 2002. Coupling whole-tree transpiration and canopy photosynthesis in coniferous and broad-leaved tree species. Canadian Journal of Forest Research 2, 295-309.

Cleavitt, N.L., Eschtruth, A.K., Battles, J.J., Fahey, T.J., 2008. Bryophyte response to eastern hemlock decline caused by hemlock woolly adelgid infestation. Journal of the Torrey Botanical Society 1, 12-25.

Danehy, R.J., Colson, C.G., Parrett, K.B., Duke, S.D., 2005. Patterns and sources of thermal heterogeneity in small mountain streams within a forested setting. Forest Ecology and Management 1-3, 287-302.

de Beurs, K.M., Townsend, P.A., 2008. Estimating the effect of gypsy moth defoliation using MODIS. Remote Sensing of Environment 10, 3983-3990.

Dennison, P.E., Nagler, P.L., Hultine, K.R., Glenn, E.P., Ehleringer, J.R., 2009. Remote monitoring of tamarisk defoliation and evapotranspiration following saltcedar leaf beetle attack. Remote Sensing of Environment 7, 1462-1472.

Dilling, C., Lambdin, P., Grant, J., Buck, L., 2007. Insect guild structure associated with eastern hemlock in the southern Appalachians. Environmental Entomology 6, 1408-1414.

Drever, C.R., Lertzman, K.P., 2003. Effects of a wide gradient of retained tree structure on understory light in coastal Douglas-fir forests. Canadian Journal of Forest Research 1, 137-146.

Eastern Brook Trout Joint Venture, 2010. Eastern Brook Trout Joint Venture Homepage. [Online.] http://easternbrooktrout.org, accessed January 2010.

Eaton, J.G., McCormick, J.H., Stefan, H.G., Hondzo, M., 1995. Extreme value analysis of a fish/temperature field database. Ecological Engineering 4, 289-305.

Eklundh, L., Johansson, T., Solberg, S., 2009. Mapping insect defoliation in Scots pine with MODIS time-series data. Remote Sensing of Environment 7, 1566-1573.

Ellison, A.M., Bank, M.S., Clinton, B.D., Colburn, E.A., Elliott, K., Ford, C.R., Foster, D.R., Kloeppel, B.D., Knoepp, J.D., Lovett, G.M., Mohan, J., Orwig, D.A., Rodenhouse, N.L., Sobczak, W.V., Stinson, K.A., Stone, J.K., Swan, C.M., Thompson, J., Von Holle, B., Webster, J.R., 2005. Loss of foundation species:

Consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment 9, 479-486.

Eschtruth, A.K., Battles, J.J., 2008. Deer herbivory alters forest response to canopy decline caused by an exotic insect pest. Ecological applications 2, 360-376.

Eschtruth, A.K., Cleavitt, N.L., Battles, J.J., Evans, R.A., Fahey, T.J., 2006. Vegetation dynamics in declining eastern hemlock stands: 9 Years of forest response to hemlock woolly adelgid infestation. Canadian Journal of Forest Research 6, 1435-1450.

Fink, D.B., 2008. Artificial shading and stream temperature modeling for watershed restoration and brook trout (*Salvelinus fontinalis*) management. M.S. thesis James Madison University: James Madison University, Virginia.

Ford, C.R., Vose, J.M., 2007. *Tsuga canadensis* (L.) Carr. mortality will impact hydrologic processes in southern Appalachian forest ecosystems. Ecological applications 4, 1156-1167.

Gastli, A., Charabi, Y., 2010. Solar electricity prospects in Oman using GIS-based solar radiation maps. Renewable and Sustainable Energy Reviews 2, 790-797.

Gesch, D.B., 2007. The National Elevation Dataset, in Maune, D., ed., Digital Elevation Model Technologies and Applications: The DEM Users Manual, 2nd Edition: Bethesda, Maryland, American Society for Photogrammetry and Remote Sensing, 99-118.

Glenn, E.P., Huete, A.R., Nagler, P.L., Nelson, S.G., 2008. Relationship between remotely-sensed vegetation indices, canopy attributes and plant physiological processes: What vegetation indices can and cannot tell us about the landscape. Sensors 4, 2136-2160.

Gower, S.T., Norman, J.M., 1991. Rapid estimation of leaf area index in conifer and broad-leaf plantations. Ecology 5, 1896-1900.

Gravelle, J.A., Link, T.E., 2007. Influence of timber harvesting on headwater peak stream temperatures in a northern Idaho Watershed. Forest Science 2, 189-205.

Gross, J.E., Goetz, S.J., Cihlar, J., 2009. Application of remote sensing to parks and protected area monitoring: Introduction to the special issue. Remote Sensing of Environment 7, 1343-1345.

Hadley, J.L., 2000. Understory microclimate and photosynthetic response of saplings in an old-growth eastern hemlock (*Tsuga canadensis* L.) forest. Ecoscience 1, 66-72.

Hadley, J.L., Kuzeja, P.S., Daley, M.J., Phillips, N.G., Mulcahy, T., Singh, S., 2008. Water use and carbon exchange of red oak- and eastern hemlock-dominated forests in the northeastern USA: implications for ecosystem-level effects of hemlock woolly adelgid. Tree physiology 4, 615-627.

Heithecker, T.D., Halpern, C.B., 2006. Variation in microclimate associated with dispersed-retention harvests in coniferous forests of western Washington. Forest Ecology and Management 1-3, 60-71.

Hoover, B.K., Bates, R.M., Sellmer, J.C., Hoover, G.A., 2009. Challenging Chinese hemlock (*Tsuga chinensis*) with hemlock woolly adelgid (*Adelges tsugae*) ovisacs. Arboriculture and Urban Forestry 1, 1-4.

Hudy, M., Downey, D.M., Bowman, D.W., 2000. Successful restoration of an acidified native brook trout stream through mitigation with limestone sand. North American Journal of Fisheries Management 2, 453-466.

Hudy, M., Thieling, T.M., Gillespie, N., Smith, E.P., 2008. Distribution, Status, and Land Use Characteristics of Subwatersheds Within the Native Range of Brook Trout in the Eastern United States. North American Journal of Fisheries Management 28, 1069-1085.

Jenkins, J.C., Aber, J.D., Canham, C.D., 1999. Hemlock woolly adelgid impacts on community structure and N cycling rates in eastern hemlock forests. Canadian Journal of Forest Research 5, 630-645.

Johnson, S.L., 2004. Factors influencing stream temperatures in small streams: Substrate effects and a shading experiment. Canadian Journal of Fisheries and Aquatic Sciences 6, 913-923.

Kelley, C.E., Krueger, W.C., 2005. Canopy cover and shade determinations in riparian zones. Journal of the American Water Resources Association 1, 37-46.

Kincaid, J.A., 2007. Compositional and environmental characteristics of *Tsuga canadensis* (L.) Carr. forests in the southern Appalachian Mountains, USA. Journal of the Torrey Botanical Society 4, 479-488.

Kincaid, J.A., Parker, A.J., 2008. Structural characteristics and canopy dynamics of Tsuga canadensis in forests of the southern Appalachian Mountains, USA. Plant Ecology 1-16.

Kreutzweiser, D.P., Capell, S.S., Holmes, S.B., 2009. Stream temperature responses to partial-harvest logging in riparian buffers of boreal mixedwood forest watersheds. Canadian Journal of Forest Research 3, 497-506.

Lewis, J.D., Licitra, J., Tuininga, A.R., Sirulnik, A., Turner, G.D., Johnson, J., 2008. Oak seedling growth and ectomycorrhizal colonization are less in eastern hemlock stands infested with hemlock woolly adelgid than in adjacent oak stands. Tree physiology 4, 629-636.

Lovett, G.M., Canham, C.D., Arthur, M.A., Weathers, K.C., Fitzhugh, R.D., 2006. Forest ecosystem responses to exotic pests and pathogens in eastern North America. BioScience 5, 395-405.

Maingi, J.K., Luhn, W.M., 2005. Mapping insect-induced pine mortality in the Daniel Boone National Forest, Kentucky using Landsat TM and ETM+ data. GIScience and Remote Sensing 3, 224-250.

Malcolm, I.A., Soulsby, C., Hannah, D.M., Bacon, P.J., Youngson, A.F., Tetzlaff, D., 2008. The influence of riparian woodland on stream temperatures: Implications for the performance of juvenile salmonids. Hydrological Processes 7, 968-979.

McClure, M.S., Cheah, C.A.S., 1999. Reshaping the ecology of invading populations of hemlock woolly adelgid, *Adelges tsugae* (Homoptera: Adelgidae), in eastern North America. Biological Invasions 2-3, 247-254.

Moore, R.D., Spittlehouse, D.L., Story, A., 2005. Riparian microclimate and stream temperature response to forest harvesting: A review. Journal of the American Water Resources Association 4, 813-834.

Morin, R.S., Liebhold, A.M., Gottschalk, K.W., 2009. Anisotropic spread of hemlock woolly adelgid in the eastern United States. Biological Invasions 1-10.

Nagler, P., Glenn, E., Thompson, T., Huete, A., 2004. Leaf area index and Normalized Difference Vegetation Index as predictors of canopy characteristics and light interception by riparian species on the Lower Colorado River. Agricultural and Forest Meteorology 103-112.

Nuckolls, A.E., Wurzburger, N., Ford, C.R., Hendrick, R.L., Vose, J.M., Kloeppel, B.D., 2009. Hemlock declines rapidly with hemlock woolly adelgid infestation: Impacts on the carbon cycle of southern appalachian forests. Ecosystems 2, 179-190.

Orwig, D.A., 2002. Ecosystem to regional impacts of introduced pests and pathogens: Historical context, questions and issues. Journal of Biogeography 10-11, 1471-1474.

Orwig, D.A., Foster, D.R., 1998. Forest response to the introduced hemlock woolly adelgid in southern New England, USA. Journal of the Torrey Botanical Society 1, 60-73.

Orwig, D.A., Foster, D.R., Mausel, D.L., 2002. Landscape patterns of hemlock decline in New England due to the introduced hemlock woolly adelgid. Journal of Biogeography 10-11, 1475-1487.

Pettorelli, N., Vik, J.O., Mysterud, A., Gaillard, J., Tucker, C.J., Stenseth, N.C., 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends in Ecology and Evolution 9, 503-510.

Pollock, M.M., Beechie, T.J., Liermann, M., Bigley, R.E., 2009. Stream temperature relationships to forest harvest in Western Washington. Journal of the American Water Resources Association 1, 141-156.

Pons, X., Ninyerola, M., 2008. Mapping a topographic global solar radiation model implemented in a GIS and refined with ground data. International Journal of Climatology 13, 1821-1834.

Quiñonez-Piñón, R., Mendoza-Durán, A., Valeo, C., 2007. Design of an environmental monitoring program using NDVI and cumulative effects assessment. International Journal of Remote Sensing 7, 1643-1664.

Ringold, P.L., Van Sickle, J., Rasar, K., Schacher, J.S., 2003. Use of hemispheric imagery for estimating stream solar exposure. Journal of the American Water Resources Association 6, 1373-1384.

Roberts, S.W., Tankersley Jr., R., Orvis, K.H., 2009. Assessing the Potential Impacts to Riparian Ecosystems Resulting from Hemlock Mortality in Great Smoky Mountains National Park. Environmental management 44, 335-345.

Rohr, J.R., Mahan, C.G., Kim, K.C., 2009. Response of arthropod biodiversity to foundation species declines: The case of the eastern hemlock. Forest Ecology and Management 7, 1503-1510.

Ross, R.M., Bennett, R.M., Snyder, C.D., Young, J.A., Smith, D.R., Lemarie, D.P., 2003. Influence of eastern hemlock (*Tsuga canadensis* L.) on fish community structure and function in headwater streams of the Delaware River basin. Ecology of Freshwater Fish 1, 60-65.

Ross, R.M., Redell, L.A., Bennett, R.M., Young, J.A., 2004. Mesohabitat use of threatened hemlock forests by breeding birds of the Delaware River basin in northeastern United States. Natural Areas Journal 4, 307-315.

Rowell, T.J., Sobczak, W.V., 2008. Will stream periphyton respond to increases in light following forecasted regional hemlock mortality? Journal of Freshwater Ecology 1, 33-40.

Royle, D.D., Lathrop, R.G., 1997. Monitoring hemlock forest health in New Jersey using Landsat TM data and change detection techniques. Forest Science 3, 327-335.

Royle, D.D., Lathrop, R.G., 2002. Discriminating Tsuga canadensis hemlock forest defoliation using remotely sensed change detection. Journal of Nematology 3, 213-221.

Ruiz-Arias, J.A., Tovar-Pescador, J., Pozo-Vázquez, D., Alsamamra, H., 2009. A comparative analysis of DEM-based models to estimate the solar radiation in mountainous terrain. International Journal of Geographical Information Science 8, 1049-1076.

Schomaker, M.E., Zarnoch, S.J., Bechtold, W.A., Latelle, D.J., Burkman, W.G., Cox, S.M., 2007. Crown-condition classification: a guide to data collection and analysis. Gen. Tech. Rep. SRS-102. Asheville, NC: U.S. Department of Agriculture, Forest Service, Southern Research Station, 78 p.

Snyder, C.D., Young, J.A., Lemarié, D.P., Smith, D.R., 2002. Influence of eastern hemlock (*Tsuga canadensis*) forests on aquatic invertebrate assemblages in headwater streams. Canadian Journal of Fisheries and Aquatic Sciences 2, 262-275.

Spaulding, H.L., Rieske, L.K., 2010. The aftermath of an invasion: Structure and composition of Central Appalachian hemlock forests following the establishment of hemlock woolly adelgid, *Adelges tsugae*. Biological Invasions DOI 10.1007/s10530-010-9704-0.

Stadler, B., Müller, T., Orwig, D., Cobb, R., 2005. Hemlock woolly adelgid in New England forests: Canopy impacts transforming ecosystem processes and landscapes. Ecosystems 3, 233-247.

Sullivan, K.A., Ellison, A.M., 2006. The seed bank of hemlock forests: Implications for forest regeneration following hemlock decline. Journal of the Torrey Botanical Society 3, 393-402.

Trotter, R.T., Shields, K.S., 2009. Variation in winter survival of the invasive hemlock woolly adelgid (Hemiptera: Adelgidae) across the eastern United States. Environmental Entomology 3, 577-587.

U.S. Department of Agriculture, Forest Service, 2008. Gypsy moth in North America: gypsy moth defoliation maps. [Online.] http://www.fs.fed.us/ne/morgantown/4557/gmoth/atlas/, accessed January, 2010.

Ward, J., Montgomery, M., Cheah, C., Onken, B., Cowles, R., 2004. Eastern hemlock forests: guidelines to minimize the impacts of hemlock woolly adelgid. U.S.

Department of Agriculture, Forest Service, Northeastern Area State and Private Forestry, Morgantown, WV, 32 p.

Waterman, J.R., Gillespie, A.R., Vose, J.M., Swank, W.T., 1995. The influence of mountain laurel on regeneration in pitch pine canopy gaps of the Coweeta Basin, North Carolina, U.S.A. Canadian Journal of Forest Research 11, 1756-1762.

Webb, B.W., Hannah, D.M., Moore, R.D., Brown, L.E., Nobilis, F., 2008. Recent advances in stream and river temperature research. Hydrological Processes 7, 902-918.

Weston, P.A., Harper, R.W., 2009. Potential of *Tsuga* spp. from western North America and Asia as replacements for eastern hemlock (*Tsuga canadensis*). Arboriculture and Urban Forestry 1, 5-9.

Whitledge, G.W., Rabeni, C.F., Annis, G., Sowa, S.P., 2006. Riparian shading and groundwater enhance growth potential for smallmouth bass in Ozark streams. Ecological Applications 4, 1461-1473.

Willacker, J.J. Jr., Sobczak, W.V., Colburn, E.A., 2009. Stream macroinvertebrate communities in paired hemlock and deciduous watersheds. Northeastern Naturalist 16, 101-112.