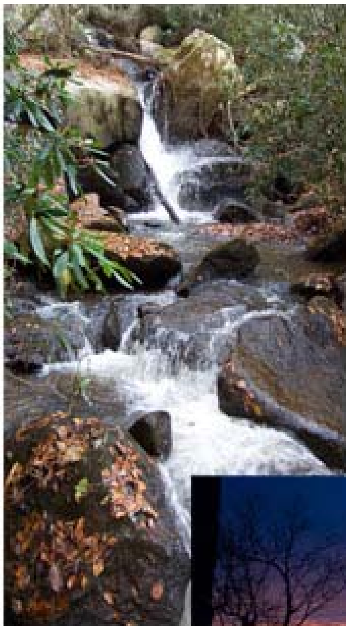

Land Use Change and Implications for Biodiversity on the Highlands Plateau

A report by the Carolina Environmental Program

Part A

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Contents

Introduction

In the Fall of 2004, twelve undergraduate students from the University of North Carolina at Chapel Hill had the opportunity to complete ecological coursework through the Carolina Environmental Program's Highlands field site. This program allows students to learn about the rich diversity of plants and animals in the southern Appalachians. The field site is located on the Highlands Plateau, North Carolina, near the junction of North Carolina, South Carolina and Georgia. The plateau is surrounded by diverse natural areas which create an ideal setting to study different aspects of land use change and threats to biodiversity.

The Highlands Plateau is a temperate rainforest of great biodiversity, a patchwork of rich forests, granite outcrops, and wet bogs. Many rare or interesting species can be found in the area, with some being endemic to a specific stream or mountaintop. Some of these are remnants of northern species that migrated south during the last ice age; others evolved to suit a particular habitat, with a slightly different species in each stream. Another reason the Highlands area is so diverse is its varied terrain, with high mountaintops, steep slopes, and rich coves. The sheer range of ecosystem types makes Highlands a unique and valuable area.

Before attempting to preserve the great biodiversity of the Highlands Plateau, it is necessary to have a thorough understanding of the area, its species, and its systems. Without this understanding, any conservation attempts are as likely to fail as to succeed. In addition to this, although much study has been done on the Southern Appalachians, there is still much that is not known. Because of its wide range of ecosystems, the Highlands Plateau makes an ideal study area for understanding the Southern Appalachians in general.

The Hemlock Woolly Adelgid (*Adelges tsugae* Annand) has recently added its name to a long list of historical threats to the biodiversity of the southern Appalachians. To mention a few, that list includes the introduction of non-native species, as is the case with the adelgid. For many, the chestnut blight and the balsam woolly adelgid are rather memorable examples of how introduced species drastically altered the biodiversity of the Southern Appalachians. The environmental threats posed by acid rain and ozone have further weakened the ecosystems of the southern Appalachians. Human disturbance such as logging and the creation of roads has negatively affected biodiversity due to fragmentation and loss of habitat. Sedimentation and poorly-managed riparian zones pose some of the greatest threats to aquatic communities. Thus it is important that we make ecosystem management decisions wisely. The combined effect of numerous historical forces has brought about a decline in the biodiversity of the Highlands Plateau and scientific studie is necessary for addressing these issues in the future.

Our studies focused on a diverse range of subjects which dealt with these threats to biodiversity on the Highlands Plateau. Amy Lorang worked with Tom Goforth to create a method of predicting plant distributions using GIS and then to test the method, she conducted a pilot study focusing on *Tsuga caroliniana*, or Carolina Hemlock. Josh Brown worked with Brian Kloeppel to predict the future of hemlock forests in western North Carolina at Coweeta Hydrologic Laboratory. Shelley Rogers worked with Patrick Brannon of the Nature Center, studying the potential effects of microhabitat change caused by HWA-induced hemlock defoliation on salamander populations. Katie Burke worked with Barry Clinton at the Coweeta Hydrologic Laboratory on carbon cycling and the importance of buffers in riparian zones. Michael Nichols, with the direction of Duke Rankin from the USDA Forest Service, studied threats to southern Appalachian bogs and examined management options to restore one such habitat in Dulany Bog. Megan Mailloux worked with Robert Tucker on the effects of mowing on wildflower populations along Horse Cove Road.

The six projects in this Capstone Report examined land-use change and the biodiversity of the Highlands Plateau. This unique place and its assemblage of organisms is the perfect out-door laboratory for environmental study. Many of these results and conclusions will hopefully influence future decision making processes and result in the formation of better informed policies. The Highlands Plateau, with its beautiful mountain views and pristine streams, deserves more than the admiration of every visitor passing by: it also requires the continued scientific study and conservation of its natural wonders.

Using GIS to predict plant distributions: a new approach

Amy Lorang

Introduction

Environmental conservation is becoming more and more important in modern society. With increasing pollution, development, and environmental change, the conservation of remaining wild areas is vital to the future. Unfortunately, it is not possible to conserve all areas; instead, those areas that are most important or rare must be conserved first. Many areas are chosen for conservation based on the presence of endangered or rare species, but it is often impossible to check each proposed area for presence or absence of targeted species. Given the fact that species distributions are related to the environment, it is possible to predict species distributions based on the environmental characteristics of a region. Scientists have been trying to define the growth requirements and predict species distributions for hundreds of years. As the task of defining growth requirements is often easier said than done, there have been countless studies done using many different environmental factors and finding very different results.

This study has two main objectives: (1) to propose a method to predict species distributions using a geographic information system (GIS) and (2) to conduct a pilot study in the Southern Appalachians as an example of this method. GIS, an application that allows maps to be easily edited on a computer, was used in this study because of its versatility and ability to connect data spatially. The Southern Appalachians region, more particularly the region around the Highlands Plateau, was chosen for the pilot study because of its highly variable terrain and high biodiversity. The study area includes Transylvania, Macon, Swain, Clay, Graham, and Jackson counties in southwestern North Carolina, Oconee and Pickens counties in northwestern South Carolina, and Rabun County in northeast Georgia (Figure 1). As will be later shown, although this pilot study contained is flawed and of limited usefulness, the method behind the theory is valid and deserves further exploration.

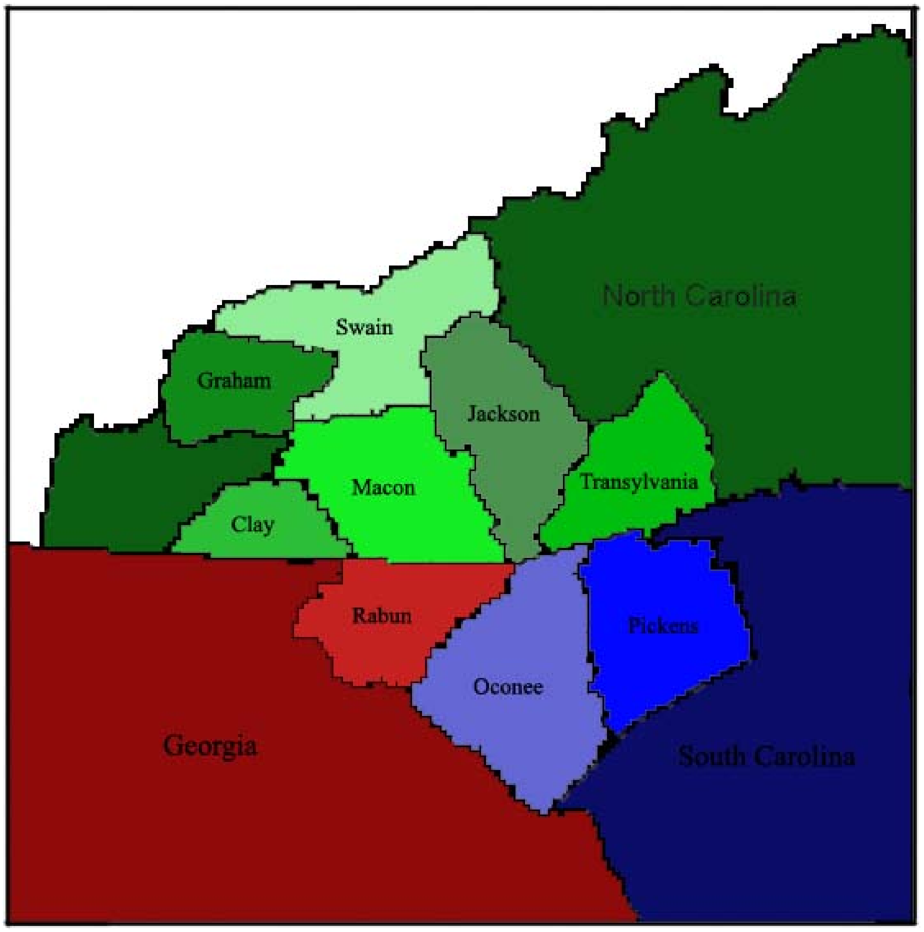


Figure 1: Map of study counties

Objective 1: Study Method

Background Information

There are two main approaches to saving the environment: the landscape-level, or from the top down, and the species-level, or from the bottom up. Each approach has merits and drawbacks. Overall, the landscape, or community, level of research and protection is good because landscapes are made of many interrelated processes, and this approach deals with them holistically. Landscape-level research also allows for the protection of a majority of species "without laborious individual attention" (Schafale and Weakley 1990). On the other hand, landscape-level research often takes more time, effort, and money than species-level research because it is on a larger scale, so it is often difficult to obtain funding for landscape-level research. Protecting or researching individual species is convenient because it is much easier to find the time and money to do research on a smaller scale. Therefore, research on the species-level is more likely to be completed. Moreover, when protecting species it is possible to focus on "rare species that occur in only a fraction of... their habitat" (Schafale and Weakley 1990).

Given the fact that there are benefits and drawbacks to both landscape-level and species-level research and protection, it is not surprising that both have been studied extensively. One example of community-level research is the "Classification of Natural Communities of North Carolina, Third Approximation," (Schafele and Weakley, 1990), which is an extensive and detailed list of natural communities and their features. According to this study, classifying communities results in greater ease in record keeping, breaks complexity into manageable units, and allows for generalizations about natural communities. These community categories, however, are still broad and contain variation. In addition, there are rarely discrete boundaries between communities because many plant species occur along continuous gradients that blur the lines between communities (Schafale and Weakley 1990).

The "All Taxa Biodiversity Inventory" study, or ATBI, in the Great Smoky Mountains National Park is one example of basic species-level research. According to the ATBI prospectus written in 1997, "park rangers can never fully appreciate park ecosystem losses, threats, and restoration opportunities until they know most of what lives in their park, its relative abundance, distribution, and general ecological role" (Peet et al. 1997). Unlike more specific studies which focus on a single species in great detail and try to answer questions about it, ATBI attempts to inventory all of the organisms in an area. More specifically, the goals of ATBI are to document species, map general taxon distribution and relative abundance, and research the ecological role, natural history, and potential uses of each species (Peet et al. 1997). This type of study attempts to define an area by defining its smallest parts (i.e. each species). Although this approach may seem to be the antithesis of the community-level approach, they are simply different ways of defining a landscape for conservation purposes.

This study combines the landscape-level approach and the species-level approach. On the one hand, it focuses on individual species, thus making it possible to collect very detailed information. On the other hand, by looking at this species on a landscape scale, broader information about communities and the area as a whole is obtained. The specificity of the study, as well as the fact that all data comes from sources that already exist (rather than new data being gathered for this study), makes this method relatively inexpensive. By using GIS, it is possible to look at the landscape as a whole on a small scale, as well as looking at detail in a specific area on a larger scale. This study method combines the benefits of both landscape-level and species-level approaches by using GIS to look at the occurrence of selected species on a landscape scale.

Although GIS software such as ArcView or ArcGIS is relatively new, it is quickly gaining popularity and has already been utilized in a wide variety of scientific studies. For example, Soza et al. (2000) used GIS to map historical and recent collections of plants to identify "black holes" in the known flora of California. Using ArcView, they were able to obtain maps of overall collection intensity, distribution of specific taxa, collection activity by various collectors, and key regions warranting more attention. Bentrup and Leininger (2002) used GIS in a very different way by using soil type and climatic information to assess the best locations for growing particular crops. Spears et al. (1990) used GIS to assist with the integrated pest management of the gypsy moth, an invasive exotic species. Liebhold and Halverson (1990) tackled the gypsy moth problem from another angle using GIS to describe and analyze gypsy moth spatial dynamics. As illustrated by this diversity of studies, GIS is a versatile tool and is rapidly gaining popularity in the scientific community as well as in land planning and other areas.

Objective 2: Pilot Study

Background information

Geology

The diverse topography of the Southern Appalachians is a product of its long and complicated geologic history. The beginning of the Appalachians was over a billion years ago, when the supercontinent Rodinia rifted apart. During the next 700 million years, the continental crust stretched apart and drifted back together to form the second supercontinent, Pangea. During this time, new oceans, island arcs, and continental margins formed, resulting in large amounts of sediment being deposited in the area of the Great Smoky Mountains. As continental crust drifted back together these oceans eventually closed, and heat and pressure caused by crustal movement metamorphosed the sediments into new rock types. Rising magma also resulted in intrusive igneous bodies that are now exposed at Whiteside Mountain and many other parts of the Appalachian Mountains. Crustal collisions and subduction during four separate mountain building episodes thrust these rocks into tall mountains, the remnants of which now form the Appalachians. The Appalachians, thought to have originally been comparable to the Himalayas, were completed about 300 million years ago, when Pangea began to break apart (Pitillo et al. 1998). The Appalachians have been eroding since the late Permian Period.

The long and complicated formation of the Appalachians has resulted in a very complex geology in the region. The geology of the Appalachians in the Highlands region consists of folded meta-sedimentary, meta-volcanic, and meta-plutonic rocks bounded in the east by the Brevard Fault and in the west by the folded sedimentary rocks of the Valley and Ridge Provence. Highlands is in the central area of the Tallulah Falls Thrust Sheet, which is approximately 20 miles wide, trends northeast, and contains a sequence of biotite gneiss, metagraywacke, kyanite schist and gneiss, muscovite/biotite gneiss, hornblende gneiss, and amphibolite. Minor marble occurs along the Brevard Fault Zone.

Soils

Most of the soils in the Highlands region, derived from dominant felsic rocks, are acidic and support low floristic diversity. Soils derived from scattered layers of mafic amphibolite and hornblende gneiss are circumneutral and higher in floristic diversity. Mass movement of land (i.e. landslides and slow creep) results in a mix of soil and nutrients on slopes, at the base of slopes, and in coves and valleys elsewhere (Pitillo et al. 1998). This diverse landscape of rock and soil forms the base for the diversity of the Southern Appalachians.

Paleo-climate

The plants and animals of the Southern Appalachians are a product of Earth’s cycle of glacial and interglacial periods. During the last glacial period, glaciers in the north and a colder climate in the south forced plants and animals of the northern deciduous forests to move south. When the glaciers retreated, the return of a warmer climate resulted in these colder-climate organisms moving up mountainsides, where high elevations maintained northern-like climates. This resulted in “refugia,” or islands of species high on mountain tops (Pitillo et al. 1998). Because these mountain refugia were isolated for a long period of time, many species evolved to become endemic to particular mountains or to the Southern Appalachians. Because of the refugia caused by the last glacial period, the biota of the Southern Appalachians became a mixture of southern species at low elevations and northern species at high elevations.

Plant-Environment Interactions

Multiple factors determine the relationship between plant assemblages and the environment. Among these are elevation, water supply, soil nutrient availability, bedrock, climatic factors, and natural and human disturbance. Many variations in these ecological factors lead to the myriad different plant community types found in the Southern Appalachians (Pitillo et al. 1998). Microhabitats such as rich coves or high mountainsides host very different natural communities. Although ecological theory has long held that there is a “climax,” or final successional stage, current theory holds that various disturbances such as fires, storms, and disease result in a more “patchwork” pattern of succession. For example, pine forests are eventually replaced by hardwood forests, but disturbance can keep a forest from becoming homogeneous as patches are returned to an earlier successional stage. This combination of microhabitats and disturbance leads to great diversity, as different natural communities are further diversified into different successional stages.

Methods

As mentioned previously, there are as many methods to define species growth requirements and predict distributions as there are studies that have attempted it. In a paper discussing geomorphology in the Southern Appalachians, Abella (2003) mentioned at least three other papers that had attempted to quantify geomorphology as a way to predict species distributions (Abella 2003). Lipscomb and Nilsen (1990) focused on slope, aspect, topographic position, and irradiance when defining the distribution of rhododendron and mountain laurel (Lipscomb and Nilson 1990). Other environmental characteristics that have been used include the thickness of various soil horizons, elevation, and base saturation. In fact, as can be seen in Table 1, rarely do two studies agree on what factors should be used to define and predict plant distributions (Abella et al. 2003).

	Lipscomb and Nelson (1990)	Jones (1988)	Hutto et al (1999)	Abella et al (2003)	Carter et al (2000)	Hix and Pearcy (1997)	McNab et al (1999)
Slope	√	√					
Aspect	√					√	
Elevation	√				√		
Landform			√	√	√		
Base saturation							√
A horizon thickness						√	√
B horizon texture		√					
Oe and Oa horizon thickness			√				
Solum thickness				√	√		
Irradiance	√						

Table 1: Comparison of characteristics used to define plant distributions in selected studies

The species and soil data used in this pilot study were obtained from the North Carolina Vegetation Survey (NCVS). NCVS is a "confederation of plant ecologists" (Peet et al. 1997) that was established in 1987. The purpose of NCVS is to characterize the natural vegetation of North Carolina and South Carolina. More specifically, the objectives of NCVS are to describe and classify vegetation, interpret vegetation-environmental relationships, inventory vegetation, and monitor ecosystem conditions over the long term. NCVS has two main constituencies that have requirements that are sometimes at odds: the scientific and conservation communities. To reconcile their disparate needs, a new scientific protocol was created that was flexible enough to satisfy multiple uses (Peet et al. 1997). New NCVS plots are established and new data are obtained every year. The data collection protocol used in NCVS allows the data to be used in multiple and diverse ways. Because of this flexible protocol, NCVS data was ideal for this study.

The species chosen for this pilot study was *Tsuga caroliniana,* or Carolina hemlock. Unlike *T. canadensis,* or Canadian hemlock, *T. caroliniana* has a very restricted range. Both species are currently threatened by the hemlock woolly adelgid, an invasive species that eventually kills the trees that it attacks. The loss of hemlock trees would greatly alter forest composition in the Southern Appalachians. There are two methods that are currently being used to save hemlock trees: release of the a predator beetle, which preys specifically on hemlock woolly adelgids, and injection of trees with chemicals that kill the adelgids for a limited amount of time. Knowledge of where *T. caroliniana* grows would increase the efficiency of both efforts by allowing researchers to focus on those populations that are most critical.

The variables examined in this study were chosen for many reasons. There are two main groups of variables studied: soil characteristics and other environmental factors. General environmental variables used in many other studies include elevation, slope, aspect, minimum temperature, and precipitation. Of these, although minimum temperature and precipitation are critical in determining widespread species ranges, within the study area they varied little and so were not included. Slope was eliminated because the derived slope layer showed slopes to be either very steep or flat throughout the study area. Elevation and aspect relate to the land morphology, or land shape, as well as helping to determine the climate of an area. Soil factors were chosen not only because they are vital in determining plant growth in various ways, but also because they were assessed on many CVS plots (Table 2).

	Definition	How it affects plants	Units
Base saturation	% basic cations in soil	affects soil pH	%
pH	acidity or alkalinity of soil	chemistry of plant nutrient exchange with soil	Log 1/H+ concentration
Cation exchange capacity (CEC)	amount of exchangeable cations in soil	ease of plant nutrient exchange with soil	meq/100g
Organic matter	amount of organic matter	water holding, aeration, microorganisms, nutrients, and organic colloids	%
Percent silt, sand, and clay	proportion of particle size of soil	pore space size, surface area, soil oxygen, and drainage	%
Nitrogen (N)	macro-nutrient	required for normal growth, many functions	Ppm
Potassium (K)	macro-nutrient	fruit and seed production	Ppm
Phosphorus (P)	macro-nutrient	normal plant growth, many functions	Ppm
Density	soil weight per unit volume	availability of air and water and space for root growth	g/cc

Table 2: Detailed information on soil variables used in pilot study

Data Analysis

Environmental data for this study were obtained from various sources (Table 3). A database in ArcMap was created with general geographic layers (study states and counties, rivers, and topography), a regional geology layer, a layer for the NCVS plot occurrences of *Tsuga caroliniana*, and layers for each environmental factor for all NCVS plots in the study area.

	Source	Spatial resolution
State outlines	Environmental Systems Research	1:100,000
County outlines	Environmental Systems Research Institute, Inc. (ESRI)	1:100,000
Digital elevation model	U.S. Geological Survey (USGS), EROS Data Center	30 m
Landcover classes	U.S. Geological Survey (USGS), National Land Cover Dataset	30 m

Table 3: Metadata for downloaded GIS layers

Once added to ArcGIS, each environmental layer was edited in different ways to yield the most useful results. The occurrence of *Tsuga caroliniana* was added as a separate layer of x,y points based on latitude and longitude from the CVS data. To obtain layers for soil variables, the average of the soil data for each plot was found where there were multiple values tested for one CVS plot. Each variable was then added as a separate layer of x,y points based on latitude and longitude, then interpolated to raster to obtain continuous soil data for the entire study area. Finally, aspect was obtained from the elevation layer.

The first step in predicting the distribution of the study species was to determine the plant growth requirements based on the data. This was done by finding the range of values that the species was found on for each environmental factor. The raster calculator was then used to transform each layer of values to a layer of 1s and 0s, depending on whether that point fit within the range found. In other words, these layers predicted the presence or absence of the species for each environmental factor. These layers were then added in the raster calculator to yield a final map of the predicted presence or absence of *Tsuga caroliniana* based on all of the environmental variables.

Results

A comparison of the environmental ranges of *Tsuga caroliniana* found in this study with those from other sources revealed that data is patchy, and different sources sometimes contradict each other (Table 4). The elevation range (570-1210m) found in this study was more or less the same range as those from both the USDA Plants Database and "Silvics of North America," by the Forest Service, with *T. caroliniana* being found at slightly lower elevations in this study. The range of aspects (NW-SE) from this study likewise matched that from the USDA. Finally, the pH range (4.0-5.1) found was similar to the USDA range, although both the minimum and maximum pH values were lower than the USDA values. In general, the ranges of environmental values found in this study matched those from both the USDA and the Forest Service.

The final map obtained from the GIS analysis shows the predicted distribution of the study species, based on the NCVS data used. Each point on the final map that predicts the presence of the species falls within the range of each environmental variable, while those points that predict absence fall outside of the range of at least one variable. A comparison of the predicted distribution map from this study, Figure 2, with that from the USDA, Figure 3, shows that the predicted distribution from this study is more limited than that from the USDA. The USDA map shows the presence or absence of the species in the county as a whole; that is, one sighting of the target species in a county is enough to count that county as containing the species. The USDA map shows the presence of *Tsuga caroliniana* in every county in the study area. The distribution map from this study, on the other hand, is more precise, and predicts the exact locations within the study counties where *T. caroliniana* is likely to occur.

	USDA	**Forest Service***	**This study**
Elevation (m)	610 – 1520	700 – 1200	570 – 1210
Aspect	N and E		NW – SE
Landcover			deciduous forest, evergreen forest, and mixed forest
Base Saturation			25 – 50
pH	4.2 – 5.9	“highly acid”	4.0 – 5.1
CEC			4 – 8
% clay			4 – 15
% silt			13 – 27
% sand			64 – 81
Density			0.5 – 1.0
Potassium (K)			43 – 131
Phosphorus (P)			2 – 54
Nitrogen (N)			29 – 64
Organic matter			5 – 30
* This data refers to *Tsuga Canadensis*			

Table 4: Environmental growth requirements for *Tsuga caroliniana*

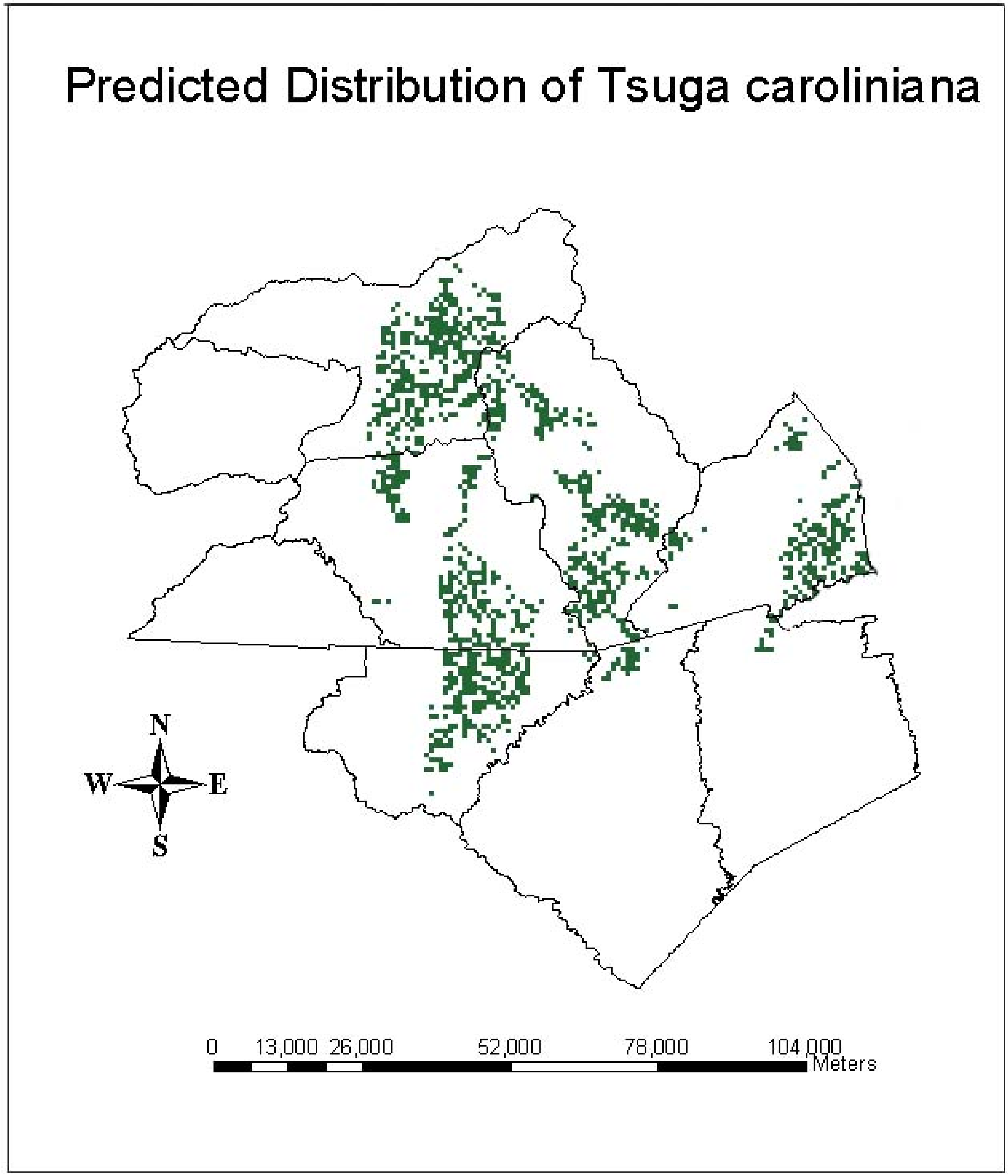


Figure 2: Predicted distribution of *Tsuga caroliniana* from this study

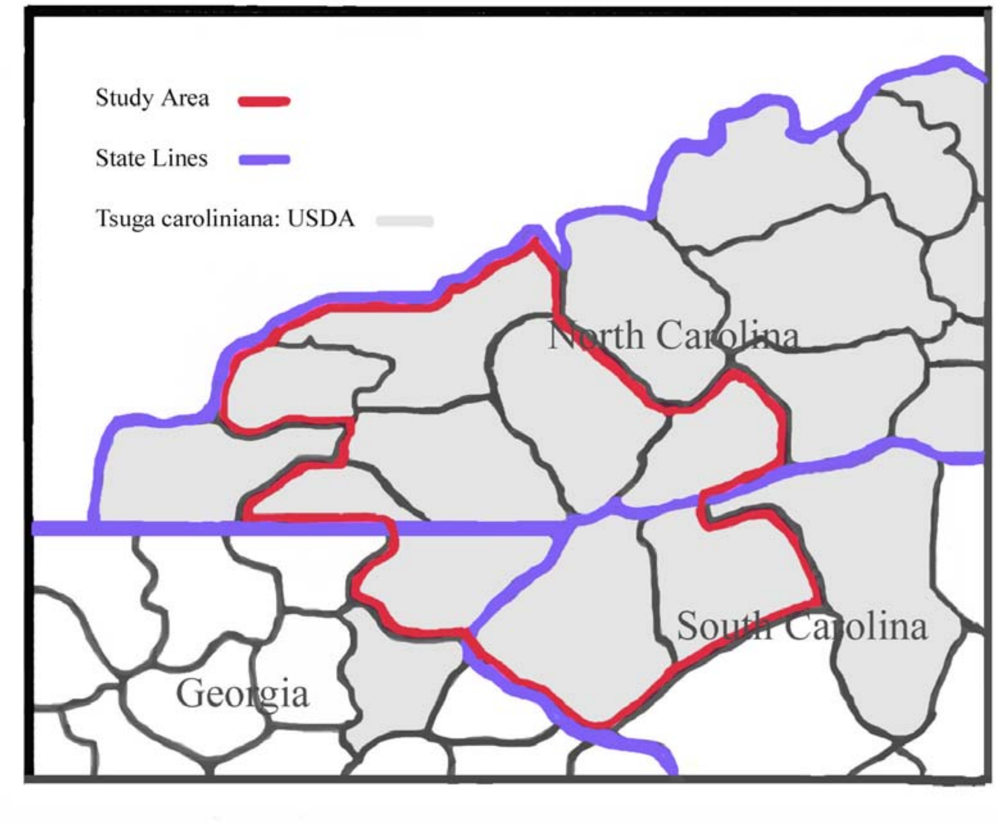


Figure 3: Distribution of *Tsuga caroliniana* from the USDA

Discussion

The environmental ranges for *Tsuga caroliniana* predicted more or less match the ranges from both the USDA and the Forest Service. The predicted distribution map from this study further refines the distribution map from the USDA by not only predicting presence/absence in a county, but showing where in the study area *T. caroliniana* is most likely to be found. Note, however, that this map shows only the *predicted* distribution of the species, *based on the data available in the study*. There are several factors in this study that limit its practicality and usefulness. The first of these is the small amount of NCVS data used in the analysis. An analysis based on only 15 plots includes a large amount of error simply because of the small sample used. Another source of error in this study is the soil data used. For the NCVS plots that had soil data, the results of analysis were fairly accurate. Much of the study area, however, contained NCVS plots that had no soil data, and the accuracy of the interpolated data are therefore questionable. The first step in eliminating error in this study would be to improve the quality and quantity of the source data used.

The next step in this study would be to test the distribution maps by creating new plots in areas predicted to have *Tsuga caroliniana*, and determining if the *T. caroliniana* found is where it was predicted to occur. The ArcGIS models would then be edited based on these new data. Only after this process is repeated several times could the distribution maps constructed by this method be considered accurate.

Predicted distribution maps constructed with this study method using GIS could be used in several ways. Researchers or environmental groups studying or trying to preserve certain species could identify probable sensitive areas. After verifying that the species did actually grow in the predicted location, plots could be surveyed, or negotiations initiated to acquire the land parcel. Similarly, developers could predict the presence or absence of rare or endangered species on a land parcel being considered for development before any work was started. Because maps constructed with this method show only the *predicted* distribution of species, they cannot be used as an end unto themselves. Nevertheless, using maps of specific predicted distributions should save time and money by giving all interested parties a place to start when looking either for individual species or at a specific parcel of land.

This study is just one example of the versatility of GIS. Although this study was limited by a shortage and incompleteness of available data, the theory and techniques used are valid. Application of this study method could aid scientific researchers, environmental groups, and land developers, among others. This method would not be possible without GIS technology and is only one example of the wide application of those technologies. Wider use of GIS in research and planning can only improve the efficiency and accuracy of both.

References Cited

Abella, S.R., Shelburne, V.B., and MacDonald, N.W. 2003. Multifactor classification of forest landscape ecosystems of Jocassee Gorges, southern Appalachian Mountains, South Carolina. Canadian Journal of Forest Research 33:1933-1946.

Abella, S.R. 2003. Quantifying ecosystem geomorphology of the Southern Appalachian Mountains. Physical Geography 24(6):488-501.

Bentrup, G. and Leininger, T. 2002. Agroforestry: mapping the way with GIS (geographic information systems). Journal of Soil and Water Conservation 57(6):150A(6)

Carter, R.E., Meyers, N.J., Shelburne, V.B., and Jones, S.M. 2000. Ecological land classification in the high rainfall belt of the southern Appalachian Mountains. Castanea, 65: 258–272.

Hix, D.M, and Pearcy, J.N. 1997. Forest ecosystems of the Marietta Unit, Wayne National Forest, southeastern Ohio: multifactor classification and analysis. Canadian Journal of Forest Research 27:1117–1131.

Hutto, C.J., Shelburne, V.B., and Jones, S.M. 1999. Preliminary ecological land classification of the Chauga Ridges region of South Carolina. Forest Ecology and Management 114: 385–393.

Jones, S.M. 1988. Old-growth forests within the Piedmont of South Carolina. Natural Areas J. 8: 31–37.

Langdon, K.R., Francis Jr., P.A., Parker, C.S., Pickering, J. and Ballentine, K.C. "All taxa biodiversity inventory of the Great Smoky Mountains National Park and other national Parks." 1997. 21 Nov 2004. <http://www.discoverlife.org/sc/gsmnp-atbi.prospectus.html>

Liebhold, A. M. and Halverson, J.A. 1990. Using GIS to describe and analyze gypsy moth spatial dynamics. Resource Technology 90: 684-692.

Lipscomb, M.V. and Nelsen, E.T. 1990. Environmental and physiological factors influencing the natural distribution of evergreen and deciduous ericaceous shrubs on northeast- and southwest- facing slopes of the Southern Appalachian Mountains. American Journal of Botany 77(4):517-526.

McNab, W.H., Browning, S.A., Simon, S.A., and Fouts, P.E. 1999. An unconventional approach to ecosystem unit classification in western North Carolina, U.S.A. Forest Ecology and Management 114: 405–420.

Peet, R.K., Wentworth, T.R., Duncan, R.P., and White, P.S. 1997. The North Carolina vegetation survey protocol: a flexible, multipurpose method for recording vegetation composition and structure.

Pitillo, J.D., Hatcher, Jr., R.D. and Boul, S.W. 1998. Introduction to the environment and vegetation of the Southern Blue Ridge province. Castanea 63(3):202-216.

"Plants Database." *Natural Resources Conservation Service.* 22 Nov 2004. 21 Nov 2004. <http://plants.usda.gov>

Schafale, M.P. and Weakley, A.S. 1990. Classification of the natural communities of North Carolina—3rd approximation. North Carolina Natural Heritage Program.

"Silvics of North Carolina." *United States Department of Agriculture, Forest Service.* 22 Nov 2004. Dec 1990. <http://www.na.fs.fed.us/spfo/pubs/silvics_manual/table_of_contents.htm>

Soza, V.L., Boyd, S., and Sanders, A.C. 2000. Phytogeographic "black holes" in southern California botany, a geographic information systems (GIS) model based on herbarium collections of two representative genera, Camissonia and Salvia. The American Journal of Botany 87(6): S180

Spears, B. M., Dull, C.W., and Rubel, D.N. 1990. Use of a geographic information system in gypsy moth integrated pest management. Resource Technology 90: 693-702.

Impacts of Hemlock Woolly Adelgid on Canadian and Carolina Hemlock Forests

Josh Brown

Introduction

Hemlocks are stately conifers that inspire awe in humans and play critical roles in forest ecosystems. The ten species of the *Tsuga* genus are found in cool temperate areas of North America and East Asia. Two species grow side-by-side in the Southern Appalachians; the Canadian hemlock (*Tsuga canadensis* (L.) Carr.) and the Carolina hemlock (*Tsuga caroliniana* (Engelm.)). Canadian hemlock is known throughout its range as a park tree, a landscape center piece, and most of all as a forest giant. Carolina Hemlock, on the other hand, is not a well-known tree. Most people have never seen one and would fail to recognize it even if they did. Growing high above the populated valleys of the mountains, the Carolina hemlock clings to high-elevation bluffs and rock outcrops. Both of these species play a vital role in Southern Appalachian biodiversity and both are presently threatened by an invasive exotic species. To further the understanding of these beautiful trees, the pest that now plagues them, and their unique habitats, this study was conducted in western North Carolina in Fall 2004.

The Canadian hemlock, also called the eastern hemlock or hemlock spruce, has a wide range in eastern North America. It grows from northern Alabama to southern Quebec. Its east-west distribution is from Minnesota to Nova Scotia (Harlow 1996). The species grows along the entire range of the Appalachian Mountains. Isolated stands also grow along the eastern seaboard and upper Midwest, where there are microclimates conducive to hemlock growth. Canadian hemlock is generally restricted to regions with cool, humid climates. It grows from sea level to 730 m in the Northeast and from 610 to 1,830 m in the Southern Appalachians (Harlow et al. 1996). It is found in mixed and pure stands in moist valleys, coves, ridges, and lake shores. In the mountains it is often restricted to north and east slopes (Coker and Totten 1934). This tree is found on many different soil types, growing in sandy loams to glacial till. Optimal soil conditions, however, are moist to very moist with good drainage.

The Carolina hemlock has marked differences in habitat characteristics from Canadian hemlock. The two species rarely co-occur in the same area due to their dissimilar habitat requirements. Carolina hemlocks compete best on dry, rocky ledges and cliffs, whereas Canadian hemlock require more mesic conditions (Rentch et al. 2000). The Carolina hemlock also favors south-facing slopes, the complete opposite of the Canadian. Moreover, Carolina hemlock has a very limited range, being found only in North Carolina in large numbers with scattered populations in South Carolina, Virginia, Georgia, and Tennessee (Figure 1). Its average elevation range in the mountains of these states is between 700 and 1,220 m (Duncan 1988). Overall these trees have a scattered distribution across the landscape. In addition to occurring in their typical habitat, they also occasionally occur in more mesic sites and as an understory tree on ridge tops.

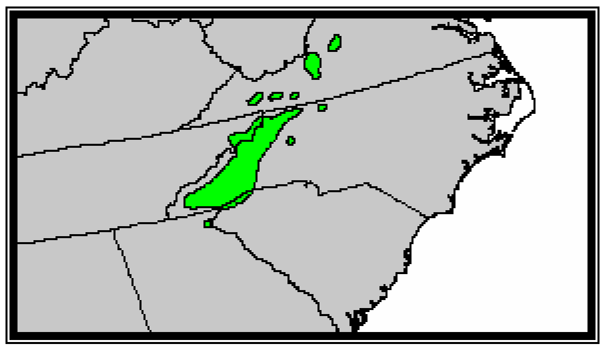


Figure 1: USGS map showing the distribution of Carolina hemlock (created in 1971).

The threat that these two trees now face is an invasive exotic insect called the Hemlock Woolly Adelgid (*Adelges tsugae*). The adelgid (HWA) is native to East Asia and was originally introduced to the West Coast of the United States in 1927 (Cullina 2002). The western hemlock species, *Tsuga heterophylla* and *Tsuga mertensiana,* are resistant to the HWA and were able to cope with this new pest. Canadian and Carolina hemlocks are not resistant and suffer severely when infested. In 1951, the HWA was accidentally introduced into eastern Virginia, probably on nursery stock from California (US Department of the Interior 2000). The adelgid made its way west, north, and south until it came to the rich forests of Appalachia where the Canadian hemlock is a dominant tree. With a seemingly endless food supply and no natural enemies the HWA population exploded. It was not long after that the first Carolina hemlocks became infested. The adelgid has been steadily increasing its range since the 1990s (Figure 2). By traveling on birds and nursery stock the HWA has been able to jump ahead of its own front to infest outlying counties, such as Monroe County, NY and Westmoreland County, PA. It arrived in Surry County, NC in 1995 and from there spread to most of the state. During the course of this study, I discovered the adelgid near Pickens, SC and reported it to the district ranger as the first sighting in that county. At this time there is no way to stop the spread of HWA southward. It is likely that the insect will completely engulf the Canadian hemlock's range in the Southern Appalachians, as it has already spread across the entirety of the Carolina hemlock's range.

The two hemlock species addressed in this paper are now in serious danger of extinction in the Southern Applications. The Hemlock Woolly Adelgid is showing no signs of stopping. In the mild climate of the southern mountains, the HWA can infest and kill a tree within four to six years (Vose personal communication). This unnatural event has put pressure on the scientific community to conduct research on both the adelgid and the hemlocks. Consequently, this study was conducted to help further the investigation of this environmental problem. The questions I addressed are: (1) where do Carolina hemlocks grow in the Highlands area and what site characteristics are common? This action seemed prudent since this tree is much more likely to go globally extinct in the near future. A distribution map will assist researchers in future surveys of the decline of this species. (2) What are the co-dominant tree species of Canadian and Carolina hemlock forests? (3) Which species are likely to take the place of the two hemlocks in their respective habitats? Using a list of co-occurring species, their density, and their basal area, a prediction will be possible for which species will thrive when hemlocks disappear from the landscape.

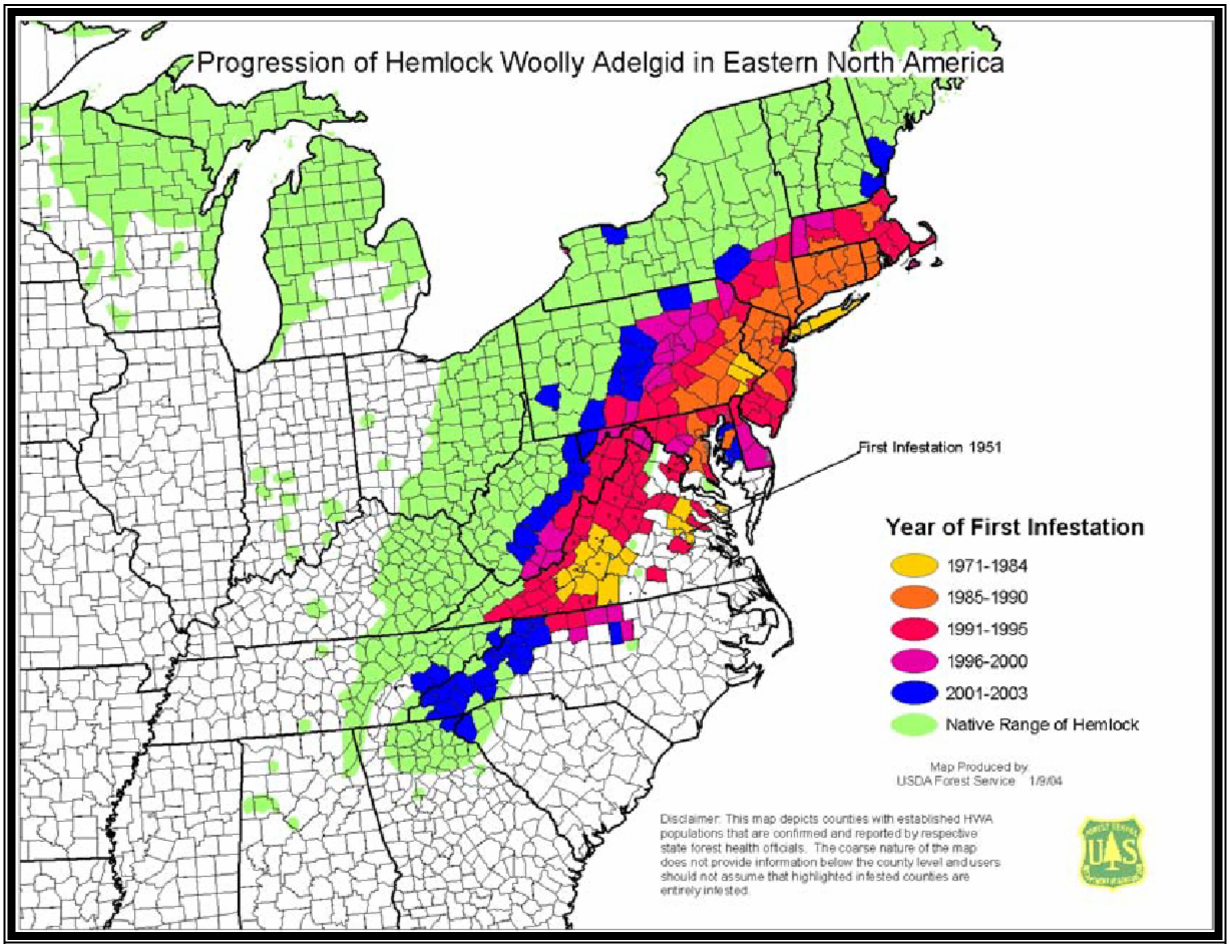


Figure 2: Forest Service HWA distribution map last up-dated on 1/9/04.

Study Sites

The Coweeta Basin is located in the Southern Appalachians, just south of Franklin, NC in Macon County. The basin was first settled in 1842. Light semiannual burning and grazing took place in the watershed until 1909. From 1909-1923, the more accessible valleys and lower slopes were logged. The Forest Service purchased the land in 1924 and prevented further unsustainable resource extraction from the basin (Monk et al. 1985). For the last 80 years the only anthropogenic disturbances in Coweeta have been experimental studies conducted by USDA Forest Service and university researchers. The parent rock of the basin is Carolina gneiss. The mean annual temperature is 13˚ C and annual precipitation ranges from 1,813 mm to 2,500 mm for the area. The elevational range of the basin is 685 m to 1,590 m.

Devil's Courthouse is located near the town of Highlands, NC. It is a mid-elevation granite outcrop on Forest Service land. The soil is very shallow, being derived from granitic gneiss, which is extremely resistant to erosion (Zahner 1994). It is also acidic due, in part, to the large input of hemlock needles into the top soil. The outcrop has a northeastern aspect (17˚). The harsh conditions found at this location have limited human impacts. There is, however, a well-used trail that leads to this outlook. This trail's presence results in heavy foot traffic that may have destroyed or suppressed herb and sapling growth. Devil's Courthouse is part of the Cowee Range and is situated next to the well-known landmark of Whiteside Mountain. The elevation of this rock promontory is approximately 1,382 m. The average rainfall for this site is not known, but it is estimated to be > 2,500 mm per year (Zahner 1994).

The third study site is similar to Devil's Courthouse. The site is located on Black Rock Mountain, another granite outcrop in the Cowee Range. Black Rock's ridge serves as the border between Macon and Jackson County, NC. It is on Forest Service land and has a foot-trail running across its summit. The trail is not heavily used and therefore current human influences are limited. Black Rock was created in the same mountain building event as Devil's Courthouse over 300 million years ago and it too is made of granitic gneiss (Zahner 1994). The elevation of the site is 1,318 m and the aspect is southwest (216˚). The average precipitation for this mountain is likely similar to the town of Highlands, averaging 2,286 mm, since it is located only a mile away. (Refer to the Carolina Hemlock Map for more precise locations of the above sites.)

Tree Characteristics

To identify study sites, the two trees in this project had to be recognized instantly in the field. Specific characteristics were learned, and both growth habit and appearance were memorized. Initially, an experienced botanist was consulted to help with the field identification of both trees.

Canadian hemlock has several characters useful to identify it in the field. It typically grows between 24 and 30 m tall and spreads 8-12 m. Its bark is brown to purplish and deeply furrowed when mature. The needles are distinctly flattened in two ranks on either side of the stem (Figure 3A). They are dark green on top with two mint green strips on the bottom. The needles average about 10-15 mm in length (Cullina 2002). The cones are yellow, purple, brown, or gray and are about 15-20 mm long (Duncan 1988).

The needles of Carolina hemlock resemble those of the Canadian hemlock except for their arrangement around stems; they come off all sides of the stem and point in all directions (Figure 3B). This leaf arrangement makes the tree look shaggier and more spruce-like. This is also a trait shared with Mountain hemlock (*Tsuga mertensiana*), a closely related species of the Pacific Northwest (Cullina 2002). The leaves are 15-18 mm long and the cones, which are yellow to light brown, are 25-35 mm long. The cones are also narrowly elliptical with scales that are longer than they are broad. Cones are one of the most useful ways to identify the tree if branches are not low enough to examine.

A. 

B. 

Figure 3: A. *Tsuga canadensis* branch with two flat rows of needles. B. *Tsuga caroliniana* stem where the needles come off in all directions. (pictures taken by author on 11/14/04 and 5/20/04 respectively)

Materials and Methods

To study Canadian hemlock communities, 12 plots of 20 x 20 m were installed in the Coweeta Basin. The location of each plot was randomly selected within a riparian area. The 12 plots were clustered into three groups at different locations in the basin. Four of these plots were placed at a low elevation site near Coweeta Creek, where few to no Canadian hemlocks grew.

These plots, dominated by a hardwood oak forest, were used as the control group to insure that all variables were accounted for. The other two sets of four were situated higher up the basin. One group followed the Shope Fork Creek, while the other followed Ball Creek. Due to topography, plots were arranged in a somewhat linear manner up an elevational gradient.

In each plot all woody vegetation >1 cm in diameter at 1.37 m above the soil was identified and numbered. Diameter at breast height (DBH) was also measured for each stem that met the above specifications. For all trees with a diameter >15 cm a permanent dendrometer band was installed. Each band consists of a long, thin piece of stainless steel, a spring, and two measuring points. These devices expand and contract with the tree and are the most precise way to measure diameter growth. The installation of dendrometer bands will facilitate measuring the trees in the future. Metal tags were attached with aluminum nails or wire to all trees in the plot to permanently number them.

Of the eight Canadian hemlocks plots at Coweeta, a treatment was randomly applied to half. The treatment, in this case was, the purposeful extermination of the hemlocks. Four plots were randomly selected and every hemlock within the plot was girdled. The trees were girdled in August 2004, so most were still standing and retaining needles at the time of my survey. The trees, deprived of water and nutrients, will defoliate by next year and eventually fall to the forest floor. This treatment was designed to simulate future loss of Canadian hemlock due to HWA. The results of this long-term experiment will yield valuable data on the effects of hemlock loss in a riparian area.

At Devil's Courthouse and Black Rock Mountain, representative Carolina hemlock stands were surveyed using similar methods to the survey of Canadian hemlock at the Coweeta field sites. A 10 x 10 m plot was used because of the limited size of the stand. The plot was oriented in such a way as to encompass a majority of the trees present at the site. Diameters at breast height and species composition were recorded in an analogous manner to the Coweeta plots. Dendrometer bands and permanent number tags were not installed, since regular follow-up data gathering is not probable. The site proved to be more difficult to survey due to thick underbrush and steep terrain.

In the process of generating a map of Carolina hemlock distribution in the area several data gathering methods were used. First, local forestry experts and naturalists were consulted. These authorities on mountain flora identified several locations where Carolina hemlock occurs. After these areas were visually surveyed and found to contain hemlocks, data collection began. A hand-held Geographic Positioning System (GPS: Garmin Etrex) device was used to find the coordinates of the center of each stand or tree. Elevation was measured using an altimeter that was calibrated to local conditions. Aspect was determined using a digital compass. The level of HWA infestation was assessed and given a numerical value for each site (Table 1). Finally, a clinometer was used to determine the varying degrees of slope found at each site.

Value	Definition
0	No HWA or old wool encountered on tree
1	Fewer than 10 individuals on a tree
2	<50% of branches exhibit HWA infestations
3	50-75% of the branches exhibit HWA infestations
4	75-100% of the branches exhibit HWA infestations

Table 1: The four levels of HWA infestation found on Carolina hemlocks.

Results

The primary results of this study show the typical species composition of Canadian and Carolina hemlock forests, along with their density and basal area. The high biodiversity of the Southern Appalachians manifested itself in the great number of tree species recorded in all the surveyed plots (Table 2). A total of 23 tree species were documented. Although none of the species surveyed were endemic to the area, the assemblage of northern and southern species is found nowhere else but the Southern Appalachians. For example, Northern Red Oak (*Quercus rubra*) and Sugar Maple (*Acer saccharum*), typical Northeastern species, were found in the same plots as Sourwood (*Oxydendrum arboreum*) and Black Gum (*Nyssa sylvatica*), trees common to the South. There were some differences between plots based on their location, terrain, and distance from streams. The Canadian hemlock plots had more *Rhododendron maximum* and

Black Birch (*Betula lenta*), whereas the hardwood plots contained more Dogwood (*Cornus florida*) and Red Maple (*Acer rubrum*). This difference had to do with the preference Rhododendron and Black Birch have for riparian areas. Carolina hemlock plots contained the more drought-resistant White Pine (*Pinus strobus*) and Northern Red Oak (*Quercus rubra*), which are typically found at higher elevations.

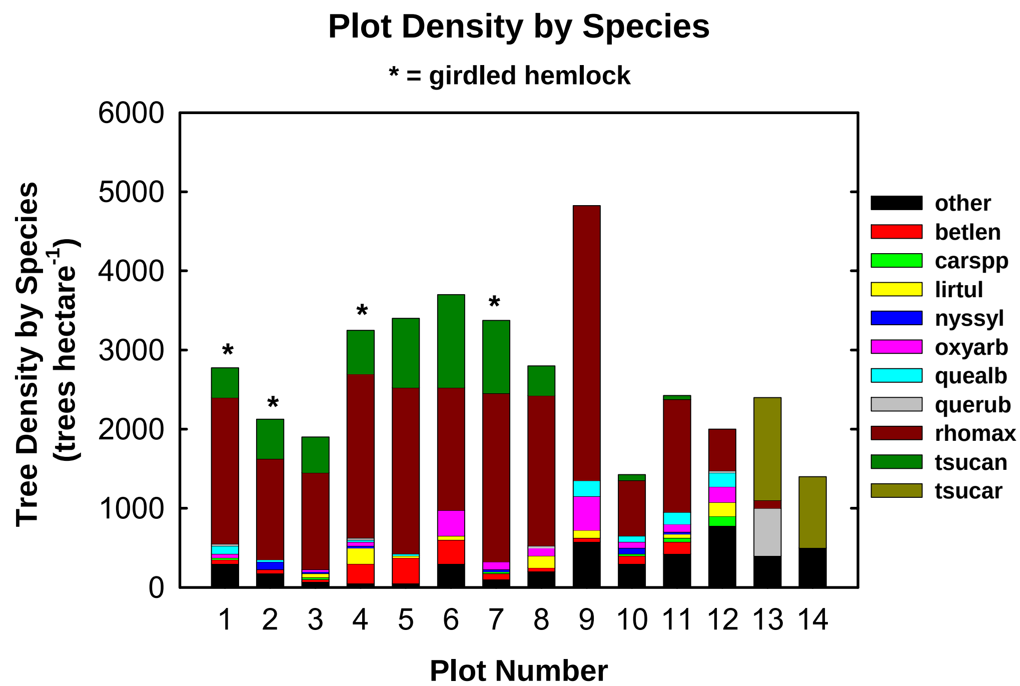
Carolina	**Hardwoood**	**Canadian**
Tsuga caroliniana	*Tsuga canadensis*	*Tsuga Canadensis*
Pinus strobus	*Pinus echinata*	*Pinus echinata*
Quercus rubra	*Quercus rubra*	*Quercus rubra*
Quercus prinus	*Cornus florida*	*Quercus prinus*
	Quercus alba	*Quercus alba*
	Nyssa sylvatica	*Nyssa sylvatica*
	Fagus grandifolia	*Sassafras albidum*
	Rhododendron maximum	*Rhododendron maximum*
	Quercus coccinea	*Quercus coccinea*
	Oxydendrum arboreum	*Oxydendrum arboreum*
	Liriodendron tulipifera	*Liriodendron tulipifera*
	Fraxinus spp	*Fraxinus spp*
	Carya spp	*Carya spp*
	Betula lenta	*Betula lenta*
	Acer rubrum	*Acer rubrum*
		Acer pensylvanicum
		Acer saccharum
		Amelanchier arborea
		Kalmia latifolia
		Hamamelis virginiana

Table 2: List of species composition for each community type. Species included were all at least 1 cm in diameter at 1.37 m in height.

The density of species for the Canadian and hardwood plots showed a very high concentration of Rosebay Rhododendron (*Rhododendron maximum*). This species was a majority of tree density in every Canadian hemlock plot surveyed in the Coweeta Basin (Figure 5). There were more individual Rhododendrons in these plots than all individuals of the other species combined. These evergreen shrubs create a very thick understory that makes sampling difficult. The growth habit of this large shrub allows it to have multiple trunks and therefore a very high density. The high density of Rhododendron is deceiving for it does not dominate the forest canopy; its growth is confined and limited to the understory. Canadian hemlock proved to be the second densest species in the Coweeta Basin Hemlock plots. The hardwood plots had high densities of both White Oak (*Quercus alba*) and Sourwood (*Oxydendrum arboreum*).

Canadian hemlocks were the dominant trees in their eight plots in terms of basal area. They made up more than half of the basal area on average in each Canadian hemlock plot (Figure 6). Carolina hemlock had both the highest density and basal area in its plots. These data show the clear dominance of these two species in these local habitats. Furthermore, it indicates that each plot was appropriately located in a climax forest where hemlock was the dominant tree (Figure 7). The hardwood plots are unquestionably dominated by White Oak (*Quercus alba*), as one might expect in a mixed oak forest.

The survey of Carolina hemlock in the Highlands area produced a reliable map and valuable data on the species (Appendix A and B). The survey was conducted between August and November 2004. More sites were surveyed in the latter half of the season due to the increased visibility of the hemlocks in the surrounding deciduous forest. In total, 168 trees were located in Macon and Jackson County, NC. Some trees were clustered in small stands, while others were found growing alone, unaccompanied by other members of its species. Geographic coordinates were taken at each site in Universal Transverse Mercators (UTMs). The point was taken at the center of each stand, as was judged by the surveyor, or if only one tree was found as close to the base of the tree as possible. Elevation, aspect, slope, and infestation level was recorded at all sites. If these characteristics differed within a stand, a separate survey was conducted on those dissimilar individuals. In the distribution map all of the sites surveyed within 50 m of each other are indicated by only one point. This was done to make the map more concise and visually comprehendible.



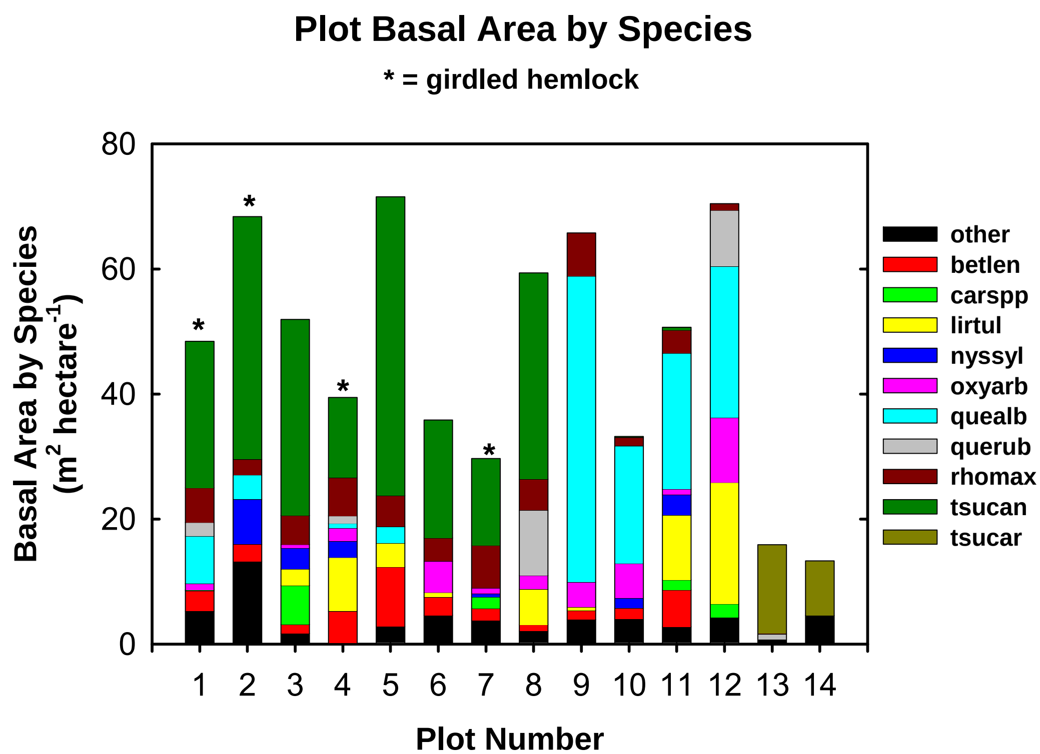


Figure 5 (top) and Figure 6 (bottom): Plots 1-8 are Canadian hemlock, Plots 9-12 are hardwood, and Plots 13 and 14 are Carolina hemlock. The asterisk symbolizes that the hemlocks in the plot below it were girdled.

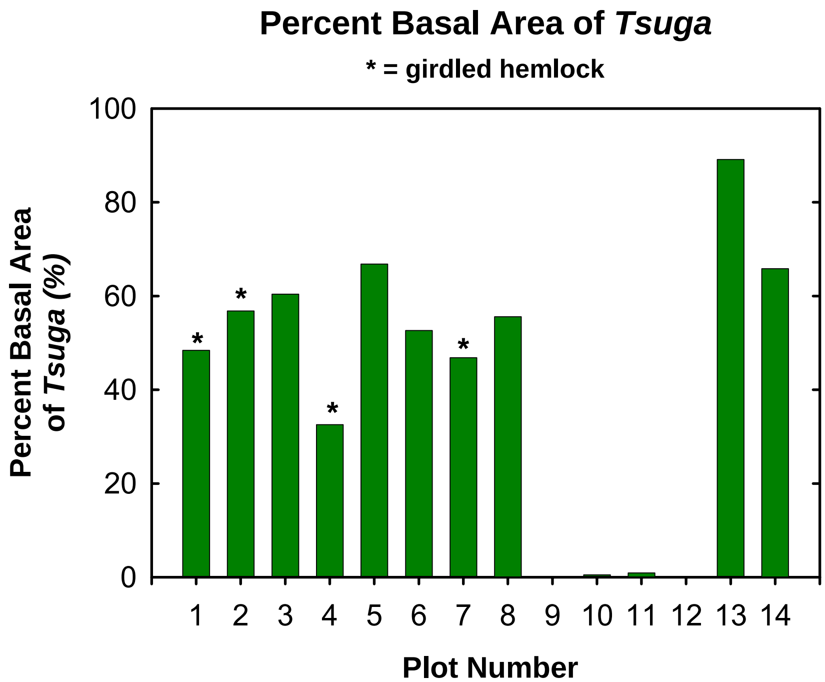


Figure 7: Percent basal area of hemlocks in all plots. Plots 1-8 are *Tsuga canadensis* plots. Plots 9-12 are hardwood plots. Plots 13 and 14 are *Tsuga caroliniana* plots. The asterisk symbolizes that the hemlocks in the plot below it were girdled.

From the characteristics collected some common observations can be stated. The average elevation of the sites was 1268 m, with the highest being 1395 m at Chimney Top Mountain and the lowest being 789 m at Whitewater River Gorge (Figure 8). This elevational range compares well to those observed by other researchers (Duncan 1988). The most common aspect of individual trees was southwest between 205˚-247˚. This also agrees with previously published literature (Coker and Totten 1945). The least infested stands at the time of the survey were Black Rock Mountain stands 1 and 2. The most infested stands at the time of the survey were Kelsey Tract stands 1-2, Rocky Mountain stands 2-3, and Devil's Courthouse stands 3-4 (Appendix A). Some branches were not low enough for inspection for the HWA therefore nothing was recorded for the infestation of these trees. Slope was not included in the final data, because of inconsistent measurement techniques.

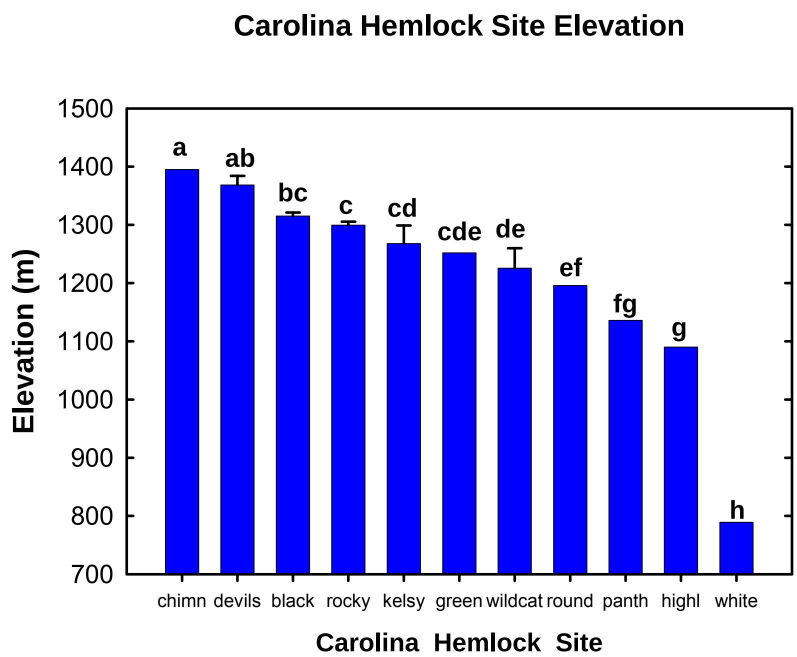


Figure 8: Average elevation and standard deviation for each Carolina hemlock site.

Discussion

After examining the data collected in this study, several trends are apparent. The basal area of all eight Canadian hemlock plots at Coweeta is dominated by Canadian hemlock and this will translate into a massive loss of a key canopy species in the near future. At the Coweeta field sites this will happen in the next few years in the plots that have been girdled, thereby allowing researchers an early opportunity to see what is to come. Researchers will be able to see the Canadian hemlock shrink from its impressive 49.9% mean basal area, as seen in the Canadian hemlock treatment plots, to 0% within a few years. When the Canadian hemlocks finally succumb to the HWA and die, huge gaps in the canopy will open. These gaps will probably be filled by any vegetation waiting in the understory that is capable of filling them. The most abundant plant in the understory is *R. maximum* shown in Figure 4 to be extremely dense in the Canadian hemlock plots with roughly 2000 stems per hector in each plot. *R. maximum* has been known to capitalize on canopy gaps created by the loss of tree species in the past. When American Chestnut (*Castenea dentata*) was killed by the Chestnut blight (*Cryphonectria parasitica*) in the 1930s, *R. maximum* greatly increased in abundance (Monk et al. 1985).

History may repeat itself in the Southern Appalachians. With the passing of the hemlocks, the opportunistic *R. maximum* may, to some extent, take its place.

A similar pattern is seen in the Carolina Hemlock plots. Although the data show that Red Oak (*Quercus rubra*), White Pine (*Pinus strobus*), and Chestnut Oak (*Quercus prinus*) are the only other dominant trees in density and basal area on the plots, it does not reflect all of the vegetation found there. The dwarf nature of the Carolina hemlock forest limits the understory to less then a meter in height; therefore, it was not surveyed. The dominant shrubs growing under and around Carolina hemlock are *R. minus* and *Kalmia latifolia*. These two plants were so thick in these plots that getting to each tree to survey was almost impossible. These dense shrubs would no doubt experience the same release that *R. maximum* would if canopy gaps were to increase (Monk at al 1985). The increase in available resources would be greater in a Carolina hemlock forest, since these trees make up a greater percent of the canopy than Canadian hemlocks do.

In both cases hemlock trees are likely to be replaced by an understory Ericaceae shrub. This kind of succession is new to the Appalachians and the long-term significance of this shift in plant composition is not known. *Kalmia latifolia*, *R. maximum*, and *R. minus* are known to reduce tree species recruitment in areas where they are already established (Monk et al. 1985). These evergreen species shade out any saplings growing under them and if left undisturbed by browsing or fire will maintain a monospecific stand. The leaves of these plants are poor in nutrients and decompose slowly, thus making the soil underneath them less productive. (Monk et al 1985) If these three plants do become dominant in former hemlock forests, then productivity and biodiversity will most likely decline.

Conclusion

The Southern Appalachians are about to experience a dramatic and irreversible change. They are losing two key species in the forest landscape. The repercussions of the decline or loss of Canadian and Carolina hemlock will ripple through the ecosystem for years to come. The Canadian hemlock, for instance, ameliorates the local microclimate, moderating temperatures within stands by creating cooler temperatures in the summer and warmer temperatures in the winter. Both hemlocks provide shelter and food for white-tailed deer (*Odocoileus virginianus*), hemlock angle moths (*Semiothisa fissinotata*), harvest butterflies (*Feniseca tarquinius*), bobcats (*Felis rufus*), red-backed voles (*Clethrionomys gapperi*), the northern water shrews (*Sorex palustris*), along with many other animals (US Department of the Interior 2000). With these beautiful trees gone, food chains will change, habitats will be altered, and the distribution of a variety of plants and animals will shift. If *R. maximum*, *R. minus*, and *Kalmia latifolia* increase dramatically after the hemlocks are gone, a new forest structure never before seen in the Southern Appalachians will dominate the landscape. A monoculture of unproductive heath thickets will cover the hills, stream banks, and rock outcrops. The appearance of the mountains will be forever changed when the hemlocks disappear, and the biodiversity of the Southern Appalachians will be reduced by yet two more tree species.

References Cited

Brown, C. and L. K. Kirkran. 1990. Trees of Georgia and Adjacent States. Timber Press: Portland, Oregon.

Coker,W. and H. R. Totten. 1934. Trees of the Southeastern States. The University of North Carolina Press: Chapel Hill.

Cullina, W. 2002. Native Trees, Shrubs, and Vines: a Guide to Using, Growing, and Propagating North American Woody Plants. Houghton Mifflin Company: Boston.

Duncan, W. and M. Duncan. 1988. Trees of the Southeastern United States. University of Georgia Press: Athens, Georgia.

Harlow, W., E. Harrar, J. Hardin, F. White. 1996. Textbook of Dendrology. McGraw-Hill: New York.

Monk, C., D. McGinty, F. Day. 1985. The Ecological importance of *Kalmia latifolia* and *Rhododendron maximum* in the deciduous forest of the Southern Appalachians. Bulletin of the Torrey Botanical Club 112: 187-193.

Rentch, J., H. Adams, R. Coxe, and S. Stephenson. 2000. An ecological study of a Carolina hemlock community in southwest Virginia. Castanea 65:1-8.

US Department of Interior National Park Service Northeast Region. 2000. “Draft environmental assessment for the release and establishment of *Pseudoscymnus tsugae* (Coleoptera: Coccinellidae) as a biological control agent for hemlock woolly adelgid (*Adelges tsugae*) at Delware Water Gap National Recreation Area.”

Vose, Jim. November 22, 2004. Coweeta Hydrologic Lab employee. personal communication.

Zahner, R. 1994. The Mountain at the End of the Trail. Highlands, North Carolina.

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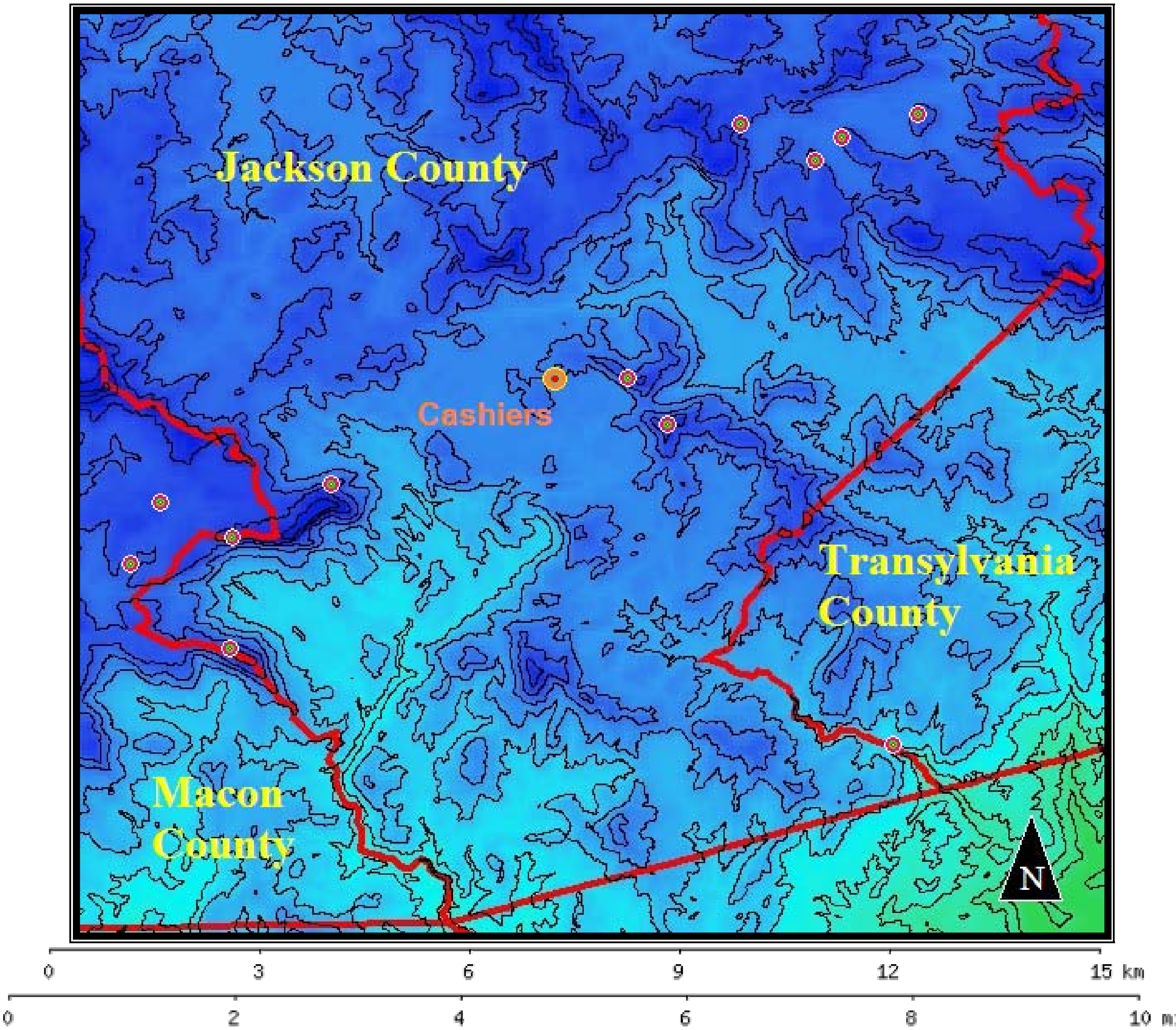
Appendix A: Carolina Hemlock Site Data

Sites 1-6 were found near the town of Cashiers. Sites 7-10 were found in the Panthertown Valley area. Sites 11-23 were found in the Cowee Range. The last site, number 24, was found in the Whitewater River Gorge.

	Name	Location (UTM)	#trees	Elevation(m)	Aspect	Adelgid(0-4)
1	ChimneyTopMt	17 0311813 3886217	3	1395	S 156˚	?
2	RockyMtstand1	17 0310990 3887469	9	1293	N 10˚	1
3	RockyMtstand2	17 0311027 3887456	9	1292	E 101˚	4
4	RockyMtstand3	17 0311026 3887428	5	1292	SW 220˚	4
5	RockyMtstand4	17 0311044 388709	2	1297	SW 239˚	?
6	RockyMtstand5	17 031141 3887328	1	1323	SW 239˚	?
7	SaltRockMt	17 0314740 3893428	3	1136	SE 120˚	1
8	LittleGreenTopMt	17 0316558 3893144	1	1244	W 250˚	1
9	BigGreentopMt	17 0315283 3892400	2	1252	SW 210˚	1
10	RoundMt	17 0314927 3890959	10	1196	SE 115˚	1
11	BlackRockstand1	17 0303694 3881303	9	1318	SW 216˚	0
12	BlackRockstand2	17 0303843 3881277	1	1322	NE 50˚	0
13	BlackRockstand3	17 0303880 3881210	1	1323	SE 130˚	?
14	BlackRockstand4	17 0303657 3881267	1	1298	S184˚	1
15	Devil'sCourtstand1	17 0305502 3884973	19	1382	NE 71˚	1
16	Devil'sCourtstand2	17 0305460 3884945	18	1386	NW 318˚	1
17	Devil'sCourtstand3	17 0305387 3884890	4	1384	NW 306˚	3
18	Devil'sCourtstand4	17 0305209 3884833	2	1322	SW 205˚	4
19	KelseyTractstand1	17 0301815 3884368	18	1299	SW 247˚	4
20	KelseyTractstand2	17 0302633 3884647	4	1237	S 200˚	3
21	HighlandFalls	17 0300905 3883591	3	1090	SE 153˚	1
22	WildCatCliffsstand1	17 0304332 3884662	4	1191	E 72˚	?
23	WildCatCliffsstand2	17 0304246 3883896	8	1260	NE 24˚	?
24	Whitewater	17 0316069 3878786	7	789	SW 210˚	2

Total # of trees --	168 individuals
Average Elevation --	1263 m
Most Common aspect for individual trees --	SW (205˚-247˚)
Least infested stands at the time of survey --	BlackRockMtstands1-2
Most heavily infested stands at the time of survey --	KelseyTractstands1-2 RockyMtstands2-3 DevilsCourtstands3-4

Appendix B: Carolina Hemlock Distribution Map-Fall 2004



Twelve Sites are shown on the map: five sites in the Cowee Range, two sites near the town of Cashiers, four sites in the Panthertown Valley area, and one site in the Whitewater River Gorge.

Pink target symbols show the location of each Carolina Hemlock stite.

Orange target symbol shows the location of the town of Cashiers.

Topography assembled from USGS Highlands, Cashiers, Big Ridge, and Reid quadrangles (WGS84/NAD83).

Effects of Adelgid-Induced Decline in Hemlock Forests on Terrestrial Salamander Populations of the Southern Appalachians: A Preliminary Study

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Introduction

The Southern Appalachians have faced a long list of threats to biodiversity over the past century. One of the most serious threats is the invasion of non-native species. For many, the chestnut blight fungus (*Endothia parasitica* (Murr.) P. J. Anderson and H. W. Anderson) of the early 1900s, is a compelling example of how an introduced species can drastically alter forest composition. It devastated not only the American chestnut population (*Castanea dentata* (Marshall) Borkh.), but also the fauna that depended on chestnuts as food or habitat for their survival, including the extinction of seven species of native moths (Davis 2000). The balsam woolly adelgid (*Adelges piceae* Ratz.), introduced in the 1950s, is another example of an invasive exotic species that has significantly altered forest composition, decimating Frasier Fir (*Abies fraseri* (Pursh) Poiret) stands and the unique communities of plants and animals that they supported (Bruck and Robarage 1988; Rabenold et al. 1998; Smith and Nicholas 1998). Decades after the introduction of the balsam woolly adelgid, the composition and structure of the stands are still in a state of recovery (Jenkins 2003).

The eastern hemlock (*Tsuga canadensis* (L.) Carr.), a shade-tolerant, late-succesional conifer with dense evergreen foliage, ranges throughout the Southern Appalachians. It is a medium- to large-sized tree that at maturity can grow to 20-30 m in height and up to 1 m in diameter (Brown and Kirkman 1990; Evans et al. 1996). Hemlocks inhabit the deeper, moist soils along streams, lower slopes, rocky hillsides, and ravines. The eastern hemlock is most commonly associated with mixed mesic hardwood forests, especially in riparian zones (Brown and Kirkman 1990; Cullina 2002). Dense hemlock stands provide for unique microhabitats by providing shade and an acidic layer to the soil that allows for a unique floral and faunal species composition and structure (Evans et al. 1996).

Recent attempts have been made to assess the biodiversity of hemlock-dominated forests (Mahan 1999). Many species of mammals such as the woodland jumping mouse (*Napaeozapus insignis* Miller) and red-backed vole (*Clethrionomys gapperi* Vigors), and amphibians including the red-backed salamander (*Plethodon cinereus* Green) are associated with hemlock forests, although none obligately so (Wydeven and Hay 1996; Mahan 1999). The microclimate of hemlock streams promotes greater diversity in fish populations than does that of hardwood streams due to the increase in invertebrate assemblages in hemlock riparian zones (Snyder et al. 2002; Ross 2003).

The hemlock woolly adelgid (*Adelges tsugae* Annand: hereafter "HWA") has recently joined the long list of historical threats to the biodiversity of the Southern Appalachians. The introduced East Asian HWA, an aphid-like species that withdraws sap from the base of hemlock needles, resulting in defoliation and eventual tree mortality, was first discovered in the Shenandoah National Park in 1988 and has caused a substantial decline in *T. canadensis* (Watson et al. 1994). The HWA, which has been widely dispersed by wind, birds, deer, and humans, is expected to affect the hemlock forest ecosystem in the eastern United States in its entirety (McClure 1990). Because smaller hemlock trees are also susceptible to HWA infection, no regeneration has been found (Orwig and Foster 1998).

The HWA not only damages *T. canadensis* populations but also alters the microclimates they create and the associated communities, leading to an overall reduction in landscape-scale biodiversity (Mahan, 1999; Evans, 2002). It has been predicted that HWA-induced canopy thinning will affect soil temperature and moisture and will likely change the hydrological cycles of streams in riparian areas dominated by hemlocks (Cobb and Orwig 2002; Evans 2002). Numerous studies have demonstrated a decline in animal populations due to the adelgid, including terrestrial vertebrates such as birds, and aquatic animals such as native brook trout (Evans et al. 1996; Mahan 1999; Snyder 2002; Tingley et al. 2002; Ross et al. 2004).

Due to their eco-physiological requirements, salamanders could be highly susceptible to microhabitat change in HWA-infested forests. The Southern Appalachians are renowned for their diversity of salamanders in terms of both species richness and evenness (Hairston 1949; Petranka et al. 1993). In many woodland habitats, salamanders are the most abundant forest-floor vertebrates (Petranka 1998; Petranka et al. 1994). Salamanders play important roles in food webs where they act as predators of small invertebrates and, in turn, are preyed upon by other vertebrates such as snakes, birds, small mammals, and other salamanders (Brodie and Howard

1973; Brodie et al. 1979; Pough et al. 1987). Members of the family Plethodontidae are lungless, terrestrial salamanders common to the Southern Appalachians (Petranka 1998). Plethodontids are at a greater risk of desiccation because they respire cutaneously (Spotila 1972; Feder 1983). Consequently, their microhabitat is restricted to moist soils and areas with generally cooler temperatures (Heatwole and Lim 1961; Hutchinson 1961; Brattstrom 1963; Jaeger 1980*b*; Feder and Lynch 1982; Feder 1983; Feder and Londos 1984; Dupuis and Bunnell 1999). Leaf litter represents important microhabitat for salamanders because it retains water and harbors a wide variety of abundant invertebrate prey (Gist and Crossley 1975; Fraser 1976; Ash 1995). Rotten logs or coarse woody debris (CWD) serve as a moisture refuge during periods of low humidity (Jaeger 1980*b*; Petranka et al. 1994; Butts and McComb 2000). Logs in the advanced stages of decay provide optimal microhabitats for salamanders because of their spongy texture and high concentrations of water (Maser et al. 1979).

Loss of hemlock foliage due to HWA infestation creates openings in the canopy that allow more sunlight to penetrate to the forest floor (Phillips and Shure 1990). Increased sunlight increases temperatures in the understory and subsequently dries the leaf litter (Ash 1995), resulting in a less suitable microhabitat for salamanders (Heatwole and Lim 1961; Hutchinson 1961; Brattstrom 1963; Jaeger 1980*a*; Feder and Lynch 1982; Feder 1983; Feder and Londos 1984; Dupuis and Bunnell 1999). For example, numerous studies have shown that canopy opening due to timber harvesting has at least a short-term detrimental effect on salamander populations as a result of a reduction in litter depth and moisture (Petranka et al. 1994; Dupuis et al. 1995; Ash 1988, 1995, 1997; Herbeck and Larsen 1999; Butts and McComb 2000; Grialou et al. 2000; Brooks 2001; Kizlinski et al. 2002; Hicks and Pearson 2003; Knapp et al. 2003). The objective of this study was to examine whether similar effects on salamander populations may be observed due to changes in local areas affected by the HWA.

Site Description

Three study sites were established in mixed hemlock-hardwood forests of the Highlands Plateau region in September 2004. Sites were selected according to the classification parameters in Table 1 (Whitley 2002), as being "intact" (85 – 100% needle retention), "intermediate" (60 – 84%), or "in decline" (0 – 59%).

Site Description	Classification	Condition	Range of Needle Retention
Intact	Healthy	no noticeable needle loss	95-100%
	Good	tree is in overall good condition, but does display slight needle loss	85-94%
Intermediate	OK	tree is in moderate condition and displays slight needle loss	75-84%
	Loss	tree is in moderate condition and displays obvious needle loss	60-74%
Decline	Sub-loss	tree is in poor condition, approximately 50% of branches with needles	30-59%
	Decline	tree is in poor condition, less than 30% of branches with needles	<30%

Table 1: Evaluation of hemlock needle loss (after Whitley 2002).

The intact site (Figure 1) is located along Turtle Pond Road off US 64 S (N 35.076º, W 83.261º) at an elevation of approximately 1000 m. This site has a southeasterly orientation and an average slope of 30°. Tree stand age is approximately 75 years (USDA Forest Service, pers. comm.) and consists primarily of eastern hemlock (*Tsuga canadensis* (L.) Carr.), red oak (*Quercus rubra* L.), white oak (*Q. alba* L.), white pine (*Pinus strobus* L.), mockernut hickory (*Carya tomentosa* (Poiret) Nuttall), and tulip poplar (*Liriodendron tulipifera* L.). Understory species were comprised mainly of dog hobble (*Leucothoe axillaris* (Lam.) D. Don), rose bay rhododendron (*Rhododendron maximum* L.), Christmas fern (*Polystichum acrostichoides* (Michaux) Schott), and fancy fern (*Dryopteris intermedia* (Willd.) Gray).

The intermediate site (Figure 2) is located along Walkingstick Road off Horse Cove Road (N 35.026º, W 83.160º) at an elevation of approximately 850 m. This site has a northwesterly orientation and an average slope of 21°. Tree stand age is approximately 90 years (USDA Forest Service, pers. comm.) and consists primarily of *L. tulipifera*, *T. canadensis*, black birch (*Betula* *lenta* L.), red maple (*Acer rubrum* L.), shagbark hickory (*C. ovata* (Miller) K. Koch), *P. strobus*, black gum (*Nyssa sylvatica* Marshall), and blackjack oak (*Q. marilandica* Muenchh.). Understory species were comprised mainly of *L. axillaris*, New York fern (*Thelypteris noveboracensis* (L.) Nieuwland), rattlesnake plantain (*Goodyera pubescens* (Willd.) R. Brown), mountain laurel (*Kalmia latifolia* L.), American holly (*Ilex opaca* Aiton), *R. maximum*, Solomon's seal (*Polygonatum biflorum* (Walter) Ell.), buckberry (*Gaylussacia ursina* (M. A. Curtis) T. and G. ex Gray.), and sassafras (*Sassafras albidum* (Nuttall) Nees.).

The decline site (Figure 3) is located near the Rock Wilderness Trail located off Bull Pen Road (N 35.016º, W 83.142º) at an elevation of approximately 850 m. This site has a southeasterly orientation and an average slope of 28°. Tree stand age is approximately 80 years (USDA Forest Service, pers. comm.) and consists primarily of *P. strobus*, *T. canadensis*, *L. tulipifera*, *Q. alba*, and *A. rubrum*. Understory species were comprised mainly of *I. opaca*, *R. maximum*, *G. ursina*, Indian cucumber-root (*Medeola virginiana* L.), and galax (*Galax aphylla* L.).

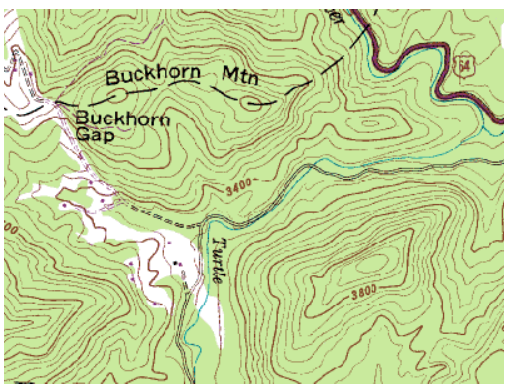

Figure 1: Turtle Pond Road/ Intact Site
http://www.topozone.com/map.asp?z=17&n=3877771&e=302680&size=s&datum=nad83&layer=DRG25

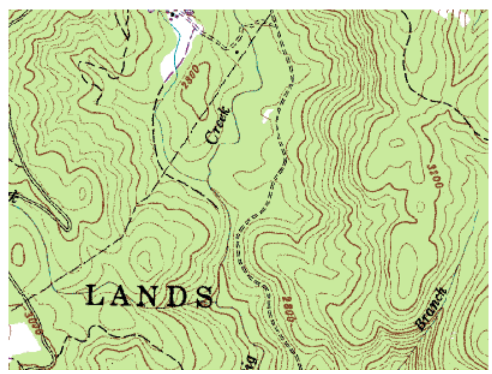

Figure 2: Walkingstick/ Intermediate Site
http://www.topozone.com/map.asp?z=17&n=3883525&e=293505&size=s&datum=nad83&layer=DRG25

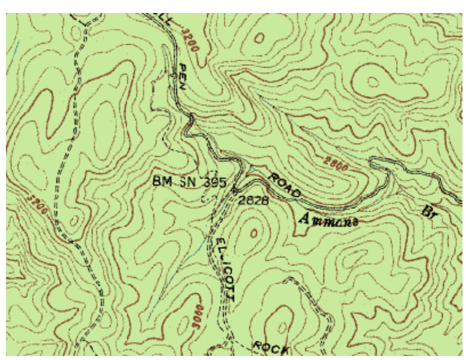

Figure 3: Wilderness Area/ Decline Site
http://www.topozone.com/map.asp?z=17&n=3877044&e=304619&size=s&datum=nad83&layer=DRG25

Materials and Methods

Plot Design

Three 100-m line-transects were established within each site oriented perpendicular to contour lines. Transects were separated from one another by a minimum of 50 m. Each transect was divided into 10-m intervals and marked with flagging tape. Refer to Figure 4 to see how transects were set up at sites.

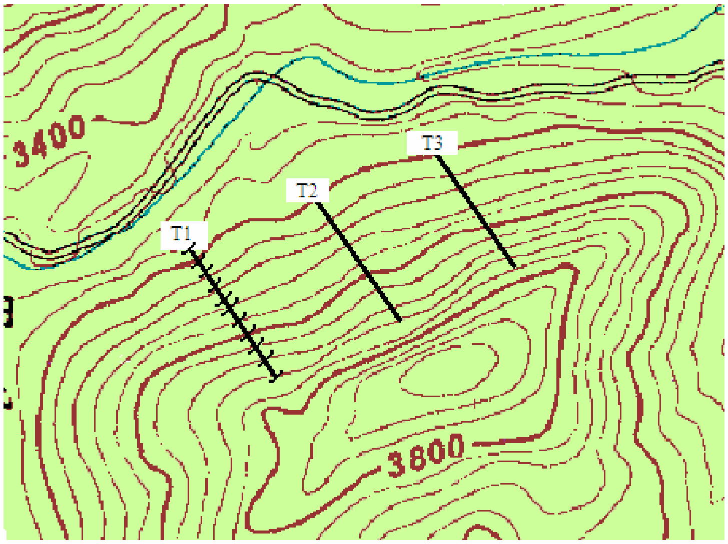

Figure 4: An example of the 100-m line-transect configuration used at each site. Transects were divided into 10-m intervals as depicted in Transect 1 (T1).

Habitat Variables

Along each transect, the diameter and length of coarse woody debris (CWD) was measured to obtain the total volume of CWD. All CWD ≥ 7 cm in diameter that was located within 5 m of either side of the transect was measured. Decay class was ranked from class 1 CWD for recently fallen logs with little decay, to class 5 for extremely decomposed logs (Table 2; Maser et al. 1979).

Decomposition Class	**State of Log**
CWD 1	Tree recently fallen, little decay, support points intact
CWD 2	Tree on ground, little decay, support points absent
CWD 3	Tree on ground, texture hard, bark absent
CWD 4	Tree on ground, texture soft
CWD 5	Tree flattened, texture extremely soft, partially buried by soil

Table 2: Classification of coarse woody debris (after Maser et al. 1979).

Logs in decay classes 1 and 2 were considered to be unusable as salamander microhabitats because they have not undergone significant decay and are generally suspended above the forest floor (Petranka et al. 1994).

A 25 cm^2 sample of leaf litter, defined as all leaves and sticks < 1 cm in diameter down to the O horizon of the soil, was taken from a randomly selected point along each transect for each salamander survey. Moisture content of each sample was determined by obtaining wet mass, drying at 80°C for 24 hours, and then re-weighing the sample to obtain dry mass. Litter moisture content was calculated as wet mass minus dry mass and expressed as a percentage of wet mass (Ash 1995).

Since ground-level temperatures may be greatly increased in areas with minimal canopy cover (Phillips and Shure 1990), canopy cover was determined using a densitometer at a point for each 10-m interval. Percentage canopy cover per transect was calculated as the number of points with cover divided by the total number of points. Temperatures were recorded for each survey by positioning a maximum-minimum thermometer, oriented with slope, at the center of each site.

Salamander Species Surveys

Nine surveys of salamander species were conducted between October 1 and November 8, 2004. Daytime area-constrained searches of natural cover objects has been shown to generate valid indices of salamander diversity (Smith and Petranka 2000). Because rotten logs serve as a daytime refuge for salamanders (Jaeger 1980*b*), the volume of CWD for each transect was considered to represent the total amount of salamander habitat searched. For each transect, all CWD within 5 m of either side of the transect that were able to be moved were turned to look for salamanders underneath. Logs were returned to their original positions after searching and a minimum of 2 days was allowed between surveys to minimize disturbance and allow the habitat to recover. For each transect, the number of each salamander species was recorded, as well as the decay class of the log under which it was found. Because search effort depended on the number of logs present, values were converted to capture rates of salamanders for CWD at each transect.

Statistical Analysis

Differences in habitat variables and in salamander capture rates between sites were examined using a one-way analysis of variance (ANOVA; Microsoft Corporation 2001). Prior to analysis, all percentage values were arcsine-transformed (Zar 1984). Correlation analyses of habitat variables with each other, and with salamander capture rates, were also performed. Due to the small sample sizes of salamanders per transect, values for correlation analyses were pooled by survey date.

Results

Hemlock needle retention differed significantly between sites ($F_{2,6}$=59.16; p<0.001; Table 3). Mean needle retention (± 1 SE) was highest at the intact site, Turtle Pond Road (0.95 ± 0.02%), lowest at the decline site, Rock Wilderness Trail (0.300 ± 0.005%), and moderate at the intermediate site, Walkingstick Road (0.900 ± 0.006%). Percentage canopy cover, however, was not significantly different between sites ($F_{2,6}$=1.39; p=0.32; Table 3). Mean canopy cover was 0.79 ± 0.06% at Turtle Pond, 0.67 ± 0.03% at Walkingstick, and 0.70 ± 0.06% at Rock.

Total volume of CWD did not differ significantly between sites ($F_{2,6}$=2.91; p=0.13; Table 3). Mean overall volume of CWD at Turtle Pond was 3.0 ± 1.0 m^3, at Walkingstick was 3.7 ± 0.5 m^3, and at Rock was 1.6 ± 0.1 m^3. Volume of logs per decay class was also not significantly different between sites for CWD Class 3 ($F_{2,6}$=0.40; p=0.69; Table 3), CWD Class 4 ($F_{2,6}$=1.17; p=0.37), or CWD Class 5 ($F_{2,6}$=1.50; p=0.30). Mean log volume in CWD Class 3 was 0.5 ± 0.3 m^3 at Turtle Pond, 0.37 ± 0.09 m^3 at Walkingstick, and 0.28 ± 0.05 m^3 at Rock. Mean log volume in CWD Class 4 was 0.81 ± 0.04 m^3 at Turtle Pond, 0.7 ± 0.2 m^3 at Walkingstick, and 0.5 ± 0.2 m^3 at Rock. Mean log volume in CWD Class 5 was 2 ± 1 m^3 at Turtle Pond, 2.6 ± 0.7 m^3 at Walkingstick, and 0.9 ± 0.2 m^3 at Rock.

Leaf litter depths between sites were significantly different ($F_{2,195}$=8.91; p<0.001; Table 3). Mean leaf litter depth was highest (3.7 ± 0.2 cm) at Ellicott Rock, lowest (2.8 ± 0.2 cm) at Walkingstick, and intermediate (3.5 ± 0.2 cm) at Turtle Pond. Percentage leaf litter moisture, however, was not significantly different between sites ($F_{2,78}$=2.06; p=0.13; Table 3). Mean litter moisture was 0.50 ± 0.02% at both Turtle Pond and Rock, and 0.44 ± 0.03% at Walkingstick.

The maximum temperature values between sites were significantly different ($F_{2,24}$=57.84; p<0.001; Table 3). Mean maximum temperature was highest (36.0 ± 0.9ºC) at Rock, lowest (23.0 ± 1.0ºC) at Walkingstick, and moderate (32.8 ± 0.5ºC) at Turtle Pond.

Habitat Variables	***df***	***F***	**P**
Percent Needle Retention	2, 6	59.16	<0.001*
Percent Canopy Cover	2, 6	1.39	0.32
Total Volume of CWD	2, 6	2.91	0.13
Volume of Class 3 CWD	2, 6	0.40	0.69
Volume of Class 4 CWD	2, 6	1.17	0.37
Volume of Class 5 CWD	2, 6	1.50	0.30
Leaf Litter Depth	2, 195	8.91	<0.001*
Leaf Litter Moisture	2, 78	2.06	0.13
Maximum Temperature	2, 24	57.84	<0.001*
Salamander Capture Rate (all CWD)	2, 78	3.56	<0.05*
Salamander Capture Rate (CWD Class 3)	2, 78	0.77	0.47
Salamander Capture Rate (CWD Class 4)	2, 78	1.31	0.28
Salamander Capture Rate (CWD Class 5)	2, 78	1.99	0.14

Table 3: Results of ANOVA for between-site variables; *p≤0.05

The intact site, Turtle Pond, yielded the greatest number of salamanders (n=44). Surprisingly, slightly more salamanders were captured at the decline site, Rock (n=28), than at the intermediate site, Walkingstick (n=24). This is due in part to the relatively greater amounts of the southern red-backed salamander (*Plethodon serratus* Grobman), which are more tolerant of xeric conditions (Spotila 1972), found at Rock (Table 4). In addition, Walkingstick had the fewest number of salamander species. The gray-cheeked salamander, *Plethodon metcalfi* Brimley, was the most abundant salamander across all three sites (n=47). Numbers of captures for individual salamander species for each site are summarized in Table 4.

Salamander Species		Intact Site: Turtle Pond	Intermediate: Walkingstick	Decline Site: Ellicott Rock	Total
Southern Gray-Cheeked Salamander	*Plethodon metcalfi* Brimley	22	13	12	47
Blue Ridge Two-Lined Salamander	*Eurycea wilderae* Dunn	7	10	4	21
Southern Red-Backed Salamander	*P. serratus* Grobman	8	1	6	15
Ocoee Salamander	*Desmognathus ocoee* Nicholls	5	0	3	8
Southern Appalachian Salamander	*P. teyahalee* Hairston	2	0	1	3
Red-Spotted Newt	*Notophthalmus viridescens* Baird	0	0	2	2
Total Captures for Site		**44**	**24**	**28**	**96**

Table 4: Total salamander capture by species per site.

The rate of salamander capture overall was significantly different between sites ($F_{2,78}$=3.56; p<0.05; Table 3). Mean overall capture rate was highest (0.7 ± 0.1 salamanders / m^3) at Rock, lowest (0.27 ± 0.06 salamanders / m^3) at Walkingstick, and intermediate (0.6 ± 0.1 salamanders / m^3) at Turtle Pond. However, the salamander capture rate for individual decay classes (Table 3), CWD Class 3 ($F_{2,78}$=0.77; p=0.47), CWD Class 4 ($F_{2,78}$=1.31; p=0.28), and CWD Class 5 ($F_{2,78}$=1.99; p=0.14), did not differ significantly between sites. Mean salamander capture rate for CWD Class 3 logs was 2.0 ± 2.0 salamanders / m^3 at Turtle Pond, 0.2 ± 0.1 salamanders / m^3 at Walkingstick, and 0.9 ± 0.3 salamanders / m^3 at Ellicott Rock. Mean salamander capture rate for CWD Class 4 logs was 0.8 ± 0.2 salamanders / m^3 at Turtle Pond, 0.4 ± 0.2 salamanders / m^3 at Walkingstick, and 1.2 ± 0.6 salamanders / m^3 at Ellicott Rock. Mean salamander capture rate for CWD Class 5 logs was 0.7 ± 0.2 salamanders / m^3 at Turtle Pond and Ellicott Rock, and 0.28 ± 0.08 salamanders / m^3 at Walkingstick.

Salamander abundance (n=96) was significantly positively correlated with litter depth (r=0.725), maximum temperature (r=0.649), and percentage litter moisture (r=0.692; Table 5), but salamanders showed no significant correlation with total CWD, Class 3 CWD, Class 4 CWD, Class 5 CWD, percentage canopy cover, or needle retention. Correlation coefficients between habitat variables are given in Table 6.

Habitat Variable	*R*
Needle Retention	-0.482
Percentage Canopy Cover	0.256
Total CWD Volume	-0.005
CWD 3 Volume	0.443
CWD 4 Volume	0.157
CWD 5 Volume	-0.140
Leaf Litter Depth	0.725*
Percentage Leaf Litter Moisture	0.692*
Maximum Temperature	0.649*

Table 5: Correlation coefficients (*r*) for salamander capture rate with habitat variables; * $p \leq 0.05$

	Needle Retention	Canopy Cover	Litter Depth	Litter Moisture	Maximum Temperature	Total CWD Volume
Needle Retention		0.147	-0.593*			0.763*
Canopy Cover			0.307			-0.081
Litter Moisture	-0.342	0.184	0.825*		0.617*	-0.488
Maximum Temperature	-0.673*	0.276	0.824*			-0.575*
Total CWD Volume			-0.603*			
CWD 3 Volume	0.223	0.702*	-0.135	-0.441	-0.046	0.176
CWD 4 Volume	0.625*	0.013	-0.464	-0.254	-0.258	0.218
CWD 5 Volume	0.601*	-0.249	-0.488	-0.347	-0.524*	0.952*

Table 6: Correlation coefficients (*r*) for habitat variables; *$p \leq 0.05$

Discussion

With the exception of the red-spotted newt, salamanders in this study belong to the family Plethodontidae, the largest of ten salamander families (Petranka 1998). Several unique characteristics of terrestrial plethodontid salamanders allow them to act as accurate indicators of the state of forest ecosystems (Welsh and Droege 2001). Defining characteristics of these salamanders are suppression of the aquatic larval stage, eggs deposited in terrestrial nests, and young salamanders resembling miniature adults (Wake and Hanken 1996). Believed to have their origin in the Appalachian streams, salamanders of this family were able to colonize mesic forests of North America quickly due to their independence of water for breeding purposes (Wake 1966). Where conditions are suitable, plethodontid salamander densities can be very high (Hairston 1987).

Lungless plethodontid salamanders exchange gases through their skin. Cutaneous respiration, coupled with a high surface area to volume ratio, puts them at high risk of desiccation. To avoid dehydration during respiration, lungless salamanders must continuously maintain wet skin and are therefore restricted to moist microhabitats (Spotila 1972; Feder 1983; Petranka 1998). Moisture in the leaf litter and upper soil layers is critical to salamanders, because a high relative humidity is necessary to maintain surface activity for foraging and mating purposes (Houck and Verrell 1993; Tilley and Bernardo 1993). Because moisture is also important for larval development, terrestrial salamanders often lay their eggs in moist, cryptic sites (Petranka 1998). Welsh and Droege (2001) reported that deep leaf litter, as opposed to the soil layer, may be the most critical habitat for salamanders. If the litter becomes too thin, patchy, or dry, then large populations of salamanders cannot exist (Ash 1995, 1997).

Water balance in salamanders is greatly affected by temperature (Hutchinson 1961). At higher temperatures, cutaneous moisture will evaporate more quickly, leaving lungless salamanders vulnerable to desiccation. Consequently, salamanders prefer cooler habitats (Hutchinson 1961; Spotila 1972), such as those provided by dense forest canopies (Chen et al. 1999). Moreover, in warmer microclimates leaf litter moisture will evaporate more quickly, which is a critical factor for salamander microhabitats because salamanders require wet skin for respiration (Chen et al. 1999).

A tall, stratified forest canopy moderates extreme weather conditions that might otherwise drastically alter microclimatic conditions (Chen et al. 1999). Salamanders are more likely to occur in forests with abundant rocky substrates, downed wood, or leaf litter (Bury et al. 1991; Corn and Bury 1991). Old growth forests are often more suitable habitats for salamanders because they provide greater amounts of CWD and moisture-retaining leaf litter. Older forests generally have a denser understory that not only provides shade, but also perch sites for hunting and refuge from predators (Petranka et al. 1994; Dupuis and Bunnell 1999). Welsh and Lind (1991) found that sites lacking rock cover, ground-cover level vegetation, or CWD in moderate decay classes (classes 3 and 4) had much lower salamander diversity.

Leaf litter in the Southern Appalachians harbors an abundance of invertebrate prey for salamanders (Gist and Crossley 1975). Plethodontid salamanders have the lowest metabolism known to vertebrates and can go for long periods of time without food (Feder 1976). Salamanders are not limited by prey abundance, but rather by accessibility of prey due to their own moisture requirements. Foraging for food is limited to the amount of time the salamander can go without desiccating (Fraser 1976; Jaeger 1980*a*). During the day, salamanders remain in subsurface retreats, such as those provided by CWD. At night, however, when the relative humidity is higher, salamanders are able to leave these retreats to search for food (Jaeger 1972; Petranka et al. 1994).

Because CWD, especially in the latter stages of decay, holds water; salamanders often use logs as refuges when moisture in the leaf litter is insufficient (Jaeger 1972; Petranka et al. 1994). Under periodic dry conditions, salamanders will remain in these retreats and leave only for limited periods of time to forage for food (Fraser 1976; Jaeger 1980*b*). If warm, dry conditions persist, salamander surface activity is restricted, and they may be forced underground (Taub 1961; Spotila 1972).

Openings in the forest canopy can have drastic effects on the microclimate and moisture regimes of the forest floor. Canopy gaps allow increased wind effects, greater solar radiation penetrating through to the forest floor, and temperature extremes that can reduce the moisture level of the leaf litter (Chen et al. 1999). This, in turn, can affect salamander populations that depend on those microhabitats (Harpole and Haas 1999). Large canopy gaps, such as those produced by clear-cutting, create different microhabitat conditions than those found in smaller canopy gaps produced by selective logging or natural blowdowns (Chen et al. 1999).

Within the Southern Appalachians, large canopy openings resulting from clear-cutting cause a reduction in leaf litter depth, dry mass, and moisture as well as amounts of CWD (Ash 1995, 1997). Declines in litter depth can cause a decrease in associated invertebrate prey (Gist and Crossley 1975). Because leaf litter parameters are critical for salamanders, loss of microhabitat combined with decreases in prey produce a drastic decline in salamander populations (Ash 1988, 1995; Petranka et al. 1993, 1994). Ash (1997) demonstrated that salamander diversity is reduced by nearly 100% within two years of clear-cutting, and requires decades to fully recover.

Impacts on salamander microhabitats appear to be much less severe in small canopy gaps. Phillips and Shure (1990) reported that smaller canopy gaps created by natural and human disturbance did not significantly change litter dynamics and species composition in comparison with the surrounding forest. This is because allochthonous leaf fall, or litter originating from adjacent areas, compensates for loss of autochthonous leaf fall, or within site litter, in small canopy gaps (Shure and Phillips 1987). Minimal changes in the microclimate created by small canopy gaps (Runkle 1981; Clinton 2003) are expected to have only minimal effects on salamander populations. Studies of selective logging act as a good simulation of the effects of small canopy gaps. Selective logging has not been found to severely affect salamander populations (Messere and Ducey 1998; Knapp et al. 2003).

Moisture and temperature can also vary widely between neighboring slopes (Whittaker 1956). Because intensity and duration of sunlight exposure is reduced, northern slopes (in the Northern Hemisphere) are generally cooler and wetter than south-facing slopes (Wales 1972). Brannon (2002) found that the percentage of heavily decayed logs was greater on north-facing slopes than on south-facing slopes, because logs decay faster in mesic conditions (Abbott and Crossley 1982). Consequently salamanders, which require moist cool conditions and decaying debris, are often more prevalent on north-facing slopes than on south-facing slopes.

In mixed hemlock-hardwood stands like those at my study sites, canopy loss due to the HWA appears to more closely resemble that of the small gaps created by selective cutting. This is because many other trees composed the forest canopy at each site. Consequently, loss of canopy in HWA-affected hemlocks would not result in the large canopy gaps that would be expected if the canopy were exclusively hemlock.

Although there was a significant difference in needle loss at my sites, overall canopy cover remained consistent. Despite their prevalence in the forests, hemlocks apparently contribute only a small portion to the entire forest canopy. Therefore, canopy opening due to needle loss was minimal.

Litter depth was also found to be significantly different between sites, which corresponded with defoliation. Leaf litter was deepest where the hemlocks were most defoliated, because the lost needles contributed additional material to the litter layer. Litter depths were greatest at the decline site, Rock Wilderness Trail, because the hemlocks there had lost a significant proportion of their needles. This added layer of needles could indirectly affect litter moisture by buffering the drying that naturally occurs from increased sunlight penetration on the forest floor, allowing the lower layers of litter to retain their moisture content (Ash 1995).

By comparison, the intermediate site, Walkingstick Road, had the lowest litter depths and yet the moisture was not significantly different from that of Rock or the intact site, Turtle Pond Road. Walkingstick occurred on a northwesterly facing slope, and such slopes are generally more mesic. On the other hand, Turtle Pond and Rock were on southeasterly facing slopes, which are generally more xeric (Wales 1972). Although leaf litter depths at Walkingstick were reduced, the orientation of the site may have helped to mediate moisture levels.

Likewise, maximum temperatures were found to be significantly higher at Rock than at the other sites. This is due to needle loss in hemlocks, coupled with the southeasterly orientation of the site. The size of canopy gaps created by the defoliation of hemlocks is related to the proportion of hemlocks in the canopy. Needle loss allows sunlight to penetrate the canopy and reach the forest floor, making conditions there warmer and consequently drier (Ash 1995). Although Walkingstick is intermediate in terms of needle loss, maximum temperature values were lowest at this site. This again may be a function of the north-facing orientation of the slope and its associated effects on microclimate. Canopy cover at this site also aids in buffering the temperature.

Populations of salamanders are often negatively affected by higher temperatures and reduced moisture. In this study, however, salamander capture rates did not appear to be reduced in areas with greater needle loss, as the overall canopy cover remained sufficient to maintain litter moisture. Deeper leaf litter, contributed to by the defoliation of hemlocks, provides a greater source of moisture, invertebrate prey, and protective cover from predators. Environmental moisture, and the microhabitat features that maintain it, may be of greater importance to salamanders than temperature. Thus, it can be concluded that ground cover was more important than canopy cover at our sites, because the ground cover directly helps to maintain the moisture critical to salamander microhabitats.

My findings indicate that if ground cover is sufficient, there should be minimal detrimental effects of HWA on terrestrial salamanders in mixed hemlock-hardwood forests. In areas where hemlocks dominate the canopy, however, the results may be dramatically different. For instance, in these areas, needle loss could produce large canopy gaps which have much more far-reaching effects on salamander microhabitat (Ash 1988, 1995; Petranka et al. 1993, 1994). For example, Brooks (2001) found that removal of overstory hemlocks from hemlock-dominated forests resulted in as much as an 83% decline in the population of redback salamanders in New England, although the response was ephemeral.

Because time constraints prevented my obtaining large sample sizes, my findings should be viewed cautiously. Future surveys should entail a longer sampling period and include sites that are even more strongly affected by HWA, possibly to the point that the standing hemlocks are dead. Another suggestion for monitoring HWA-induced changes in salamander microhabitat is to include greater numbers of replicates to better account for differences in site orientation. Because hemlocks at my sites were concentrated along streams, a final recommendation is to also examine the effects of defoliation on aquatic species of salamanders of the genus *Desmognathus*, which depend on streams for egg deposition and spend a large portion of their lives as aquatic larvae (Petranka 1998). In addition, hemlock loss reduces the abundance of invertebrate prey in streams (Snyder et al. 2002). Thus, effects of HWA-induced microhabitat change may be more pronounced in aquatic salamanders than in terrestrial salamanders. More inclusive studies such as these will help elucidate the effect of the HWA on Southern Appalachian salamander populations.

References Cited

Abbott, D. T. and D. A. Crossley. 1982. Woody litter decomposition following clear-cutting. Ecology 63: 35-42.

Ash, A. N. 1988. Disappearance of salamanders from clearcut plots. Journal of the Elisha Mitchell Scientific Society 104: 116-122.

Ash, A. N. 1995. Effects of clear-cutting on litter parameters in the southern Blue Ridge Mountains. Castanea 60: 89-97.

Ash, A. N. 1997. Disappearance and return of Plethodontid salamanders to clearcut plots in the southern Blue Ridge Mountains. Conservation Biology 11: 983-989.

Brannon, M. P. 2002. Distributions of *Sorex cinereus* and *S. fumeus* on north- and south-facing slopes in the southern Appalachian Mountains. Southeastern Naturalist 1: 299-306.

Brattstrom, B. H. 1963. A preliminary review of the thermal requirements of amphibians. Ecology 44: 238-255.

Brodie, Jr., E. D. and R. R. Howard. 1973. Experimental study of Batesian mimicry in the salamanders *Plethodon jordani* and *Desmognathus ochrophaeus*. American Midland Naturalist 90: 38-46.

Brodie, Jr., E. D., R. T. Nowak and W. R. Harvey. 1979. The effects of antipredator secretions and behavior of selected salamanders against shrews. Copeia 1979: 270-274.

Brooks, R. T. 2001. Effects of the removal of overstory hemlock from hemlock-dominated forests on eastern redback salamanders. Forest Ecology and Management 149:197-204.

Brown, C. L. and L. K. Kirkman. 1990. Trees of Georgia and Adjacent States. Timber Press: Portland, Oregon.

Bruck, R. and W. P. Robarge. 1988. Change in forest structure in the boreal montane ecosystem of Mount Mitchell, North Carolina, USA. European Journal of Forest Pathology 18: 357-366.

Bury, R. B., P. S. Corn, K. B. Aubry, F. F. Gilbert and L. C. Jones. 1991. Regional patterns of terrestrial amphibian communities in Washington and Oregon. Pp. 353-362 *In: In:* L. F. Ruggiero, K. B. Aubry, A. B. Carey and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. General technical report PNW-GTR-285. US Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Butts, S. R. and W. C. McComb. 2000. Associations of forest-floor vertebrates with coarse woody debris in managed forests of western Oregon. Journal of Wildlife Management 64: 95-104.

Chen, J., S. C. Saunders, T. R. Crow, R. J. Naiman, K. D. Brosofske, G. D. Morz, B. L. Brookshire and J. F. Franklin. 1999. Microclimate in forest ecosystem and landscape ecology. BioScience 49: 288-297.

Clinton, B. D. 2003. Light, temperature, and soil moisture responses to elevation, evergreen understory, and small canopy gaps in the southern Appalachians. Forest Ecology and Management 186: 243-255.

Cobb, R. C. and D. A. Orwig. 2002. Impacts of hemlock woolly adelgid infestation on decomposition: an overview. Pp 317-322, *In:* Proceedings: Hemlock Woolly Adelgid in the Eastern United States Symposium. Feb. 5-7, 2002, East Brunswick, NJ, USDA For. Serv./State Univ. of N.J. Rutgers, Onken, B., R. Reardon, and J. Lashomb (eds.).

Corn, P. S. and R. B. Bury. 1991. Terrestrial amphibian communities in the Oregon Coast Range. Pp. 305-318 *In:* L. F. Ruggiero, K. B. Aubry, A. B. Carey and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. General technical report PNW-GTR-285. US Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Cullina, W. 2002. Native Trees, Shrubs, and Vines: a Guide to Using, Growing, and Propagating North American Woody Plants. Houghton Mifflin Company: Boston.

Davis, D. E. 2000. Where There are Mountains: an Environmental History of the Southern Appalachians. University of Georgia Press: Athens, Georgia.

Dupuis, L. A. and F. L. Bunnell. 1999. Effects of stand age, size, and juxtaposition on abundance of western redback salamanders (*Plethodon vehiculum*) in coastal British Columbia. Northwest Science 73:27-33.

Dupuis, L. A., J. N. M. Smith and F. Bunnell. 1995. Relation of terrestrial-breeding amphibian abundance to tree-stand age. Conservation Biology 9:645-653.

Evans, R. A., 2002. An ecosystem unraveling. Pp 23-33, *In:* Proceedings: Hemlock Woolly Adelgid in the Eastern United States Symposium. East Brunswick, NJ, USDA For. Serv./State Univ. of N. J. Rutgers, Onken, B., R. Reardon, and J. Lashomb (eds.).

Evans, R. A., E. Johnson, J. Shreiner, A. Ambler, J. Battles, N. Cleavitt, T. Fahey, J. Sciascia and E. Pehek, 1996. Potential impacts of hemlock wooly adelgid (*Adelges tsugae*) on eastern hemlock (*Tsuga canadensis*) ecosystems. Pp 42-57, *In:* Proceedings of the First Hemlock Wooly Adelgid Review, Charlottesville, Va. Oct. 12, 1995, USDA For. Serv. FHTET 96-10, Morganton, WVa., Salom, S.M., T. C. Tigner, and R. C. Reardon (eds.).

Feder, M. E. 1976. Lunglessness, body size, and metabolic rate in salamanders. Physiological Zoology 49: 398-406.

Feder, M. E. 1983. Integrating the ecology and physiology of plethodontid salamanders. Herpetologica 39: 291-310.

Feder, M. E. and P. L. Londos. 1984. Hydric constraints upon foraging in a terrestrial salamander, *Desmognathus ochrophaeus* (Amphibia: Plethodontidae). Oecologia 64: 413-418.

Feder, M. E. and J. F. Lynch. 1982. Effects of latitude, season, elevation, and microhabitat on field body temperatures of neotropical and temperate zone salamanders. Ecology 63: 1657-1664.

Fraser, D. F. 1976. Empirical evaluation of the hypothesis of food competition in salamanders of the genus *Plethodon*. Ecology 57: 459-471.

Gist, C. S. and D. A. Crossley, Jr. 1975. The litter arthropod community in a southern Appalachian hardwood forest: Numbers, biomass and mineral element content. American Midland Naturalist 93: 107-122.

Grialou, J. A., S. D. West and R. N. Wilkins. 2000. The effects of forest clearcut harvesting thinning on terrestrial salamanders. Journal of Wildlife Management 64: 105-113.

Hairston, N. G. 1949. The local distribution and ecology of the plethodontid salamanders of the southern Appalachians. Ecological Monographs 19: 49-73.

Hairston, N. G. 1987. Community Ecology and Salamander Guilds. Cambridge University Press, Cambridge, United Kingdom.

Harpole, D. N. and C. A. Haas. 1999. Effects of seven silvicultural treatments on terrestrial salamanders. Forest Ecology and Management 114: 349-356.

Heatwole, H. and K. Lim. 1961. Relation of substrate moisture to absorption and loss of water by the salamander *Plethodon cinereus*. Ecology 42: 814-819.

Herbeck, L. A., D. R. Larsen. 1999. Plethodontoid salamander response to silvicultural practices in Missouri Ozark forests. Conservation Biology 13: 623-632.

Hicks, N. G. and S. M. Pearson. 2003. Salamander diversity and abundance in forests with alternative land use histories in the southern Blue Ridge mountains. Forest Ecology and Management 177:117-130.

Houck, L. and P. Verrell. 1993. Studies of courtship behavior in plethodontid salamanders: a review. Herpetologica 49: 175-184.

Hutchinson, V. H. 1961. Critical thermal maxima in salamanders. Physiological Zoology 34: 92-125.

Jaeger, R. G. 1972. Food as a limited resource in competition between two species of terrestrial salamanders. Ecology 53: 535-546.

Jaeger, R. G. 1980*a*. Fluctuations in prey availability and food limitation for a terrestrial salamander. Oecologia 44: 335-341.

Jaeger, R. G. 1980*b*. Microhabitats of a terrestrial forest salamander. Copeia 1980: 265-268.

Jenkins, M. A. 2003. Impact of the balsam woolly adelgid (*Adelges piceae* Ratz.) on an *Abies fraseri* (Pursh) Poir. dominated stand near the summit of Mount LeConte, Tennessee. Castanea 68:109-118.

Kizlinski, M. L., D. A. Orwig, R. C. Cobb, D. R. Foster. 2002. Direct and indirect ecosystem consequences of an invasive pest on forests dominated by eastern hemlock. Journal of Biogeography 29: 1489-1503.

Knapp, S. M., C. A. Haas, D. N. Harpole and R. L. Kirkpatrick. 2003. Initial effects of clearcutting and alternative silvicultural practices on terrestrial salamander abundance. Conservation Biology 17: 752-762.

Mahan, C. G. 1999. Ecosystem-based assessment of biodiversity associated with eastern hemlock forests. Park Science 19: 37-39.

Maser, C., R. G. Anderson, K. Cromack, J. T. Williams and R. E. Martin. 1979. Dead and down woody material. Pp. 78-95, *In:* J. W. Thomas (Ed.). Wildlife Habitats in Managed Forests: the Blue Mountains of Oregon and Washington. USDA Forest Service Handbook 553. 512 pp.

McClure, M. S. 1990. Role of wind, birds, deer, and humans in the dispersal of hemlock woolly adelgid (Homoptera: Adegidae). Environmental Entomology 19: 36-43.

Messere, M. and P. K. Ducey. 1998. Forest floor distribution of northern redback salamanders, *Plethodon cinereus*, in relation to canopy gaps: First year following selective logging. Forest Ecology and Management 107: 319-324.

Microsoft Corporation. 2001. Microsoft Excel Version 10.0.6501.

Orwig, D. A. and D. R. Foster. 1998. Forest response to the introduced hemlock woolly adelgid in southern New England, USA. Journal of the Torry Botanical Society 125: 60-73.

Petranka, J. W. 1998. Salamanders of the United States and Canada. Smithsonian Institution Press, Washington.

Petranka, J. M., M. P. Brannon, M. E. Hopey and C. K. Smith. 1994. Effects of timber harvesting on low elevation populations of southern Appalachian salamanders. Forest Ecology and Management 67: 135-147.

Petranka, W. P., M. E. Elderidge and K. E. Haley. 1993. Effects of timber harvesting on southern Appalachian salamanders. Conservation Biology 7: 363-370.

Phillips, D. L. and D. J. Shure. 1990. Patch-size effects on early succession in southern Appalachian forests. Ecology 71: 204-212.

Pough, H. F., E. M. Smith, D. H. Rhodes and A. Collazo. 1987. The abundance of salamanders in forest stands with different histories of disturbance. Forest Ecology and Management 20: 1-9.

Rabenold, K. N., P. T. Fauth, W. G. Bradley, J. A. Sadowski and P. G. Parker. 1998. Response of avian communities to disturbance by an exotic insect in spruce-fur forests of the southern Appalachians. Conservation Biology 12: 177-189.

Ross, R. M., R. M. Bennet, C. D. Snyder, J. A. Young, D. R. Smith and D. P. Lemarie. 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 12: 60-65.

Ross, R. M., L. A. Redell and R. M. Bennett. 2004. Mesohabitat use of threatened hemlock forests by breeding birds of the Delaware River Basin in northeastern United States. Natural Areas Journal 24: 307-315.

Runkle, J. R. 1981. Gap regeneration in some old-growth forests of eastern North America. Ecology 62: 1041-1051.

Shure, D. J. and D. L. Phillips. 1987. Litter fall patterns within different-sized disturbance patches in a Southern Appalachian Mountain Forest USA. American Midland Naturalist 118: 348-357.

Smith, C. K. and J. W. Petranka. 2000. Monitoring terrestrial salamanders: Repeatability and validity of area-constrained cover object searches. Journal of Herpetology 34: 547-557.

Smith, G. F. and N. S. Nicholas. 1998. Patterns of overstory competition in the fir and fir-spruce forests of the Great Smokey Mountains after balsam woolly adelgid infestation. American Midland Naturalist 139: 340-352.

Snyder, C. D., J. A. Young, D. P. Lemarie and D. R. Smith. 2002. Influence of eastern hemlock (*Tsuga canadensis*) forests on aquatic invertebrate assemblages in headwater streams. Canadian Journal of Fisheries and Aquatic Sciences 59: 262-275.

Spotila, J. R. 1972. Role of temperature and water in the ecology of lungless salamanders. Ecological Monographs 42: 95-125.

Taub, F. B. 1961. The distribution of the red-backed salamander *Plethodon c. cinereus*, within the soil. Ecology 42: 681-698.

Tilley, S. G. and J. Bernardo. 1993. Life history evolution in plethodontid salamanders. Herpetologica 49: 154-163.

Tingley, M. W., D. A. Orwig, R. Field and G. Motzkin. 2002. Avian response to removal of a forest dominant: consequences of hemlock woolly adelgid infestations. Journal of Biogeography 29: 1505-1516.

Wake, D. B. 1966. Comparative osteology and evolution of the lungless salamanders, family Plethodontidae. Memoirs of the Southern California Academy of Science 4: 1-111.

Wake, D. B. and J. Hanken. 1996. Direct development in the lungless salamanders: what are the consequences for developmental biology, evolution and phylogenesis? International Journal of Developmental Biology 40: 859-869.

Wales, B. A. 1972. Vegetation analysis of north and south edges in a mature oak-hickory forest. Ecological Monographs 42: 451-471.

Watson, J. K., G. M. Hunt and J. R. Rhea. 1994. Forest health evaluation of the hemlock woolly adelgid, *Adelges tsugae* Annand, infestations in Shenandoah National Park, Virginia. Progress Report. USDI, National Park Service.

Welsh Jr., H. H. and S. Droege. 2001. A case for using plethodontid salamanders for monitoring biodiversity and ecosystem integrity of North American forests. Conservation Biology 15: 558-569.

Welsh, H. H., Jr., and A. J. Lind. 1991. The structure of the herpetofaunal assemblage of the Douglas-fir forests of northwestern California and southwestern Oregon. Pp. 394-413. *In:* L. F. Ruggiero, K. B. Aubry, A. B. Carey and M. H. Huff, technical coordinators. Wildlife and vegetation of unmanaged Douglas-fir forests. General technical report PNW-GTR-285. US Forest Service, Pacific Northwest Research Station, Portland, Oregon.

Whitley, K. 2002. Hemlock woolly adelgid infestations on and around the Highlands Plateau. Pp. 26-44, Land Use and Biodiversity on the Highlands Plateau: A Carolina Environmental Program Report. Highlands Biological Station, Highlands, NC. 99 pp.

Whittaker, R. H. 1956. Vegetation of the Great Smoky Mountains. Ecological Monographs 26: 1-80.

Wydeven, A. P. and R. W. Hay, 1996. Mammals, amphibians and reptiles of hemlock forest in the Lake Superior region. Pp. 115-123, *In:* Hemlock Ecology and Management: Proceedings of a Regional Conference on Ecology and Management of Eastern Hemlock, Sept. 27-28, 1995, Iron Mountain, Michigan. Univ. Wisconsin, Madison, Wisconsin., Mroz, G. and J. Martin (eds.).

Zar, J. H. 1984. Biostatistical Analysis, 2nd ed. Prentice-Hall, Inc., Englewood-Cliffs, NJ. 718 pp.

Riparian zone structure and function in Southern Appalachian forested headwater catchments

Katie Burke

Introduction

The forest environment around streams and other bodies of water has been shown to have high biodiversity (Odum 1978, Gregory et al. 1991, Malanson 1993, Naiman et al. 1993) and to be important in maintaining water quality (Wenger 1999, Lindenmayer and Franklin 2002). These environments, called riparian zones, are defined as "the complex assemblage of organisms and their environment existing adjacent to and near flowing water" (Lowrance et al. 1985). Riparian zones sustain significant amounts of forest biomass – as much as 100 times more than other areas in the forest environment – which highlights the crucial role they play in carbon storage (Lindenmayer and Franklin 2002). Riparian corridors reduce erosion and sedimentation in streams, prevent excessive flooding, regulate nitrogen, control water temperature, contribute allochthonous materials to communities of aquatic organisms downstream, prevent eutrophication of aquatic systems, and provide habitat to a variety of species of flora and fauna (Gregory et al. 1991, Malanson 1993, Wenger 1999).

Because of these important roles that a riparian zone plays in both terrestrial and aquatic ecosystems, it is necessary to conserve and protect them from damage caused by human land use activities. Because riparian zones vary considerably with respect to forest type, slope, and stream size, debates have raged about how much of a riparian zone should be conserved in order to protect ecosystem processes sufficiently. Protected zones, called "stream buffers", range in recommended size from 9 m to 100 m (Wenger 1999, Lindenmayer and Franklin 2002), depending on environmental factors at study locations. For example, studies in western Washington have shown that riparian buffers of at least 45 m were required to sustain natural microclimatic conditions along streams in moderate to steep terrain (Brosofske et al. 1997). In the Vermont Appalachians, however, 15 m buffers were found to support 90% of the plant species surveyed (Spackman and Hughes 1995). Riparian zone width is frequently determined by a few select features, such as soil or vegetation composition, as plant species in surrounding upland areas normally differ from those adapted to more mesic conditions associated with riparian zones (Gregory et al. 1991). In other riparian studies, slope of the terrain has been found to be one of the most significant variables determining buffer width (Swift 1986, Wenger 1999). Slope and the presence of wetlands are the most significant and useful features in deciding on buffer width (Wenger 1999). Other factors that can affect appropriate buffer width include hydrology and soil type.

Another important factor in characterizing a riparian zone and its appropriate stream buffer is the size of the stream. Low-order, headwater streams and their interaction with riparian zones have been little studied. Yet these streams contribute the most allochthonous material (i.e., material that is formed elsewhere than its present location), sustain most of the biodiversity, have the highest aquatic productivity, and serve as filters for water flowing to larger bodies downstream (Forest Ecosystem Management Assessment Team 1993, Naiman and Bilby 1998, Lindenmayer and Franklin 2002). In addition, these streams make up most of the stream miles in a watershed (Osborne and Kovavic 1993, Hubbard and Lowrance 1994, Lowrance et al. 1997, Wenger 1999). Headwater streams have the most interaction between land and water, thus making them the largest source of sediment and organic material (Wenger 1999). Their occurrence on steep slopes is especially important and should be considered when assessing nutrient cycling and sediment run-off in streams. Their relatively high elevation also makes them home to species that may not be found at lower elevations. Because the Highlands Plateau has a large number of such low-order headwater streams, study of the structure and function of riparian zones in this region is crucial to the preservation of regional biodiversity.

There have been few studies in Southern Appalachian watersheds designed to assess appropriate stream buffer widths and also what factors should be evaluated in order to determine an appropriate stream buffer width. Coweeta Hydrologic Laboratory, in the Southern Appalachians, is conducting experiments to assess riparian zone structure and function in headwater watersheds with the following objectives: (1) determining the functional width of riparian zones in headwater ecosystems of the Southern Appalachians from a multiple resource perspective; and (2) testing the effectiveness of current riparian zone buffer standards from a multiple resources perspective. Other research projects associated with this study of leaf litter inputs, stream chemistry, and carbon cycling include vegetation composition and dynamics

(Coweeta Hydrologic Laboratory), sediment and nutrient filters (Coweeta Hydrologic Laboratory), aquatic conditions (Virginia Tech and Coweeta Hydrologic Laboratory), bat community structure and habitat use in riparian zones (Clemson University), and effects of riparian zone width on salamander populations (University of Missouri).

Riparian and stream ecosystems play an important role in nutrient and carbon cycling. Litter fall from riparian vegetation can regulate energy flows, provide a source of organic carbon, and substantially influence physical and chemical conditions in stream ecosystems (Campbell et al. 1992, Thomas et al. 1992, Haycock et al. 1997, Lindenmayer and Franklin 2002). Nitrogen and phosphorus in Southern Appalachian streams have been found to follow a seasonal pattern of winter minima and summer maxima, in contrast to patterns observed in northern streams, perhaps indicating the importance of winter nutrient cycling processes in soils and streams in this warmer region (Mulholland 1992). In a study in eastern Tennessee, the riparian zone was determined to be a source of ammonium and phosphorus to the stream when dissolved oxygen concentrations in the stream were low (Mulholland 1992). Understanding how nutrient concentrations in headwater streams are regulated is crucial to understanding stream productivity (Elwood et al. 1981) in addition to the amount and timing of nutrient inputs to larger water bodies downstream (Jasper et al. 1983, Mulholland 1992).

Experimental disturbances in small watersheds have shown that terrestrial processes have a major effect on stream-water nutrient concentrations (Likens et al. 1977, Swank 1988, Mulholland 1992). Because land-water nutrient cycling in riparian zones is so important, I chose to study the nutrient inputs and sinks in a riparian zone/stream ecosystem, in the hopes of providing more insight into appropriate widths of stream buffers within this larger long-term research project. The objectives of this study were to determine changes in nutrient cycling in the riparian areas around headwater streams by assessing coarse woody debris, leaf litter fall, soil moisture, soil temperature, and soil CO_2 efflux along a gradient from the stream to the uplands. By gaining a better understanding of nutrient inputs, sinks, and cycles, we can understand how much of a riparian zone must be conserved in order to maintain these important processes.

Methods

Site description – This study is being conducted in the Blue Ridge Physiographic Province of western North Carolina. The study sites are in the Ray Branch area (Figure 1) on the Wayah Ranger District of the Nantahala National Forest where elevations range from 850 to 925 m. Stream channel flow direction varies from NW to E facing. Slopes perpendicular to stream channels range from a 17% to 50% grade. All sites support mature stands of mixed hardwoods with a lesser component of eastern white pine (*Pinus strobus*) and eastern hemlock (*Tsuga canadensis*). There is a negligible amount of evergreen understory (*Rhododendron maximum*, *Kalmia latifolia*) on the sites. The streams draining the sites are perennial and generally high-gradient and low volume in nature. Mean annual rainfall for the study area is approximately 1800 mm, with < 5% falling as snow or ice. Mean annual temperature is 12.6°C and ranges from a mean of 6.7°C in the dormant season to 18.5°C during the growing season.

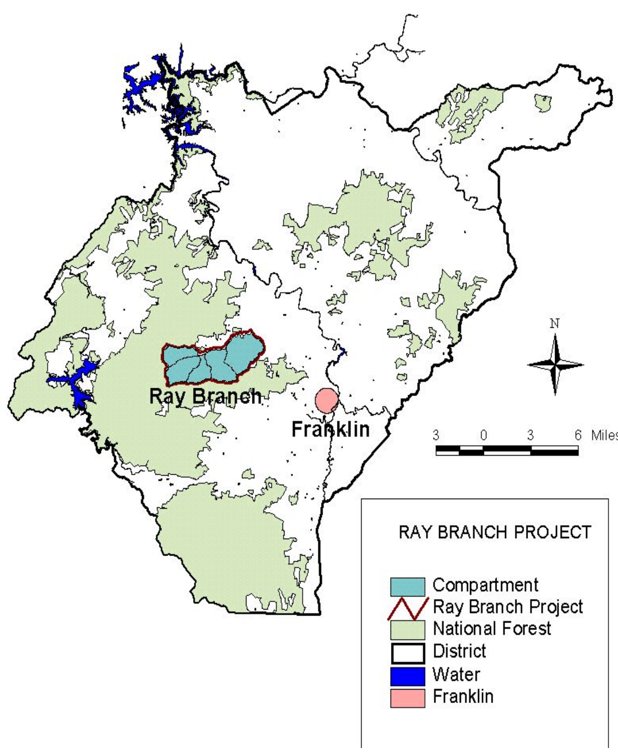

Figure 1: Map of Macon Co., NC, showing study region at Ray Branch.

Study design

Four sites were selected based on their scheduled sivicultural prescriptions and accessibility. In the larger study, three buffer widths are being examined: 0-m, 10-m, and 30-m, plus two sites as an untreated reference (one high elevation and one at a lower elevation). All sites remained uncut throughout this study. On each site, eight 50 m long transects were established up the slope from the stream in a direction perpendicular to the channel. Four transects were randomly selected on each site for measurement of soil CO_2 evolution, and leaf litter and small wood inputs. Baskets were installed at 10 m intervals along each of the four transects beginning at the stream bank and extending 50 m up the slope. Litter collections took place every two weeks and all samples were oven dried at 65°C for 36 hours, sorted into seven unique categories, and weighed to the nearest 0.01 g. The categories included oaks (*Quercus alba, Q. rubra, Q. prinus*), conifers (*Tsuga canadensis, Pinus strobus*) , magnolias (*Liriodendron tulipifera* with minimal *Magnolia fraseri*), red maple (*Acer rubrum*), rhododendron (*Rhododendron maximum*), wood, and other species and unidentifiable material. The "other material" category was mainly made up of sweet birch (*Betula lenta*)**,** beech (*Fagus grandifolia*), hickory (*Carya spp.*), serviceberry (*Amelanchier arborea*), and basswood (*Tilia americana*). Soil CO_2 evolution (μmol $m^{-2}s^{-1}$) was measured every two weeks on two of the four selected transects. Intervals ranged from 1 m immediately adjacent to the stream to 20 m further up the slope. The furthest measurement location was 50 m from the stream. Concurrent with soil CO_2, volumetric soil moisture content in the upper 20 cm of soil was measured using time domain reflectometry (TDR), and soil temperature was measured using a 15 cm temperature probe. Two stream water samples were collected weekly from each site – one above the proposed treatment and one below – to determine total suspended solids (TSS) and water chemistry. Stream water samples were analyzed for NH_4 using a Perstorp Model 3590 Autoanalyzer (Wilsonville, OR) and for NO_3 using a Dionex Model 4500i Ion Chromatograph (Sunnydale, CA). Conductivity and pH were measured using digital conductivity and pH meters (Orion Models 122 and 611, respectively). The amount of total suspended solids (TSS) in the water was determined using a vacuum filtration system with 1.5 μ glass microfiber filters (Whatman, Clifton, NJ). All analyses were conducted at the Coweeta Hydrologic Laboratory.

Results

The year-round mean concentration of NH_4 was determined to be 0.0054 mg L^{-1}. Seasonal patterns in ammonium levels varied between sites. The High Elevation Control showed seasonal patterns in NH_4 concentrations that peaked in winter and dropped to minima in summer. All other sites showed peaks in spring and lowest levels in winter (Figure 2). The year-round mean concentration of NO_3 was determined to be 0.052 mg L^{-1}. The High Elevation Control, Thirty, and Hundred sites showed seasonal patterns of NO_3 concentrations that peak in spring or summer and drop to lowest levels in fall. Other sites showed maxima in winter and minima in fall (Figure 2). The year-round mean concentration of TSS in stream water for all sites was 6.93 mg L^{-1}; however, year-round mean TSS concentration varied between sites (Figure 3). Stream water concentration means include all levels of flow.

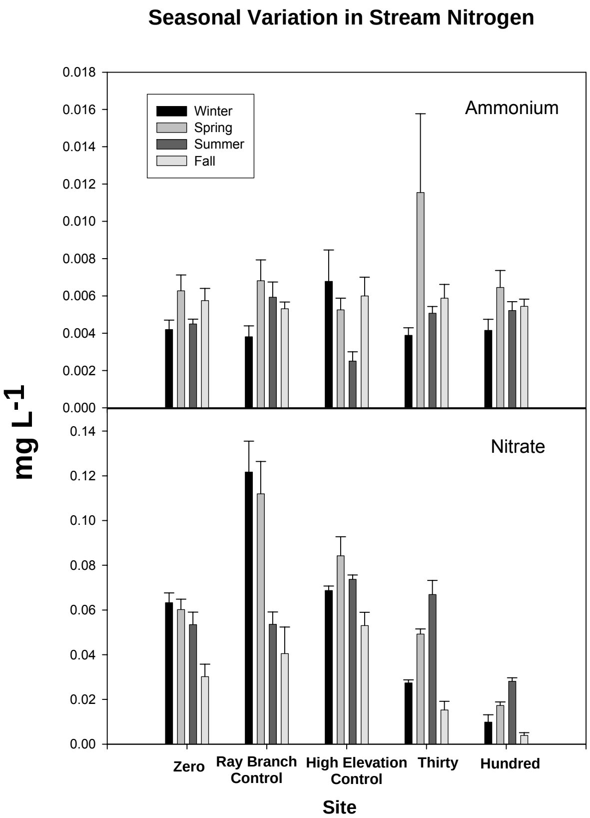

Figure 2: Stream nitrogen concentrations showed seasonal patterns, such as the increase in NH_4 and decrease in NO_3 in the fall.

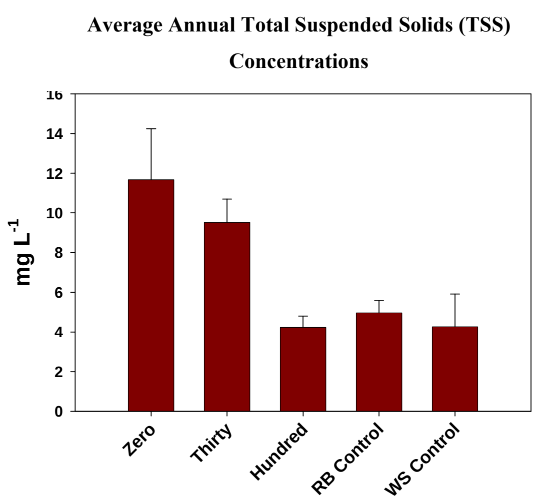

Figure 3: TSS concentrations varied between sites.

In this study, a very weak negative relationship (r^2 = 0.07) between soil CO_2 evolution and soil moisture was observed (Figure 4). A positive relationship between forest floor CO_2 evolution and soil temperature was observed (Figure 5). The relationship between soil CO_2 evolution and soil temperature (r^2 = 0.28) was much more significant than the relationship between soil CO_2 evolution and soil moisture (r^2 = 0.07) (Figures 4 and 5). In Figure 6, correlation between soil CO_2 evolution and soil temperature was most variable within the first 10 to 20 m distance along the transect, and in Figure 7, soil CO_2 evolution and soil moisture content was most variable within the first 10 to 20 m distance along the transect.

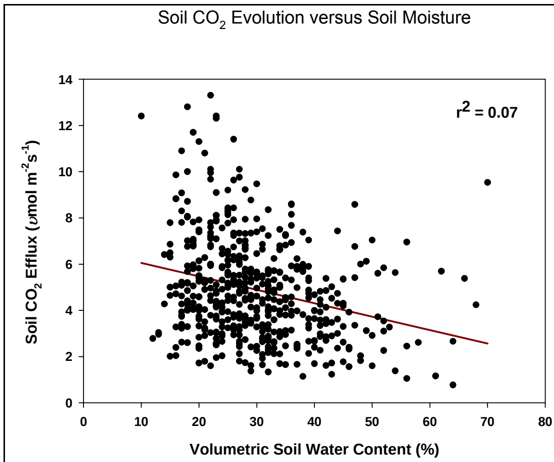

Figure 4: A weak negative relationship was observed between soil CO_2 efflux and soil moisture content.

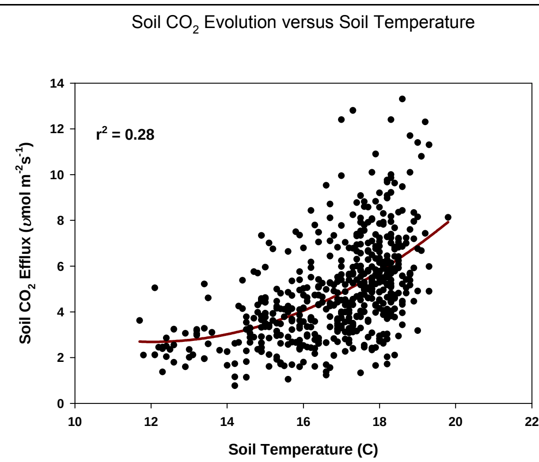

Figure 5: A positive exponential relationship was observed between soil CO_2 efflux and soil temperature.

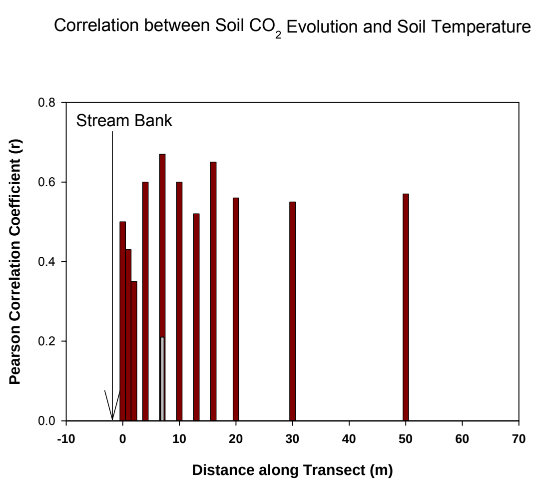

Figure 6: A positive correlation was observed between soil CO_2 efflux and soil temperature. Correlations varied most between 0 and 20 meters from the stream may

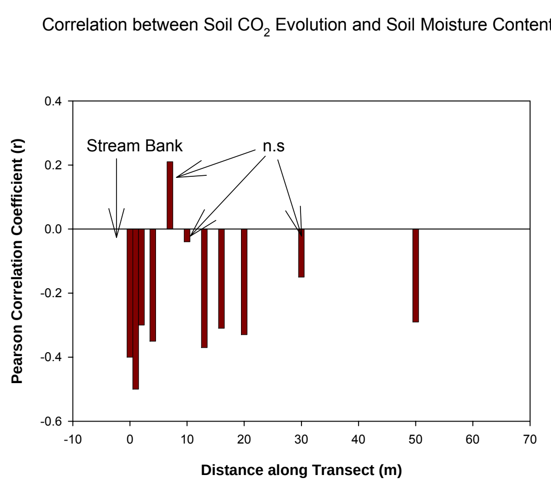

Figure 7 A weak negative relationship was observed between soil CO_2 efflux and soil moisture. Correlations at 7, 10, and 30 meters were not significant (indicated by n.s.)

Mean annual soil CO_2 flux rates did not show a significant difference along the transect from the stream. However, a possible spatial pattern could be inferred from this preliminary data, with the highest rates accruing from 0 to about 10 m and then leveling off after 10 m (Figure 8). Mean annual soil CO_2 flux rates ranged from 4.366 µmol m^{-2} s^{-1} at 2 m to 5.5263 µmol m^{-2} s^{-1} at 10 m from the stream.

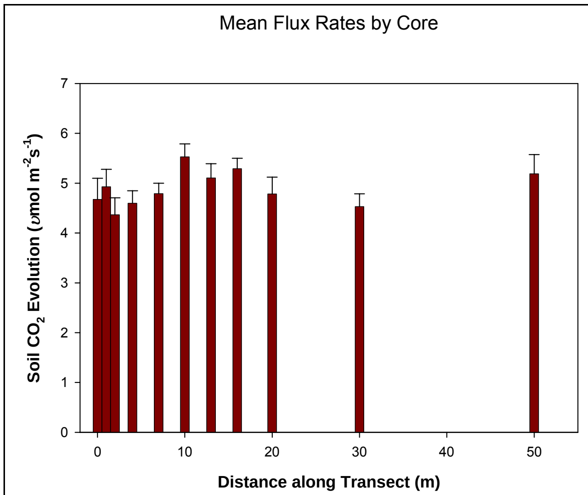

Figure 8: Mean annual soil CO_2 flux rates varied spatially along the transect from 0- to 50-m from the stream, with the highest rates accruing from 0- to about 10-m and then leveling off after 10-m. Mean annual soil CO_2 flux rates ranged from 4.366 µmol m^{-2} s^{-1} at 2-m to 5.5263 µmol m^{-2} s^{-1} at 10-m from the stream.

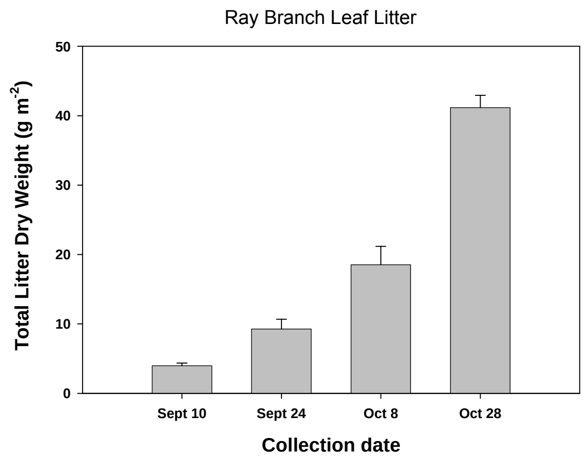

Figure 9: Autumn leaf fall increased over the collection period, but would be expected to begin dropping in terms of mass.

Leaf litter inputs varied spatially and temporally. Autumn leaf fall increased over the collection period (Figure 9), but would be expected to begin dropping in terms of mass. Leaf litter fall for wood, others, conifers (*Tsuga canadensis* and *Pinus strobus*), yellow poplar (*Liriodendron tulipifera*), and rhododendron (*R. maximum*) decreased from 0 m to 50 m from the stream; leaf litter fall for red maples (*Acer rubrum*) and oaks (*Quercus spp.*) increased as you move up the slope (Figure 10). The litter fall data indicates that oaks and red maple were most abundant 30 to 40 m from the stream (Figure 10). Wood, litter fall of other species, and total litter fall were always greatest immediately adjacent to the stream (Figure 10).

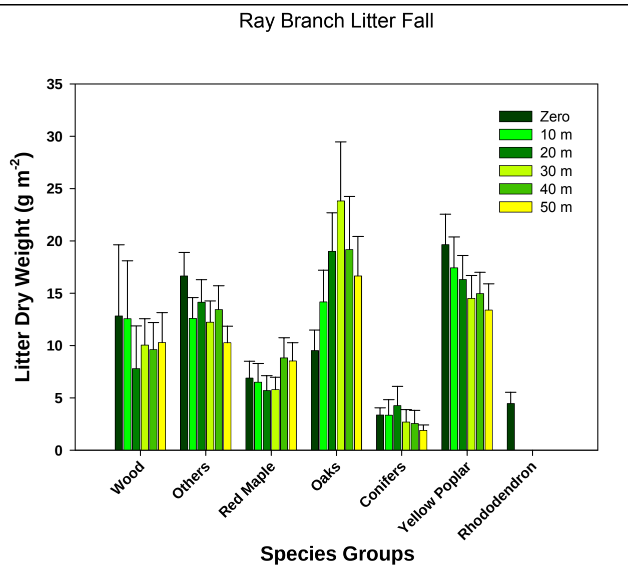

Figure 10: Leaf litter inputs varied spatially. Litter fall from wood, others, conifers, yellow poplar, and rhododendron generally decreased from 0-m to 50-m from the stream. Litter fall from red red maple and oaks generally increased as you move up the slope and peaked between 30 and 40 m from the stream.

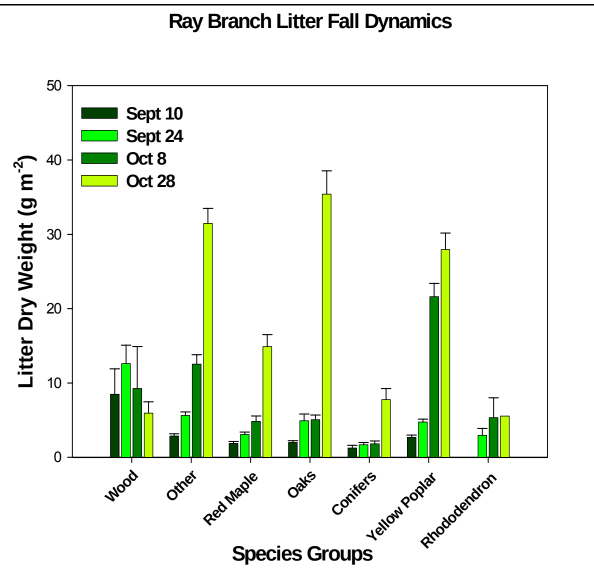

Figure 11: Leaf litter inputs varied temporally. Litter fall inputs increased over the collection period, with the exception of wood, which climaxed prematurely. Litter fall inputs during the collection period were by far the greatest on the Oct. 28 collection in all species groups (except wood).

Different tree species categories dropped leaves at different times, although all categories' leaf loss increased over the collections period, with the exception of wood (Figure 11). As observed in Figure 11, the two most dominant tree species, oak and yellow poplar, climaxed at different times. Yellow poplar began to fall about two weeks before oaks, and yellow poplar leaf-fall also climaxed before the oaks. Conifer needle loss increased through the period. The only category to climax earlier in the season was wood.

Discussion

Variation in stream chemistry showed a complexity of processes. The observed autumnal increase in ammonium concentration was most likely due to the fact that autumn leaf litter deposits organic material that is denitrified during decomposition. The High Elevation Control showed seasonal patterns in NH_4 concentration typical of Southern Appalachian streams – ammonium concentrations peaked in the winter and were lowest in the summer, but all other sites were atypical (Knoepp and Swank 1993). Except at the Zero and Ray Branch Control sites, nitrate concentrations peaked in spring and summer and dropped dramatically in the fall. Such seasonal variation in nitrate concentration reflects the organic material inputs of vegetation in spring and summer, and such seasonal patterns have been observed in the Southern Appalachians in other studies (Mulholland 1992, Clinton et al. 2003). The atypical patterns in nitrogen concentrations were accompanied by large error, indicating possible statistical outliers that may have resulted from contamination in water samples in the field early in this study. Variation in TSS concentrations between sites was most likely due to differences in slope and vegetation.

Carbon inputs to the forest floor originate from death of organisms, aging of leaf and root structures, and primary and secondary production (Wiley 1992, Vose et al. 1995). The respiration of roots, mycorrhizae, and soil micro- and macro-organisms influences carbon output from the forest floor. Most of this carbon is emitted into the atmosphere in the form of CO_2, and the rest enters aquatic systems by way of streams and groundwater (Wiley 1992). Variables that affect the rate of CO_2 efflux include soil temperature and moisture (Edwards 1975, Schlentner and Van Cleve 1985, Vose et al. 1995), soil organic matter (Ewel et al. 1987), soil and root nitrogen levels (Soderstrom et al. 1983), and root biomass (Behera et al. 1990). Data indicated no conclusive relationship between soil CO_2 efflux and soil moisture. This unrelatedness can be explained by the fact that there is more moisture closer to the stream but there is also likely more labile carbon, an important food source for microbes, due to the accumulation of fine woody debris, so the larger amounts of these two variables counteract the positive relationship that is normally observed between soil CO_2 evolution and soil moisture (Hanson et al. 1993). In extremely dry or wet conditions, soil moisture affects soil CO_2 efflux very significantly because water is necessary for decomposition, one important source of soil CO_2, yet excess water can inhibit decomposers' respiration. Of the variables that were measured, the one driving soil carbon efflux was soil temperature because soil moisture was found to be in plentiful supply throughout the study period. A similar relationship has been verified in other studies (Vose et al. 1995, Hanson et al. 1993). Variation in correlations between soil CO_2 efflux and soil moisture and between soil CO_2 efflux and soil temperature from 0 to 20 m from the stream could be attributed to variation in topography due to the presence of old logging roads or could also indicate a leveling off of variation in riparian forest processes beyond 20 m. Some variation in soil carbon may also be explained by additional belowground sources, such as micro-arthropod activity, and by variation in root abundance and size. Data may indicate that soil carbon increased from 0 to 10 m from the stream. Although it can be reasonably asserted that quantities of carbon were greatest nearest the stream, because debris and litter collect at the bottom of the slope where riparian vegetation is most dense, the most probable reasons that data failed to show this increase are because the soil temperatures were coolest near the stream, because soil nearest the stream was unconsolidated alluvium, and because soil nearest the stream was more anaerobic, thus inhibiting respiration.

Litter fall data showed spatial variation in species composition and in species abundances along a gradient perpendicular to the stream. The others category was most plentiful at 0 m because more vegetation occurred around the stream (higher species richness) and because the others category included many moisture-loving species such as sweet birch (*Betula lenta)* and basswood (*Tilia americana)*. Wood and litter fall were always greatest at 0 m on the transect because more vegetation grows adjacent to the stream than farther upslope and because the stream provides an open space where more wind can blow and sunlight can enter, causing the leaves and woody debris to fall. The litter fall data indicates a possible boundary in forest composition because oaks and red maples did not become forest dominants until about 30 to 40 meters from the stream.

Temporal variation in litter fall occurred due to seasonal change and due to the fact that different species drop their leaves at different times during the fall season. Two of the forest dominants, yellow poplar and oaks, climaxed leaf drop at different times. Yellow poplar began dropping and climaxed before the oaks. Wood climaxed earlier in the season most likely because of the two hurricanes, Frances and Ivan, which hit on September 6 and September 16, 2004, respectively. These two storms may have somewhat skewed data toward larger litter falls than usual early in the season. The variation in nutrient concentrations contributed by different tree species affects soil and stream processes temporally (Fisher and Likens 1973, Gregory et al. 1991, Mulholland 1992). The presence and abundance of different species in the riparian zone, combined with the temporal variation in leaf-fall through the autumn, cause complex temporal variation in terrestrial-aquatic nutrient cycles.

Conclusions

Soil processes vary along a gradient from the stream, which results in increased spatial variation in species distributions in riparian zones. Litter fall and coarse woody debris were greatest near the stream, indicating the importance of stream buffers in contributing organic material to the stream ecosystem and aquatic ecosystems downstream. Forest tree species composition changed along the transects from 0 to 50 m from the stream, and species grew more densely nearest the stream. Different species occurred along the spatial gradient from the stream. Trees that prefer wetter conditions, such as yellow poplars (*Liriodendron tulipifera*), rhododendron (*R. maximum*), and conifers (*Tsuga canadensis* and *Pinus strobus*), were more abundant closer to the stream, and most abundant immediately adjacent to the stream. Tree species such as oaks (*Quercus spp.*) and red maple (*Acer rubrum*) were most abundant farther from the stream. The fact that the others category contained the most litter fall immediately adjacent to the stream indicates a higher diversity of species around the stream.

The Southern Appalachian riparian forests, with a mountainous yet warm climate, experience a unique seasonal variation that affects stream nutrient content and retention (Mulholland 1992). Such variation is compounded by, and possibly partially a result of, the timing of the leaf-fall of different species and of the highly varied nutrient contributions of these species (Gregory et al. 1991). Species composition and abundance of riparian vegetation help regulate litter processing rates and consumer community structure in streams (Cummins 1974, Gregory et al. 1991), which makes the conservation of streamside vegetation essential to the functioning of terrestrial-aquatic ecosystems.

The high diversity of species in riparian zones, indicated by the higher litter fall in the others category around the stream, is another important factor in riparian zone structure. This observation is supported by the fact that high species diversity has been noted in various studies by Odum (1978), Gregory et al. (1991), Malanson (1993), and Naiman et al. (1993). The high biodiversity resulting from variation of soil processes and species composition along the gradient from the stream is the reason riparian zones need to be studied, conserved, and managed well.

Stream buffers are an effective way to manage and conserve this biodiversity and to protect the functions that these land-water linkages perform (Lindenmayer and Franklin 2002). Unnatural disturbances of these important ecosystems should be kept to a minimum in order to maintain water quality and terrestrial and aquatic habitats. More information is needed on riparian zone structure and function in order to create better-informed stream buffer policies. Riparian buffer ordinances are often not based on scientific data and thus do not serve their purpose (Wenger 1999). My study showed that species are most dense around the stream and that forest structure and dominant species change as one moves away from the stream. For example, oaks and red maples did not become forest dominants until about 30 meters from the stream, and such changes in vegetation composition are often used to define buffer width. These changes in vegetation composition, however, may still play an important role in nutrient contributions and processes both terrestrially and aquatically. Another important, and often overlooked, factor in conserving riparian zones is soil processes. Spatial variation in soil CO_2 evolution appeared to level off at about 10 m from the stream, implying the possibility that 10 m buffer widths may be sufficient to conserve soil processes. In addition, the more or less constant soil CO_2 flux rates and soil moisture content readings beyond 10 to 20 m from the stream may indicate a possible boundary in riparian soil processes. Effects of different widths of stream buffers on stream nutrient cycling and forest structure and function must be studied further in order to understand this variation in structure and function and to conserve this variation successfully.

When considering advisable stream buffer widths, one must also consider their economic effects so that indices of this study can be implemented successfully. Large buffer widths, especially along small, headwater streams, can reduce the area of manageable timberland, especially in upper stream reaches. Stream buffers may make upland forest fragments inaccessible or economically inviable. Stream buffers, especially when not continuous, can cause fragmentation in an ecosystem and cause high environmental disparity between densely vegetated riparian zones and upland habitats with managed forests that are younger and more simply structured (Carey et al. 1999, Lindenmayer and Franklin 2002). Also, riparian buffer system policies may not consider natural disturbance regimes and the historic range of streamside vegetative cover. A complete understanding of socioeconomic effects, as well as ecological effects, needs to be explored with respect to riparian buffers.

My study characterized the variation in structure and function of a riparian forest along a gradient from a stream. Data thus far indicate variation in soil CO_2, species composition, and stream chemistry. This variation should be considered by managers and policy-makers so that enforced buffer widths are effective in protecting stream ecosystem health. Assessment of variation in soil processes and stream chemistry before and after a logging treatment will allow effective detection of any treatment effects. More long-term study of such variation will contribute to understanding the importance of structure and function in riparian buffers to stream ecosystem health.

References Cited

Behera, N., S.K. Joshi, and D. Pati. 1990. Root contribution to total soil metabolism in a tropical soil from Orissa, India. *Forest Ecology and Management* 36:125-134.

Brosofske, K.D., J. Chen, R.J. Naiman, and J.F. Franklin. 1997. Harvesting effects on microclimate gradients from small streams to uplands in western Washington. *Ecological Applications* 7:1188-2000.

Campbell, I. C., K. R. James, B. T. Hart, and A. Devereaux. 1992. Allochthonous coarse particulate organic material in forest and pasture reaches of two south-eastern Australian streams. I. Litter accession. *Freshwater Biology* 27: 341-352.

Carey, A. B., B. R. Lippke, and J. Sessions. 1999. Intentional systems management: Managing forests for biodiversity. *Journal of Sustainable Forestry*. 9(3/4): 83-125.

Clinton, Barton D., J.M. Vose, J.D. Knoepp, and K.J. Elliott. 2003. Stream nitrate response to fire in the Appalachians. Pages 174-181. *In* K.E.M. Galley, R.C. Klinger, and N.G. Sugihara (eds.). Proceedings of Fire Conference 2000: The first National Conference on Fire Ecology, Prevention and Management. San Diego, CA, Nov. 27-Dec. 1, 2000. Misc. Pub. No. 13, Tall Timbers Research Station, Tallahassee, FL.

Cummins, K.W. 1974. Stream ecosystem structure and function. *Bioscience*. 24:631-641.

Edwards, N.T. 1975. Effects of temperature and moisture on carbon dioxide evolution in a mixed deciduous forest floor. *Soil Science Society of America Journal* 39:361-365.

Elwood, J. W., J. D. Newbold, R. V. O'Neill, R. W. Stark, and P. T. Singly. 1981. The role of microbes associated with organic and inorganic substrates in phosphorus spiraling in a woodland stream. *Verhandlungen der Internationale Vereinigung für Limnologie* 21:850-856.

Ewel, K.C., W.P. Cropper, Jr., and H.L. Gholz. 1987. Soil CO_2 evolution in Florida slash pine plantations. I. Importance of root respiration. *Canadian Journal of Forest Resources* 17:325-329.

Fisher, S.G., and G.E. Likens. 1973. Energy flow in Bear Brook, NH: an integrative approach to stream ecosystem metabolism. *Ecological Monographs*. 43:421-439.

Forest Ecosystem Management Assessment Team. 1993. *Forest ecosystem management: An ecological, economic, and social assessment.* USDA Forest Service, Portland, Ore.

Gregory, S. V., F. J. Swanson, A. McKee and K. W. Cummins. 1991. An Ecosystem Perspective of Riparian Zones: Focus on links between land and water. *Bioscience* 41(8): 540-551.

Grimm, N. B. 1987. Nitrogen dynamics during succession in a desert stream. *Ecology* 68: 1157-1170.

Hanson, P.J., S.D. Wullschleger, S.A. Bohlman, and D.E. Todd. 1993. Seasonal and topographic patterns of forest floor CO_2 flux from an upland oak forest. *Tree Physiology* 13:1-15.

Haycock, N.E., T.P. Burt, K.W.T. Goulding, and G. Pinay. 1997. Buffer zones: Their processes and potential in water protection. Quest Environmental, Harpenden, UK.

Hubbard, R. K. and R. R. Lowrance. 1994. Riparian forest buffer system research at the Coastal Plain Experimental Station, Tifton, Georgia. *Water, Air and Soil Pollution* 77: 409-432.

Jasper, S., and 12 others. 1983. Primary productivity in a large, temperate lake with river interflow: Kootenay Lake, British Columbia. *Canadian Journal of Fish and Aquatic Science.* 40: 319-327.

Knoepp, J.D. and W.T. Swank. 1993. Site preparation burning to improve southern Appalachian pine-hardwood stands: nitrogen responses in soil, soil water, and streams. *Canadian Journal of Forest Resources.* 23:2263-2270.

Likens, G. E., F. H. Bormann, R. S. Pierce, J. S. Eaton, and N. M. Johnson. 1977. *Biogeochemistry of a forested ecosystem*. Springer.

Lindenmayer, D. B. and J. F. Franklin. 2002. *Conserving Forest Biodiversity: A Comprehensive Multiscaled Approach*. Washington: Island Press, The Center for Resource Economics.

Lowrance, R., R. Leonard and J. Sheridan. 1985. Managing riparian ecosystems to control nonpoint pollution. *Journal of Soil and Water Conservation* 40(1): 87-91.

Lowrance, R., and 12 others. 1997. Water quality functions of riparian forest buffers in Chesapeake Bay watersheds. *Environmental Management* 21(5): 687-712.

Malanson, G. P. 1993. *Riparian Landscapes.* Cambridge, UK: Cambridge University Press.

Meyer, J. L., and G. E. Likens. 1979. Transport and transformation of phosphorus in a forest stream ecosystem. *Ecology.* 60: 1255-1269.

Mulholland, P.J., 1992. Regulation of Nutrient Concentrations in a Temperate Forest Stream: Roles of Upland, Riparian, and Instream Processes. Limnology and Oceanography 37(7):1512-1526.

Naiman, R. J. and R. E. Bilby, eds. 1998. *River ecology and management: Lessons from the Pacific coastal ecoregion*. New York: Springer-Verlag.

Naiman, R. J., H. Decamps and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. *Ecological Applications* 3(2): 209-212.

Odum, E. P. 1978. Ecological importance of the riparian zone. Pp 2-4 in: Johnson and McCormick (tech. coords.), *Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems.* U.S. Dept. of Agriculture Forest Service Gen. Tech. Report WO-12.

Osborne, L. L. and D. A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. *Freshwater Biology* 29: 243-258.

Schlentner, R.E., and K. Van Cleve. 1985. Relationships between CO_2 evolution from soil, substrate temperature, and substrate moisture in four mature forest types in interior Alaska. *Canadian Journal of Forest Resources*. 15:97-106.

Soderstrom, B., E. Baath, and B. Lundgren. 1983. Decrease in soil microbial activity and biomasses owing to nitrogen amendments. *Canadian Journal of Microbiology*. 29:1500-1506.

Spackman, S. C. and J. W. Hughes. 1995. Assessment of minimum corridor width for biological conservation: Species richness and distribution along mid-order streams in Vermont, USA. *Biological Conservation* 71: 325-332.

Swank, W. T. 1988. Stream chemistry responses to disturbance, pp. 339-358. *In* W. T. Swank and D. A. Crossley, Jr. [eds.], Forest hydrology and ecology at Coweeta. Springer.

Swift, L. W. Jr. 1986. Filter strip width for forest roads in the Southern Appalachians. *Southern Journal of Applied Forestry* 10: 27-34.

Thomas, K., R. H. Norris, and G. A. Chilvers. 1992. Litterfall in riparian and adjacent forest zones near a perennial upland stream in the Australian Capital Territory. *Australian Journal of Marine and Freshwater Research* 43: 511-516.

Vose, J.M., B.D. Clinton, and V. Emrick. 1995. Forest floor CO_2 flux from two contrasting ecosystems in the Southern Appalachians. Pages 165-167. *In*: K.W. Gottschalk and S.L.C. Fosbroke (eds.), Proceedings of the 10th Central Hardwood Forest Conference. Morgantown, WV: Mar. 5-8, 1995. Pub. USDA Forest Service, Northeastern Forest Experiment Station.

Wenger, Seth. 1999. *A Review of the Scientific Literature on Riparian Buffer Width, Extent and Vegetation.* Athens, GA: Office of Public Service & Outreach, Institute of Ecology, University of Georgia.

Wiley, J. & W. 1992. Forest carbon cycling. Pages 806-808. *In:* A.H. El-Shaarawi and W.W. Piegorsch (eds.), *Encyclopedia of Environmetrics, Vol 2.* New York.

Successional Dynamics of Dulany Bog

Michael Nichols

Introduction

Dulany Bog, located near the intersection of NC 107 and Bull Pen Road in Jackson County, NC, represents one of the rarest communities of the southern Appalachian Mountains, the southern Appalachian bog (Pittillo 1993). As a result, species that occur in the bog, like golden club (*Orontium aquaticum*) and populations of the threatened swamp pink (*Helonias bullata*), tend to be rare as well (Pittillo 1993). Several abiotic factors limit the distribution of Southern Appalachian bogs. They are found in valley bottoms not subject to flooding, have very wet organic or mineral soils, and are very acidic (Schafale and Weakley 1990). The soils are saturated for a majority of the year, and the bogs are normally fed by ground water seepage. Vegetation forms zoned patterns of herb and shrub dominated areas, with mats of sphagnum moss covering much of the surface (Schafale and Weakley 1990). These distinctive wetland habitats are home to a wide array of plants and animals, including pitcher plants (*Sarracenia spp.*), sundews (*Drosera spp.*) and other rare plants (Sargent and Carter 1999).

Bog communities have been disturbed by human activities and, like other wetlands worldwide, are rapidly disappearing. For centuries, humans have exploited peatlands for fuel, agriculture and forestry (Gorham and Rochefort 2003). Bogs have been extensively damaged or destroyed in other countries as well, making the conservation of remaining peat lands an important goal internationally (Cheshire Biodiversity 1997). Hefner and Brown (1984) claimed that wetland loss in the Southeast was occurring at a rate of 156,000 ha per year (Riggsbee 1998). Between 1985 and 1995, an estimated net loss of 46,800 ha of wetlands occurred in the United States, a rate 60% slower than the loss rate during the 1970's and 1980's (USFWS 1997). Only 10% of the original Southern Appalachian bogs remain, comprising a total area of <500 acres (Sherpa 2004). In the southeastern United States, only 2.5% of pitcher plant (*Sarracenia spp.*) habitat remains (Denhof 2000). The population of *Helonias bullata* in Dulany Bog is one of only eight populations in the southern Appalachian Mountains (Sutter 1984).

Succession by native and non-native invasive woody species is a major problem facing degraded bog communities, especially when conditions arise that favor the growth of these species. Woody invasion leads to shading out and eliminating pitcher plants and other herbaceous species from the bog (Denhof 2000). In some glaciated areas, bogs are considered a successional stage in the process of a kettle lake becoming a forested wetland (Sargent and Carter 1999). In contrast, raised bogs typical of Europe are considered to be the end result of succession on lake mud and woody invasion of these bogs is rare (Van Breemen 1995). In bogs where woody invasion takes place, natural succession is considered a threat to bogs, leading to degradation or destruction of the habitat. This natural process is accelerated rapidly by drainage of the bog soils (Sargent and Carter 1999). Successional woody invasion of the wetland and the subsequent shading of the herbaceous layer is a major concern in Dulany Bog. Only two of the *Helonias bullata* populations in the southern Appalachians have produced seed recently (Sutter 1984). Herbaceous plants of mountain bogs need full sun to encourage flowering and seed production (Denhof et al. 2004).

Favorable conditions for invasion by woody species occur when the bog's hydrology is altered, lowering the water table, and fire, which can slow or prevent woody invasion, is suppressed (Denhof 2000). Beaver activity, which has been responsible for periodically flooding the bog, is controlled by road crews to prevent flooding of roads adjacent to the bog (Riggsbee 1998). The DOT has dredged and straightened the stream along the eastern edge of the bog. Seven years with unusually low rainfall may have created drier conditions in Dulany Bog (Wykle personal communication). An apparent ditch through the bog also may have altered the bog's hydrology (Schafale et al. 1999). Fire was used annually to manage the bog from the early 1940s until 1963, then once again in 1973 (Floyd 1973). The hydrology of Dulany Bog has been altered by human activity, and fire, which once maintained a more open bog community, has been absent for 30 years.

The encroachment of woody species into Dulany Bog has proceeded to a point where woody succession threatens to destroy the open character of the bog and shade the herbaceous layer. Given the current threats and impacts to Dulany Bog, natural processes are unlikely to maintain the bog in its current condition, and, as a result, restoration and management practices must be implemented to protect this rare plant community and unique habitat. The Highlands Biological Foundation and the USDA Forest Service have entered into a cooperative agreement to manage selected portions of the bog to explore the feasibility and effectiveness of restoring the bog. The purpose of this study is: 1) to survey areas of the bog to obtain baseline data for future management treatments to Dulany Bog; 2) to study areas of the bog with different aged shrub canopies and use a space for time substitution to speculate about the successional dynamics of the community; 3) to examine different methods for controlling woody invasion of bog communities; and 4) to use this information to propose a strategy for restoring and maintaining this bog.

Literature Review

Southern Appalachian Bogs

Bogs are unique wetland habitats with complex biotic and abiotic systems and processes. Mountain bogs tend to form as a result of a layer of weathering resistant rock which forms a natural dam site across a stream. Erosion of softer rock formations upstream creates basins behind these dams (McDowell 1973). The water that collects in these basins forms areas too wet for most trees to grow and such areas have poorly drained, dark soils and cold water (Sargent and Carter 1999). Sphagnum moss "wicks up" water from below and thrives in these cool, wet, nutrient-poor, acidic conditions with low levels of oxygen (Ottawa Field-Naturalists' Club 2002). Sphagnum is able to dominate these areas by creating the acidic, nutrient-poor, anoxic conditions that allow it to outcompete other plants (Van Breemen 1995). The low levels of nutrients available in the system supports a specific group of plants such as carnivorous species like pitcher plants and sundews which obtain supplemental nutrients by digesting insects (Sargent and Carter 1999).

Bogs are also important to animal communities, although few species live exclusively in the bog. Bogs attract insects, mice, frogs and toads, and are important areas for predators like raccoons and herons that feed on these species (Sargent and Carter 1999). Bogs are important habitats for animals that are adapted to living in bogs, such as the bog turtle (Riggsbee 1998). Southern Appalachian bog habitats may also support certain bird populations, particularly neotropical migrants, although there is not enough evidence to support or reject this hypothesis (Riggsbee 1998).

Conservation

The main goal of conservation efforts in southern Appalachian bogs should be to conserve intact habitat with a surrounding buffer zone (Denhof 2000). Bogs are very sensitive to disturbance and without a sufficient buffer area to protect them from human activities, species diversity may decline (Sargent and Carter 1999; Denhof 2000). Buffer zones also serve to protect the external hydrologic regime (Eggelsmann 1980). An ideal buffer width around the wetland is at least 100 m (Sargent and Carter 1999). Threats to bogs include drainage, excess nutrient input, modification for agriculture and forestry, natural succession (accelerated by drainage), inappropriate or insufficient management, and fire (Cheshire Biodiversity 1997). Homes adjacent to Dulany Bog may have negatively impacted the bog and measures may be needed to protect the bog from pollutant runoff from these home sites (Pittillo 1993). Broad strips of grass should be planted between the bog and agricultural fields to reduce nutrient runoff into the bog (Denhof et al. 2004). In addition to buffer strips around the bog, sympathetic management from surrounding landowners is needed to prevent possible disturbance from surrounding areas (Sargent and Carter 1999). This includes keeping domesticated animals and pets out of the area (Slattery and Kesselheim 1995).

Previous Management

Historically, fire has been a key factor in managing Dulany bog. Rare catastrophic fires helped to maintain mountain bog communities (Denhof et al. 2004). In many areas, bogs burned with the surrounding area in fires set by Native Americans to clear game food plots (Sargent and Carter 1999; Denhof 2000). Dulany Bog was formerly part of a cattle grazing area and was burned annually for at least 20 years before being acquired by the Highlands Biological Foundation in 1963. The bog was burned twice in 1963, then left unburned until 1973, when fire was used to control the successional advancement of alder (*Alnus serrulata*) and swamp rose (*Rosa palustris*) (Floyd, 1973). Since the winter of 1973-74, there has been no use of fire to manage Dulany Bog, with the exception of burning piles of alder that were cut and removed by hand (Riggsbee 1998; Wykle, pers. comm.). Pittillo (1993) noted however, that the slashing and burning used to control the growth of alder in Dulany Bog has met with only limited success.

Restoration

When a bog is overgrown with trees and shrubs to the point that pitcher plants, sundews, orchids, and other herbaceous plants struggle for light, the bog needs to be restored. The goal for bog restoration and management efforts should be to return the habitat to an open wet wildflower meadow with pitcher plants, sundews, orchids, sphagnum, native grasses and sedges, and other herbaceous flowering plants. In the restoration phase, active management is needed annually until woody and invasive species are reduced to <50% cover (Denhof et al. 2004). Maintaining the herbaceous layer is crucial to restoration and conservation of bogs in the southeast (Denhof 2000). Full sun or at least afternoon sun with a minimum of six hours of full sunlight is best for recovery. Most active management should take place in the dormant season (late fall and winter) to give bog plants a chance to produce and scatter seeds (Denhof et al. 2004). Restoration work should be carried out with a minimum of traffic and machinery to prevent damage to the habitat (Denhof 2000). Several methods, including hand-clearing, mowing, and fire, can be used to control the invasion of woody species and restore overgrown bogs to a more open condition. The pros and cons of each of these methods are examined below.

Hand-clearing

One of the most labor-intensive restoration and management strategies is hand clearing. This practice is a lot of work, but it helps to restore the bog to a more open condition when other techniques, such as fire and mowing, cannot be used. Small trees and shrubs should be cut and removed from the area. Some managers pile this debris in areas of the bog where fire does not carry well. Herbicide may be applied on cut stems to kill the root system and prevent re-sprouting. If this method is ineffective on a large scale over the entire bog, it may be applied locally around targeted plants like pitcher plants or, in Dulany Bog's case, *H. bullata*, to promote seed set (Denhof et al. 2004). Hand clearing of alder was performed in Dulany Bog in 1988-

1989, but it had regrown to cover most of the bog's interior when visisted ten years later (Schafale et. al 1999).

Mowing

Mowing works well to remove woody plants that have invaded the bog if fire cannot be used (GPCA 2004). Mowing should be conducted annually in the dry season to prevent rutting and compacting the bog soil (Denhof et al. 2004). Unfortunately, conventional mowing equipment, such as a tractor or walk-behind mower, would be ineffective in Dulany Bog due to the very soft soil. Specialized equipment exists that would likely be effective, but it is considerably more expensive (Wykle pers. comm.). Another drawback to mowing is that cut materials need to be raked out of the area to prevent the accumulation of humus (Denhof et al. 2004).

Fire

Prescribed fire can be used to remove woody plants to restore and maintain bogs, if there is enough herbaceous material to carry the fire (Denhof et al. 2004). Disturbances such as fire are important for maintaining diversity, including threatened and declining plant species in peat bogs and have been shown to increase species richness, diversity, and evenness (Norton and De Lange 2003). Fire rarely travels in mountain bogs, however, due to the wet conditions (Denhof et al. 2004). Maintaining a good herbaceous layer will restore the fuel layer beneficial to future burns. Winter burns are useful to remove woody plants and restore the herbaceous fuel layer. In some cases, the Atlanta Botanical Garden uses both summer and winter burns in combination to restore severely degraded bog habitats. Winter burns are also effective in controlling the encroachment of sedge and grass species that can choke out less vigorous herbaceous species. If bogs are burned early in the year, flower buds will not be damaged (Denhof 2000). Burning helps to release chemicals that have washed into a bog, reduces nutrient accumulation, and exposes mineral soil needed for herbaceous seedlings to establish (Denhof et al. 2004). Burning also helps to cause die-back in woody species and creates higher light conditions. Leaching of nutrients out of the bog system is much greater in burns during winter, as it occurs before plants can take up the nutrients. This favors pitcher plants and other species that require low levels of nutrients, but burning at the same time every year should be avoided because burning at different seasons may promote diversity (Denhof 2000; Denhof et al. 2004). Similarly, fires of varying scales and intensities are likely to promote greater overall floristic diversity (Whelan 1995). Other than burning at different times to promote diversity, fires should be conducted in late fall or winter to allow seed set of herbaceous plants (Denhof, et. al 2004). Another restoration method involving fire, when working with highly degraded areas, is to cut the shrubs and trees, and later scorch the re-sprouting stems with a propane flame thrower. This damages or kills the cambium, making the woody plants susceptible to disease and decay, especially in summer when pathogen concentrations are highest (Denhof 2000). However, intense fires associated with very dry conditions have been found to damage the soils of other peatlands, slowing vegetation recovery. Severe fires have caused major alterations to the underlying soils of moorlands and in extreme cases, turned thin layers of peat into layers of ash (Maltby et al. 1990). On the other hand, less intense management burns have been effective while maintaining the integrity of the underlying peat (Norton and De Lange 2003). If prescribed burning is deemed an appropriate management tool, public fear and intolerance of prescribed fire may present an obstacle to management practices. Close community relations and education programs are needed to demonstrate the benefits of prescribed fire, as the general public knows very little about fire ecology (Denhof 2000).

Hydrology

In addition to removing invasive woody species from the bog, work must be done to restore and maintain the complex hydrology of the bog. Restoration and maintenance of a bog's hydrology and soil structure are as important as maintaining the vegetation (Denhof 2000). The main goals for restoring and maintaining the hydrological system are to ensure that a source of clean water enters the bog and to slow and spread the water moving through the bog so that the water moves across the bog in a sheet. Mountain bogs are fed by a combination of streams and seeps and require clean sources of water (Denhof et al. 2004). Water courses in bogs should not be altered from their natural condition because this can create drier conditions in the bog and increase the invasion of tree and shrub species (Sargent and Carter 1999). Also, it is vital not to break through the hardpan when restoring the hydrology of the bog, as this may increase drainage of the wetland (Denhof 2000). Protection of both the water source and outlet is vital to restoring and maintaining the complex hydrology (Denhof et al. 2004). Sphagnum moss helps to slow and spread the water moving through the bog, so its presence should be encouraged and protected.

Siltation

Sediment deposition is another problem facing Dulany Bog, as well as other wetlands. Water-borne silt entering the bog covers native plants and their seedlings, layering the bog with soils of different compositions and pH levels, and introduces seeds of invasive species. Sources of erosion upstream should be identified and fixed, allowing native plants to re-cover stream banks. Silt fences can be used to catch erosion and filter out sediment. Drainage ditches should be filled in with dirt or blocked with woody debris to slow and spread the water. Streams running through the bog should similarly be dammed to slow and spread the water and prevent the stream form channeling deeper and deeper (Denhof et al. 2004). These dams help to slow and spread the water, reduce erosion, act as a sieve to prevent silt from choking out bog plants, and maintain or restore a high water table, which is an integral element of these systems (Denhof 2000).

Beavers

Beaver activity has historically helped to maintain bog communities in the mountains (Denhof et al. 2004). Their activity, which has been discouraged by the DOT in Dulany Bog (Riggsbee 1998), may be helpful in flooding out unwanted woody species. Pittillo (1993) argued that their activities should be encouraged in the bog. Hydrologic change by beavers can help create paludal bogs, bogs formed on ground that was once dry land. Herbivory may also be helpful in controlling woody growth in the bog as some animals prefer woody plants in winter (Sargent and Carter 1999).

Maintenance

Once the bog community has been restored to a more open condition, it must be regularly maintained. Without regular management, woody species will return and dominate the bog again. Well-established woody thickets can begin to return within one year without management practices to slow their growth (Denhof et al. 2004). Woody invasion of Dulany Bog was a problem in 1973 after management had ceased ten years earlier. The last management done to the bog was in 1988-89, and alder is once again well established in these areas (Schafale et al 1999).

Maintaining a good herbaceous layer will provide fuel for future management burns. Summer burns are effective to control resprouting of woody invasives (Denhof 2000). If fire cannot be used, the herbaceous layer can be hand cut, weed whacked, or scythed in late fall or early winter. If the entire area cannot be maintained regularly, some management is better than none at all. For example, the areas directly around pitcher plants and *H. bullata* in the heart of the bog can be cleared to promote flowering and seed set (Denhof et al. 2004).

Methods

Dulany Bog, a southern Appalachian bog undergoing woody succession, was selected as the study site for this research. Located south of Cashiers, NC near the intersection of NC 107 and Bull Pen Road, this bog is roughly 40 acres in size with nine acres of bog proper (Floyd 1973; Pitillo 1993). Three units of varying shrub canopy ages were established to examine changes in vegetation composition, basal area of woody species, pH, and nutrient concentrations at different stages of succession. Pickett (1989) concluded that this space-for-time substitution could be used to determine qualitative trends in succession.

To facilitate future management, the units were circular, with a radius of 8 m, the largest area that readily encompassed bog vegetation. One unit was placed into an area undergoing woody succession, primarily by *Alnus serrulata*. Another unit was placed in an area characterized by a more open canopy, with a larger abundance of wetland grasses, ferns, and shrubs such as *Rosa palustris*. A third unit was established in an area with a thicker, taller canopy of *A. serrulata*. Each unit was bisected by a 32-m transect running parallel to the flow of water through the bog. Twelve 1 x 1 m sampling plots were placed along each transect: six experimental plots in the treatment area, and six control plots outside the treatment area (Figure 1). In each plot, the percent cover of sphagnum, ferns, wetland grasses and other species was recorded. An arcsine transformation was performed on percent cover data before analysis, then converted back to percent cover (Sokal and Rohlf 1995, p. 421). Woody species in the plots were sampled individually by measuring the diameter of each stem. The plots were photographed for qualitative assessment, and the subsurface water was sampled for both temperature and pH. Six plots in each unit (two upstream, two in the treatment area, and two downstream) were chosen at random to test for nitrates and phosphates.

Shrub canopy ages were determined by cutting six of the largest stems of *Alnus serrulata* within 3 m of the transect in the central portion of each unit and then counting the number of rings of each specimen. Statistical significances between units were tested using single factor analysis of variance (ANOVA), using Microsoft Excel©.

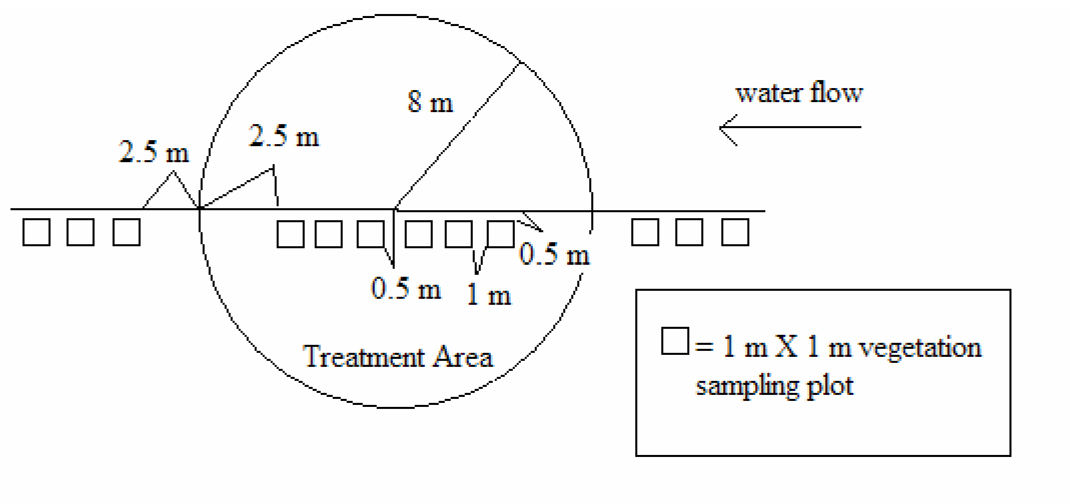

Figure 1: Basic design of sampling unit

Results

The mean ages of the shrub canopies, as determined by number of annual rings from alder trees growing in the units, were 12.83 ± 1.47, 19.33 ± 5.43, and 23.00 ± 2.00 years for units 1, 2, and 3, respectively (Table 1). The mean ages of units 1 and 3 were statistically different ($P<0.001$; Table 3). The mean age of unit 2 was statistically different from that of unit 1

(P=0.018; Table 3). Total basal area per m^2 varied among the three units, with unit 2 having the most total basal area, and unit 1 having the least total basal area (Figure 2). Unit 1 had the least total basal area and was statistically different from unit 2 (P=0.018) and unit 3 (P<0.001; Figure 2). Sphagnum cover decreased from unit 1 to unit 3 (Figure 3). The sphagnum cover of unit 3 was significantly different from that of units 1 and 2 (P<0.001; Table 3). Wetland grasses increased in cover from unit 1 to unit 3, but unit 2 had zero cover by wetland grasses (Figure 3). Unit 1 had more fern cover than unit 2 and unit 3 had no ferns in the sampling plots (Figure 4). Unit 1 was the only unit to have pitcher plants in the sampling plots (Figure 4). The average pH of the subsurface water increased from unit 1 to unit 3 and all differences were statistically significant (P<0.05; Figure 5). The negative trend in the graph of average canopy age vs. sphagnum cover yields an intercept with the x-axis at 32.43 years (Figure 6). The average Nitrate Nitrogen concentration was 1 ppm for all three units and <0.05 ppm of Orthophosphate (Table 2). Raw data from this project is available in an appendix in the Reinke Library of the Highlands Biological Station.

	N	**Mean (yrs)**	**± s**	
Unit 1	6	12.83	1.47	A
Unit 2	6	19.33	5.43	B
Unit 3	6	23.00	2.00	B

Table 1: Shrub canopy ages. Units with different letters have statistically different canopy ages.

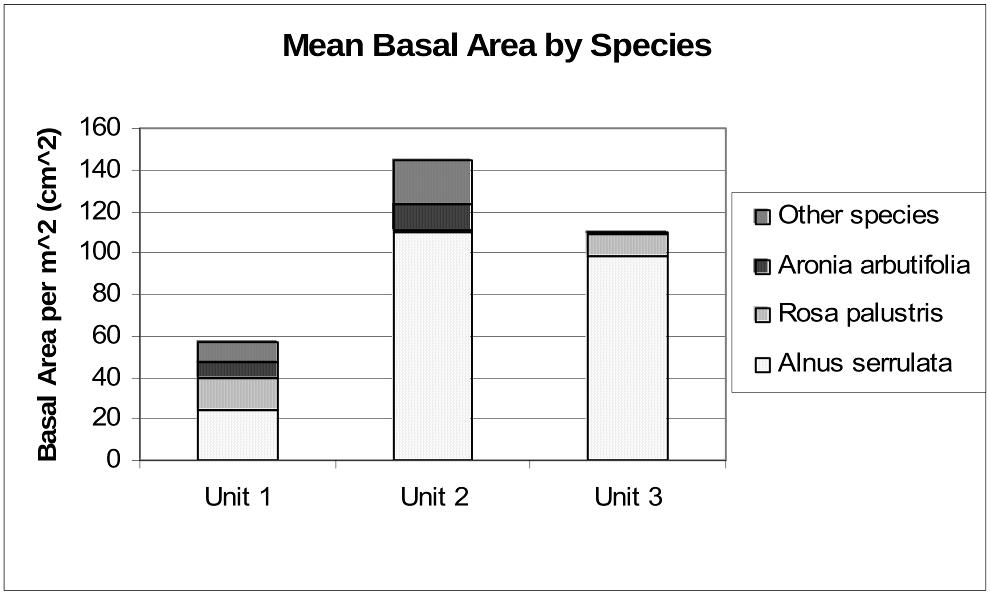

Figure 2: Basal area by species, calculated by adding the computed basal areas of each stem, then dividing by the total area of sampling plots for each species.

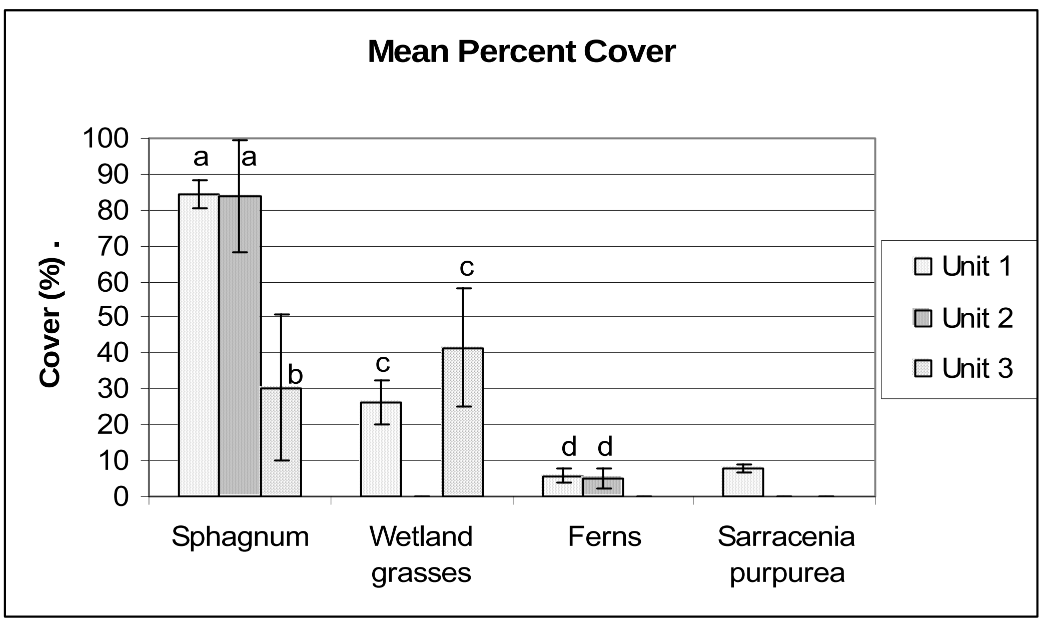

Figure 3: Mean percent cover for sphagnum, wetland grasses, ferns, and *Sarracenia pupurea*. Means with unlike letters are statistically different (P < 0.05).

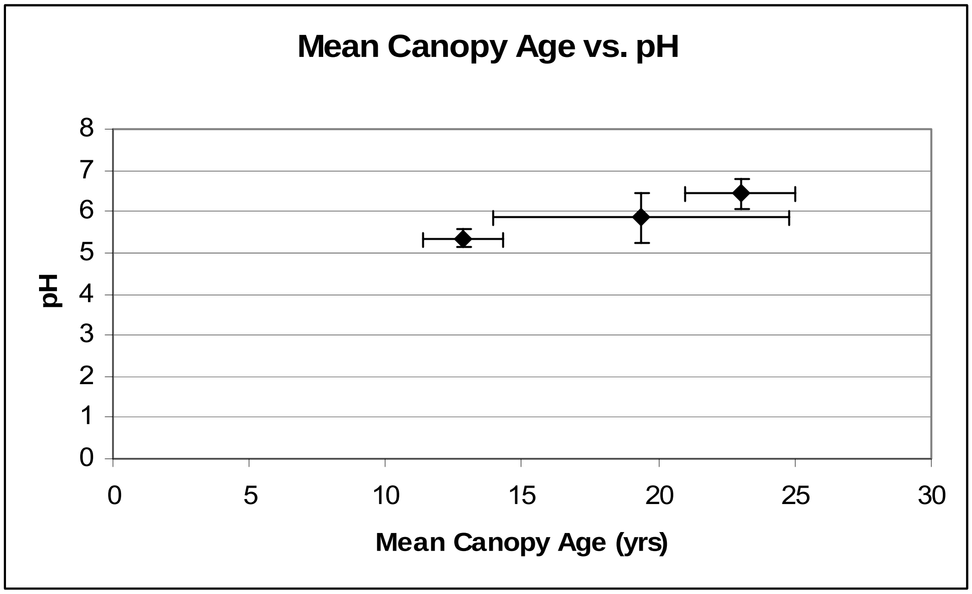

Figure 4: Mean canopy age vs. pH.

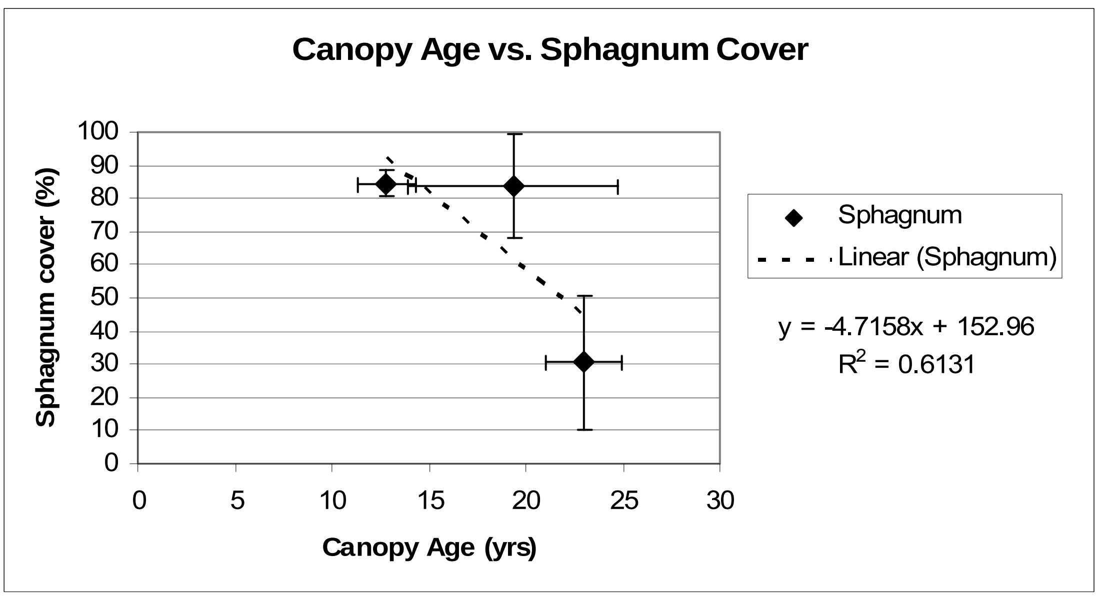

Figure 5: Average canopy age vs. sphagnum cover. The regression line would intercept at 32.43 years.

	Nitrate (ppm Nitrate Nitrogen)	Phosphate (ppm Orthophosphate)
Unit 1	1	<0.05
Unit 2	1	<0.05
Unit 3	1	<0.05

Table 2: Average nutrient concentrations of subsurface water.

	df	F	P-value
Units 1 & 2			
Sphagnum	1; 22	0.011	0.918
Wetland grasses	1; 22	13.321	0.001
Ferns	1; 22	0.009	0.925
Sarracenia purpurea	1; 22	7.007	0.015
pH	1; 22	7.467	0.012
Total basal area	1; 22	5.481	0.028
Mean canopy age	1; 10	8.014	0.018
Units 2 & 3			
Sphagnum	1; 22	15.553	<0.001
Wetland grasses	1; 22	13.295	0.001
Ferns	1; 22	1.175	0.29
pH	1; 22	8.202	0.009
Total basal area	1; 22	0.017	0.896
Mean canopy age	1; 10	2.410	0.152
Units 1 & 3			
Sphagnum	1; 22	24.099	<0.001
Wetland grasses	1; 22	1.376	0.253
Ferns	1; 22	2.031	0.168
Sarracenia purpurea	1; 22	7.007	0.015
pH	1; 22	76.463	<0.001
Total basal area	1; 22	3.096	0.092
Mean canopy age	1; 10	100.568	<0.001

Table 3: Results of single factor ANOVAs between units. Significant values are $p < 0.05$.

Discussion

The data collected from the three units in Dulany Bog show possible trends of changes in community and habitat structure as woody species are allowed to invade a southern Appalachian bog due to alterations in natural disturbance and hydrologic regimes. These trends, however, may be the result of influences besides woody succession and other studies are needed to prove their validity. As the shrub canopy ages and becomes denser, the amount of sphagnum and other wetland species may decrease. The linear regression in Figure 5 shows a correlation between mean canopy age and sphagnum cover. Its intercept at 32.43 years may indicate the need for management efforts. Although it is unlikely that the sphagnum layer would be completely eradicated from the bog surface, the data suggest that its cover might be severely diminished.

The mean pH values of the subsurface water in the bog were much higher than expected. One would expect the pH to be strongly acidic, but the data prove otherwise. Decreased sphagnum cover and increased shading by woody invasives may account for the rise in the average pH of the units with older shrub canopies and less sphagnum cover. Restoration of high light intensities and increased sphagnum cover may help to create more acidic conditions. Also, erosion of mafic rocks upstream of the bog may contribute to the higher than expected pH values of the bog. Sources of amphibolite bedrock exist to the North and East of Dulany Bog and the bedrock beneath the bog itself also contains amphibolite (Robinson et al. 1992). When weathered, this rock releases calcium ions and raises the pH of surface runoff and ground water (Goforth pers. comm.).

One would expect the total basal area of woody species per square meter to increase as the shrub canopy's age increased. The data summarized in Figure 1, however, show that average basal area of *Alnus serrulata* and total basal area were highest in unit 2, and decreased in unit 3, which may have an older shrub canopy. This may be the result of the shrub layer being thinned out naturally to fewer, larger stems as time progresses. Alternatively, this apparent difference may be the result of sampling variation as the total basal areas of these two units are not significantly different.

Nutrient concentrations of subsurface water sampled from all three units proved to be very low and uniform across the three units. This could be the result of several possibilities: 1) there could be very little nutrients available to plants in the bog; 2) nearly all available nutrients have been taken up by plants within the community; or 3) the sampling method used in this study may be ineffective in measuring the nutrient availability and a different method, such as soil sampling, may give different results. Low nutrient concentrations, however, should be expected in bogs.

This study demonstrates the need for management practices in Dulany Bog to control the successional advancement of woody species and subsequent shading of the herbaceous layer. Management practices are needed to restore and maintain a more open bog community and promote the growth, spread, and diversity of herbaceous species in Dulany Bog. If nothing is done to control the shading of the herbaceous layer by *Alnus serrulata* and other invasive woody species, diversity of the herbaceous layer will likely be decreased and characteristic bog plants may not have enough light to promote flowering and seed set. Sphagnum cover may decrease, significantly altering the community structure.

Management Recommendations

Restoration practices to promote a more open bog community must not focus solely on removal of woody plants. Restoration of the bog's hydrology is equally, if not more, important than clearing woody species to restore higher light intensities to the herbaceous layer. If nothing is done to slow and spread the flow of water in the bog and maintain a high water table, current ditches and streams in the bog will become more channelized, creating drier conditions in the bog, which will support the invasion of woody species. Beaver activity may help to control woody invasion of the bog so their activities should be encouraged.

In addition to maintaining the hydrologic system within the bog, the watershed upstream of the area must be protected and managed to ensure a steady source of clean water into the bog. Sources of sediment, nutrient, and chemical runoff upstream must be controlled. Silt fences should be used to control sources of erosion, and pools should be constructed upstream of the bog to filter out sediment from the water.

Historically, fire has been a key element in bog management, including Dulany Bog, and its usefulness as a management tool should not be dismissed. Fire may be useful in removing woody species during the restoration phase of managing Dulany Bog as well as controlling re-sprouting in the maintenance stage. Fire may also be used to suppress the dominance of grass and sedge species, should these begin to choke out and shade herbaceous species. When properly used, prescribed burns can promote diversity of herbaceous species within the bog. However, conflicts with public opinion, visibility for motorists on NC 107, and the possibility of uncontrolled fire or difficulties in getting the fire to carry across the bog may render prescribed burning of Dulany Bog impractical.

Manually cutting and removing woody plants from the bog is likely impractical to be used as a large scale treatment for Dulany Bog. This method of treatment is very labor and time intensive, and even more so if herbicide is to be applied to each stem. Once areas of the bog are restored to a more open condition, they may still need to be maintained on a regular schedule to minimize woody invasion. In addition to being time consuming, this strategy requires a lot of traffic through the bog, which can trample vegetation and create drainage channels. Likewise, use of machinery to mow or cut the bog will damage the soil. Hand-held brush cutters or weed whackers would require a lot of traffic in the bog and may still prove to be time consuming.

Periodic flooding of the bog, either through beaver or human activity, may be effective if it is shown to kill the woody species invading Dulany Bog. This strategy could be less time consuming than some of the other methods and could treat large areas of the bog with little or no trampling of the vegetation which could also damage the bog soils and hydrology. As a restoration method, however, this would leave large amounts of dead, woody debris in the bog.

The best management strategy may involve a combination of several methods. For example, flooding the bog could be used to kill the woody plants which could later be burned to remove the woody debris. Alternatively, some of the alder could be cut and left to dry and aid in carrying fire across the bog. If fire would not carry well or be effective in the restoration process, it may still be an effective maintenance tool to prevent re-growth of the shrub layer.

Conclusion

Southern Appalachian bogs are unique wetland habitats that are becoming increasingly rare. Conservation efforts are needed to protect the remaining bog areas from further destruction and to mitigate the impacts of human activity. Due to the disruption of natural maintenance and disturbance regimes, active management strategies are needed to prevent the succession of bog communities into forested wetlands or other community types. This natural process is accelerated by human disturbances, especially those which create drier conditions in the bog. When no mechanism exists to control invasion by woody species, this successional process may take place in as little as 30 years from the time seedlings of invasive woody plants become established in the bog community, assuming favorable growing conditions.

Assuming that differences in vegetation between the three units is due solely to the effects of the progressively older shrub canopies, projections can be made about the future status of Dulany Bog if nothing is done to control the woody invasion of species like *Alnus serrulata*. The regression line in Figure 5 suggests a decline in sphagnum cover over time. Sphagnum cover is a key element of the bog system, and its decrease or elimination would mean drastic consequences for the natural bog communities in the area, as well as the maintenance of the bog's complex hydrologic system. Herbaceous species specifically adapted to live in the bog, such as *Helonias bullata* or *Sarracenia purpurea*, will be affected by both the decreased light concentrations beneath a dense shrub canopy and changes in the community structure associated with a decline in sphagnum cover.

References Cited

Cheshire Biodiversity. 1997. Lowland Raised Bog, [online]. Available: http://www.cheshire-biodiversity.org.uk/habitat-lpeatland.htm

Denhof, C., Hodges, M., and Ceska, J. 2004. Bog Management Recommendations: Tips for private landowners – Companion to the GPCA bog brochure, "Bogs of Georgia: Conservation Efforts of the Georgia Plant Conservation Alliance."

Denhof, C. 2000. Atlanta Botanical Garden: Restoration efforts on herbaceous bogs of the southeastern United States. Tipularia. 15: 2-5.

Eggelsmann, R. 1980. Hydrological aspects of peatland utilization and conservation in northwestern Germany. Proc. 6^{th} int. peat congress. 28-30.

Floyd, W. J. 1973. Vascular plants of the Dulany Bog tract. Report to Highlands Biological Station, Highlands, NC. 6 p.

Georgia Plant Conservation Alliance. 2004. Bogs of Georgia: Conservation efforts of the Georgia Plant Conservation Alliance.

Goforth, T. personal communication. 23 November 2004.

Gorham, E., & Rochefort, L. 2003. Peatland restoration: A brief assessment with special reference to *Sphagnum* bogs. Wetlands Ecology and Management. 11: 109-119.

Maltby, E., Legg, C. J., and Proctor, M. C. F. 1990. The ecology of severe moorland fire on the North York Moors: Effects of the 1976 fires, and subsequent surface and vegetation development. Journal of Ecology. 78 (2): 490-518.

McDowell, G. W. 1973. Notes and news. Castanea. 38: 407-408.

Norton, D. A. and De Lange, J. 2003. Fire and vegetation in a temperate peat bog: Implications for the management of threatened species. Conservation Biology. 17 (1).

Ottawa Field-Naturalists' Club. 2002. We Can Save Alfred Bog, [online]. Available: http://www.ofnc.ca/conservation/alfredbog/alfredbog.html

Pickett, S.T.A. 1989. Space-for-time substitution as an alternative to long-term studies. *In* Long term studies in ecology: approaches and alternatives. *Edited by* G.E. Likens. Springer Verlag, New York. pp. 110-135.

Riggsbee, A. 1998. "Botanical and avifaunal survey and conservation management suggestions for a southern appalachian bog ecosystem." Diss. Western Carolina U., Cullowhee, NC.

Robinson, G. R., Jr., Lesure, F. G., Marlow, J. I. II, Foley, N. K., and Clark, S. H. 1992. Bedrock geology and mineral resources of the Knoxville 1° x 2° quadrangle, Tennessee, North Carolina, and South Carolina. U. S. Geological Survey Bulletin. 1979.

Sargent, M.S and Carter, K.S., ed. 1999. Managing Michigan Wildlife: A Landowners Guide. Michigan United Conservation Clubs, East Lansing, MI. 297pp. Available http://www.dnr.state.mi.us/publications/pdfs/huntingwildlifehabitat/Landowners_Guide/Habitat_Mgmt/Wetland/Bogs_Fens.htm.

Schafale, M., Kauffman, G., Wyatt, R., Wykle, B., McKeage, A., et al. (1999). Dulany Bog: Natural communities and management issues. Report to Highlands Biological Station, Highlands, NC.

Schafale, M. P., and Weakley, A. S. 1990. Classification of the Natural Communities of North Carolina, Third Approximation. North Carolina Natural Heritage Program Division of Parks and Recreation NC Dept. of Environment, Health, and Natural Resources. 325 pp. 1990.

Sherpa Guides. 2004. [online]. Available:
http://sherpaguides.com/north_carolina/mountains/natural_history/mountain_bogs.html

Slattery, B. E. and Kesselheim, A. S. 1995. WOW! The Wonders of Wetlands: An educator's guide. Bozeman, MT: Environmental Concern and The Watercourse.

Sokal, R. R. and Rohlf, F. J. 1995. Biometry. 3rd ed. New York: W. H. Freeman and Co.

Sutter, R. D. 1984. The status of *Helonias bullata* L. (Liliaceae) in the southern Appalachians. Castanea. 49: 9-16.

U.S. Fish and Wildlife Service. 1997. Status and trends of wetlands in the conterminous United States. United States Department of the Interior.

Van Breemen, N. 1995. How *Sphagnum* bogs down other plants. TREE. 7: 270-275.

Whelan, R. J. 1995. The ecology of fire. Cambridge University Press, Cambridge.

Wykle, Bill. Highlands Biological Station. personal communication.

Mowing and its Effect on the Wildflowers of Horse Cove Road on the Highlands Plateau

Megan Mailloux

Introduction

Roadside wildflowers are an important component of the biodiversity of the Highlands Plateau. They are what many tourists observe first as they arrive into the area. Roadsides provide a unique habitat to a vast diversity of flora, which in turn may be the only locale for some to survive. During the summertime when about 15,000 seasonal residents and tourists come to stay in the area, their experience is enhanced by the gorgeous display of wildflowers along the roadsides (Carla 2004). As stated by Glenda Zahner, one local resident, "the Chamber of Commerce needs to know that lots of people used to drive up to Highlands just to see the roadside wildflowers."

A great number of roadside wildflowers exist in the Highlands area, but the populations may be declining due to the maintenance regime Department of Transporation (DOT) crews have implemented to care for local roadsides (Tucker 2004). Nearly all of the roads in the Highlands area are contracted by the NC DOT based in Franklin. The roads are mowed with a batwing mower, five times a year, and once during the winter with a contour mower (Wallace 2004). Local residents have raised concerns with the NC DOT about the timing and frequency of their mowing operations. For example, Glenda Zahner, a citizen of Highlands stated that "Every summer, the week before July 4th, the DOT sometimes mows down a gorgeous natural display of evening primrose, blue spiderwort, purple penstemon, white fawn's breath, red fire pinks, fleabane, daisies, etc." She also noted that part of the issue is "that they seem to want to apply the same formula and schedule for mowing the grassy roadsides in Franklin as for the rocky road-cuts along the Cullasaja Gorge road, the Walhalla road, the Cashiers road and the Dillard road, where many wildflowers abound." Residents would like to work with the DOT to try to rectify this situation.

An understanding of roadside ecology is important for implementing an ecologically sensitive mowing regime. Ecologists tend to see more generalist species or species that are better able to cope with differing environments, rather than specialist or area-sensitive species along roadsides. Most plants found along roads require high light and cannot tolerate the shade created by competing species. Most are considered native “weeds” and/or invasive species and they are “the kinds of plants that spring up in almost any disturbed spot of bare earth (Sperling 2003)." For instance, a common invasive such as *Lonicera japonica* or Japanese Honeysuckle, which was first introduced into the United States in 1806, is commonly distributed along edge associations such as roadsides in North America (Merriam 2003). Invasives and non-natives occur on roadsides mainly because they are transported by people, perhaps on car bumpers as people drive by. They can also be transported on clothing and animal fur. A good example of this type of distribution is the sticky seeds of *Desmodium nudiflorum*, a common roadside plant in the Highlands area. The seeds of many invasives can also remain dormant in the soil until the right conditions arise for growing, creating a greater diversity in the populations along roadsides. Frequently disturbed landscapes are more susceptible to invasion from exotics (Sperling 2003). All of this information it appears as though non-native species are not good for our roadsides, but some of our most-liked wildflowers are non-natives, such as clover and dandelion. It is hard to say which plants are of more importance than others in this respect.

Roadsides are prime areas for succession. Fast growing, sun-loving species are able to grow there first, followed by perennials and then woody species. Native plant communities may be less diverse because they are inhibited by roadside conditions, but invasive exotic species can contribute to higher species diversity than before because they are more tolerant of such conditions (Sperling 2003). The advantages of natives are that they contribute to erosion control, are already adapted to the climate and geology that roads cross, can prosper without care, and can ward off weeds and invasive pests. Natives are normally the most favored species along roadsides.

Maintenance of roadsides helps to determine species diversity along roadsides and one of the most common means of maintenance along roadsides is mowing. As Sperling (2003) stated in *Road Ecology*, “the prime purpose of mowing along roads is safety.” Too much vegetation can obstruct the view of drivers, as well as hide pedestrians or animals from direct sight. A clear space along the roads is required as a pullover location for cars that run into trouble during travel. Overall, roadside maintenance requires important decisions about time, space and intensity. Each habitat may require a unique care practice, as all environments do not have the same ecology or. The timing and frequency of mowing is very important. The frequency of mowing has major ecological effects as well as cost considerations. Not only does less mowing mean less cost, but mowing in general also promotes greater species richness (Sperling 2003).

Mowing has an effect on the flora of roadsides because it promotes growth of herbaceous species and limits encroachment of woody species. It essentially prevents woody species from maturing (Schippers 2001). Therefore, common and invasive species found along roadsides may benefit "in the face of repeated disturbances such as mowing" (Sperling 2003). Disturbances such as mowing can be considered one of the most important factors shaping roadside plant communities (Schippers and Joenje 2001). Few studies have been done to analyze the effects of mowing on plant communities and some of them seem to present conflicting results on the question. In a study by Maron (2001), it was found that mowing can result in an increase in species richness, an increase in the relative abundance of perennials, and a reduction in exotic species biomass. On the other hand, Neigerbauer (2000) showed that mowing directly reduces the depth of rooting of perennial herbs, such as *Rudbeckia hirta* L., indicating that mowing has a negative effect on the growth of perennials. Given that mowing must be done for safety purposes, this poses the question as to how often a roadside should be mowed and how it should be done to have the least effect on roadside flora.

The concern over maintenance along roadsides in Highlands is a credible concern because roadsides are the only available habitat for some flora to survive, because much of the other forested land and natural areas in Highlands are disturbance-free areas (with restrictions on burning, timbering, etc.). Many of these wildflowers do not grow anywhere else around Highlands so the roadsides are the only place where they can be seen and enjoyed. They provide a stunning show during the spring and summer. Along roadsides, or where there is continued disturbance, species diversity is much higher creating a more diverse visual display. More studies should be done to understand the effects of mowing on these distinctive populations in the Highlands Plateau. This area of study is significant for Highlands due to the high number of tourists who spend time in the area during the summer, when most wildflowers are in peak bloom. It is also important to know which wildflowers are present in the area in order to know what can be done to preserve their populations, and maintain diversity.

In this paper I will attempt to understand how mowing affects the vegetation along roadsides in Highlands, in particular along Horse Cove Road. According to the NC DOT website about 240 people use Horse Cove Road in any given 24-hour period. Thus, these particular flowers are viewed by many people. I will look at whether mowing increases or decreases the native wildflower populations. I will attempt to answer the following questions: Does mowing foster an increase of the cover or frequency of exotics? Does it inhibit the growth of woody species? When and with what intensity should the DOT mow the roadsides of Highlands in order to encourage the growth of more native wildflowers?

Materials and Methods

My study area lies in the town of Highlands, North Carolina, in the heart of the Southern Appalachian Mountains. The climate is a temperate rainforest, averaging over 87 inches of rain per year. Transects were located along Horse Cove Road (See Figure 1). This road winds down the southwestern slope of the Highlands Plateau, dropping in elevation from approximately 4000 feet to approximately 3000 feet.

The vegetation along the side of the road occurs on a slope. Transects were chosen according to their accessibility and appearance. Areas with excessive boulders or rhododendron were excluded, because areas harbored few wildflowers. All transects were established along a safe and workable section of the road. They also exhibited a distinct difference between unmowed upper portions versus mowed lower portions, being divided by what I call a visible mow-line. The mow-line is a visible drag-mark about half way up the slope indicating the upper edge of the mower. The area below this line gets mowed five times per year, whereas the area above it only gets mowed once during the winter. The transects were labeled HC1-HC4. At each transect, a GPS reading was taken to identify site locations using a hand-held Garmin Personal Navigator GPS III Unit (Appendix I, Table 1).

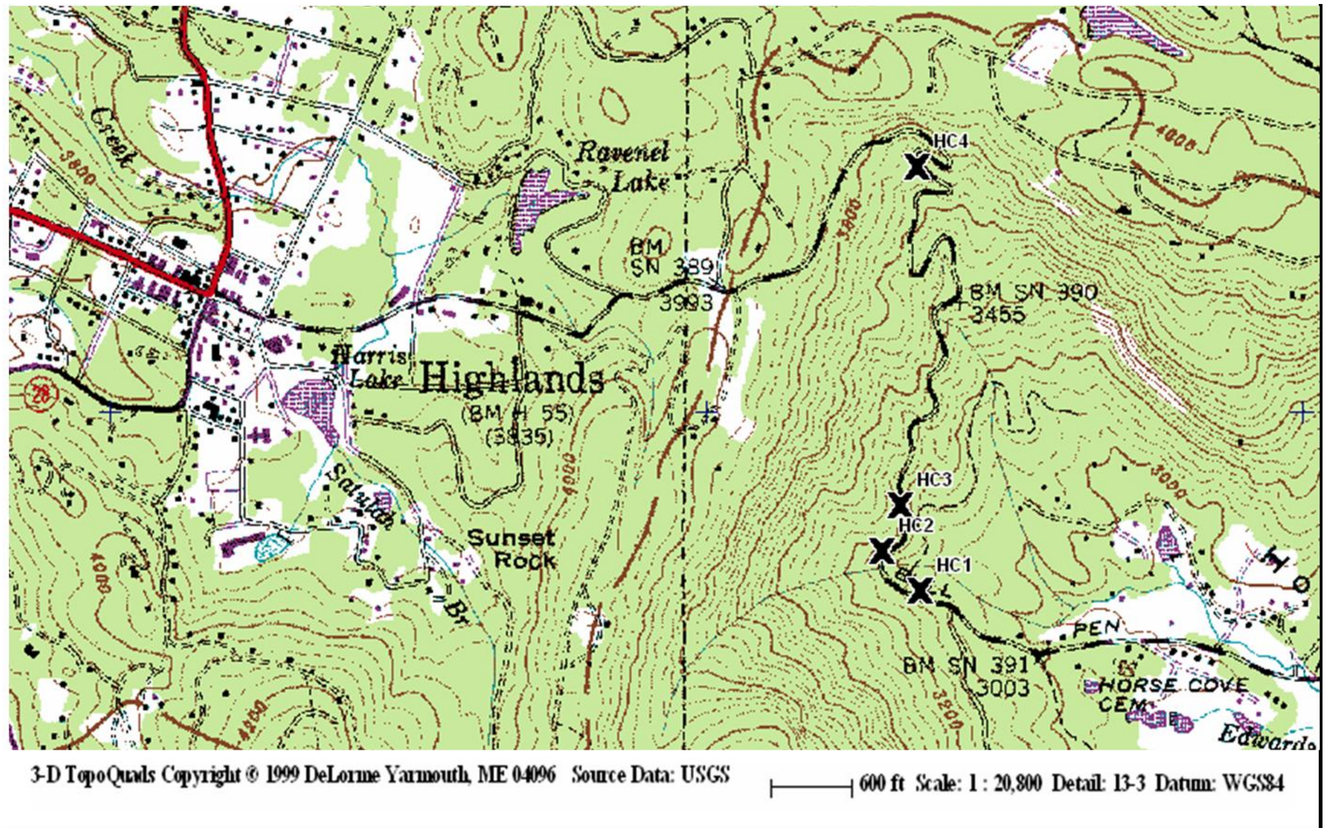

Figure 1: Map of Horse Cove Road in Highlands NC. The location of study transects are marked with an X.

Every 2m, two paired 1m x 1m plots were established perpendicular to the transect. (See Figure 2). One plot was created below the visible mow-line and the other above the visible mow-line (See Figure 3). Each plot was labeled in numerical order according to the transect in which it was located: the lower mowed plot was labeled A and the top unmowed plot, B. For example, HC1-1A was the first lower or mowed plot in transect HC1. The plots were arranged in this manner in order to observe differences between plant populations and cover differences of mowed versus unmowed plots. In total 42 plots were established along the roadside, 21 mowed plots and 21 unmowed plots.

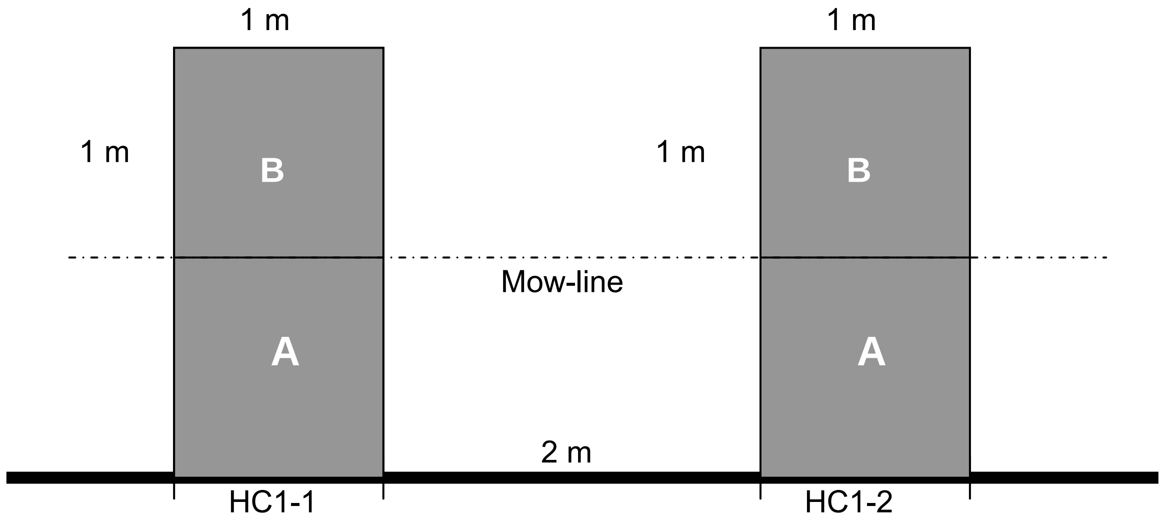

Figure 2: Plot arrangement along transect.

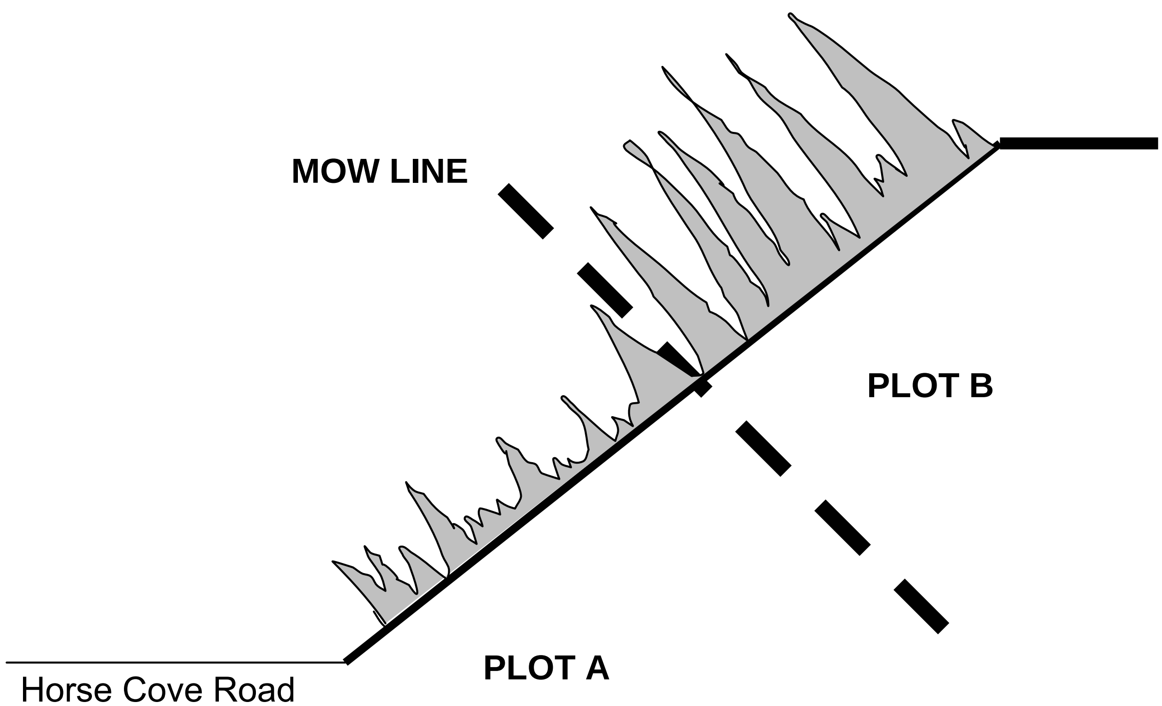

Figure 3: Crop section of plots along Horse Cove Road, mowed versus unmowed portions.

Within each plot, species presence was recorded along with cover classes for each species, leaf-litter cover, and bare ground cover. Cover was assessed visually and assigned a score as follows: trace=1, 0-1% = 2, 1-2% = 3, 2-5%= 4, 5-10%= 5, 10-25%=6, 25-50 % = 7, 50-75%=8, 75-95%=9, and 95-100%=10. If the species was unable to be identified at the time of data collection, it was collected, assigned a placeholder name, and pressed for future identification.

After all plants were identified, data were entered into a Microsoft Excel spreadsheet. A full species list was created with using names from the *Manual of the Vascular Flora of the Carolinas* (Radford et.al, 1968). Individual species were classified into the following groups: native or invasive; wildflowers, and woody species. These were determined according to *Gray's Manual of Botany* (Fernald 1950).

The data were then entered into the SAS program, and statistics tests were run. The ANOVA proc GLM and Levene's Test for Homogeneity of variances (SAS Institute 2002) tests were run on all species to determine if there were significant differences in percent cover between mowed and unmowed plots. Tests were also run to determine if there were significant differences in species richness, species evenness, and species diversity differences between mowed and unmowed plots. Lastly, species were placed into three groups, wildflowers, woody species, and invasives. Categories were determined using Radford et. al. (1968). Wildflowers were set apart to help determine the effect that mowing has on them. Woody species were set apart because mowing tends to inhibit their growth, and invasives were grouped because they favor more light. A comparison of flowering times for all wildflower species found along Horse Cove Road was also done.

Results

Average cover per plot for each species was calculated for both mowed and unmowed plot types (Appendix Table 2). Only six individual species showed statistically significant differences between plot types (Figure 4). The two ferns (*Thelypteris noveboracensis (L.) Nieuwland , Polystichum acrostichoides (Michaux) Schott),* and one woody species *Rhododendron maximum L.)* showed higher percent cover in unmowed plots. Three invasive exotic species (*Microstegium vimineum (Trinius) A. Camus, Oxalis acetosella L., Potentilla canadensis L.*) showed a higher cover in mowed plots. No wildflowers had a significant difference in cover between mowed and unmowed plots. Nevertheless, when all of the wildflower species from all plots were grouped together and analyzed, they showed a significant difference between mowed and unmowed plots (Figure 5). Wildflowers showed greater cover in unmowed plots (16% cover) than in mowed plots (7% cover). The same results appeared for Woody species and Invasive/ Native "weedy" species (Figure 5). Woody species also had significantly higher cover in the unmowed plots. Invasives showed the opposite trend and had much higher cover in the mowed (19%) versus unmowed (8%) plots. All results for the Levene's Test for homogeneity of variances showed that there were no significant differences between the variances of the treatment groups.

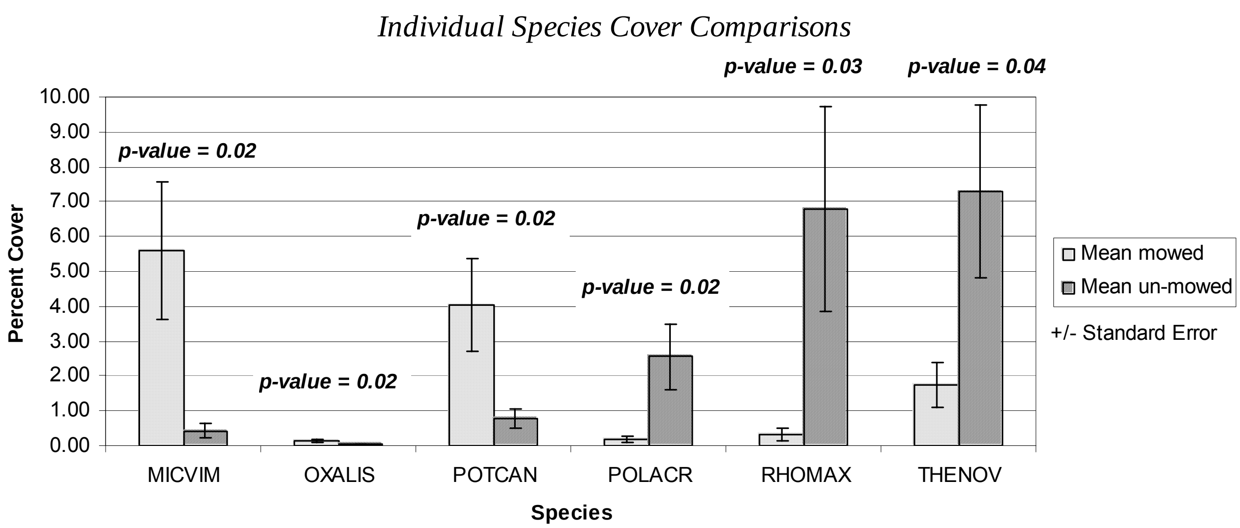

Figure 4: Individual cover comparisons between mowed and unmowed plots for six species. MICVIM is *Microstegium vimineum (Trinus) A. Camus*, OXALIS is *Oxalis spp,* POLACR is *Polystichum acrostichoides (Michaux) Schott*, POTCAN is *Potentilla Canadensis L.*, RHOMAX is *Rhododendron maximum L.*, and THENOV is *Thelypteris noveboracensis (L.) Nieuwland.* P-values were taken from SAS proc GLM analysis and were considered statistically significant if they had a value less than 0.05.

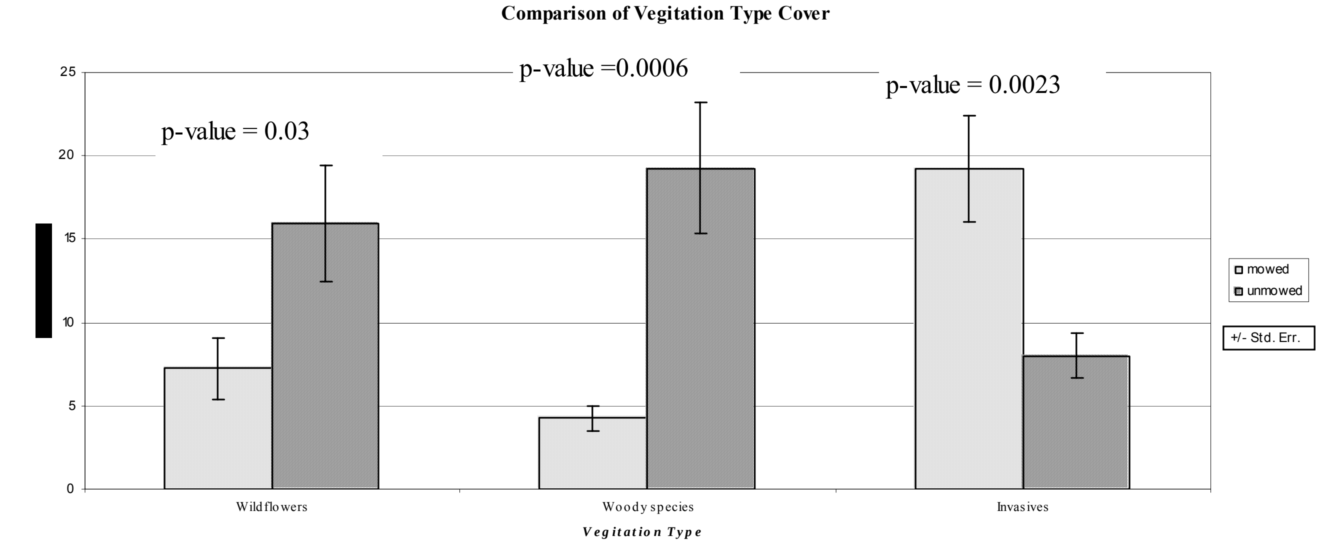

Figure 5: Plant Type Cover Comparisons Significant P-values for Plant Type Group comparisons of mowed versus unmowed plots. P-values were taken from SAS proc GLM analysis and were considered statistically significant if they had a value less than 0.05.

Similarly, I calculated the average species richness (14 species per 1m x 1m plot), species evenness (Simpson's Index) and species diversity (Shannon-Weaver Index) for each plot type (Table 1). There was no significant difference between mowed and unmowed plots for any of the measures of diversity.

	Mean Mowed	**Std. Err. Mowed**	**Mean Unmowed**	**Std. Err. Unmowed**
Evenness	0.1953784	0.0202454	0.2182672	0.0208085
Diversity	2.0656844	0.0750754	1.9723414	0.0739545
Richness	14.7142857	0.4886466	14.1904762	0.5674464

Table 1: Mean values for species evenness (Simpson's Index), species diversity (Shannon-Weaver Index), and species richness.

Figure 6 was shows bloom times of the wildflower species found in the plots along Horse Cove Road. Most of the wildflower species bloom during the summer months between April and October. In Figure 6, all of the species in the wildflower group were listed and bloom times for each species were recorded according to the *Manual of the Vascular Flora of the Carolinas* and data collected by Bill Wykle for flowering times of all species in the Highlands Biological Station Botanical Garden. The area between the dotted lines shows the time of year when most wildflowers are blooming along Horse Cove Road.

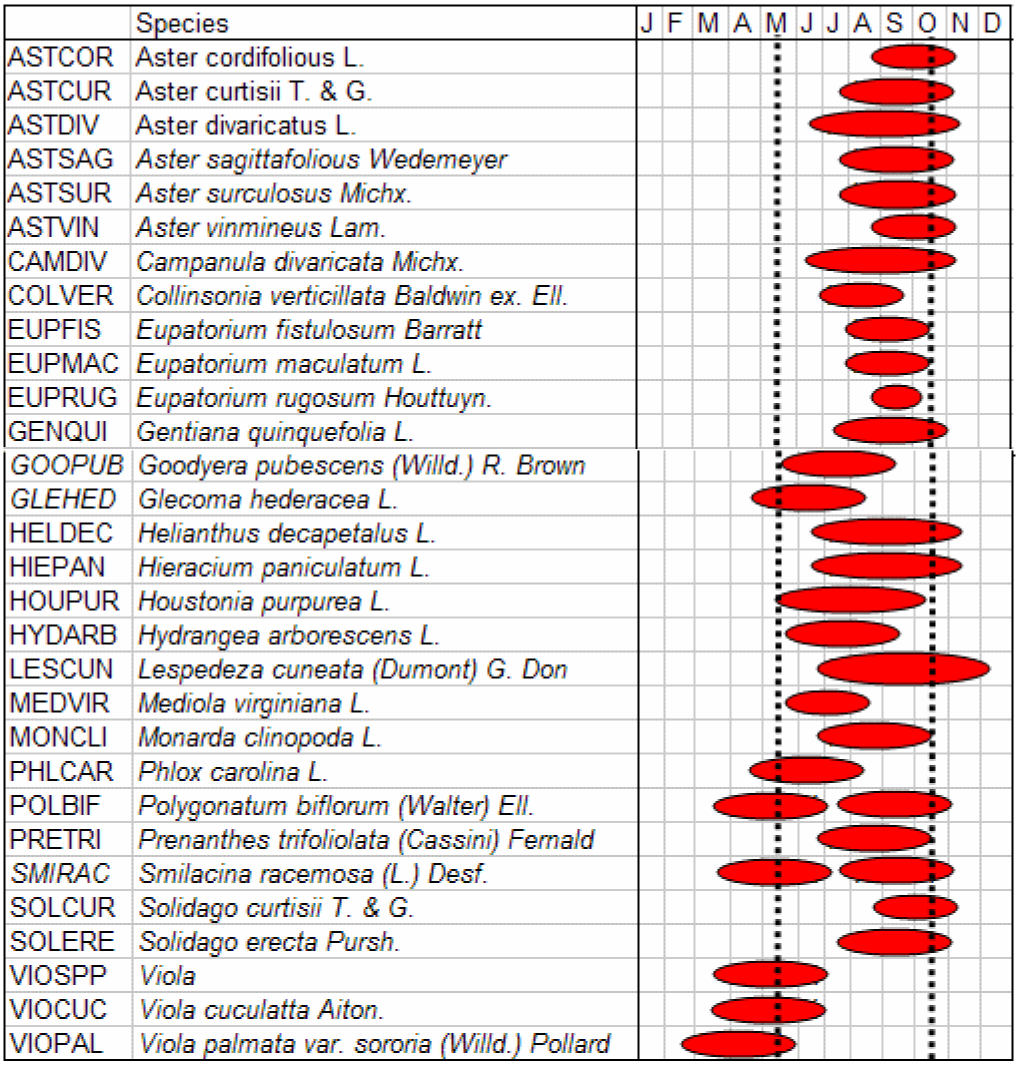

Figure 6: Blooming times of wildflowers along Horse Cove Road. Shaded ovals represent the average flowering times for each species. The area between the dotted lines represents the time when most wildflowers are in bloom.

Discussion

The appearance and maintenance of roadside flora has been an important issue in the United States over the last century. With approximately 6.2 million kilometers of public roads, a figure not including the extent of private roads (Forman and Alexander 1998), there is a need for concern over their appearance. In the 1930s, it was declared that "roadsides should be maintained as if they are our nation's front yards (Sperling 2003)." Roadsides give visitors their important first impression of our state, thus it is important to maintain them in such a way that promotes a favorable impression. For instance Lady Bird Johnson, the former first lady, promoted roadside beautification and set in motion a series of efforts which culminated in the Highway Beautification Act of 1965. Since the 1960s, road beautification has been a prominent goal of the government (Sperling 2003).

Mowing is one of the main ways in which we have some control over the appearances of our roadsides, but few studies have been done to determine what the effect of mowing is on the flora along roads. As many residents of Highlands have expressed concern about diminishing populations of wildflowers along the roadsides in the area, I hope that this study will help provide evidence that the mowing regime that has been implemented in past years should be changed to encourage more beautiful wildflower populations to thrive along the roadsides.

My study shows convincing evidence that mowing does indeed have an effect on the flora along Horse Cove Road. Frequent mowing contributes to a greater number of invasive exotic or native "weeds." The frequency of mowing currently done by the NC DOT also appears to limit the numbers of wildflowers found along Horse Cove Road. There is less wildflower species cover found in the lower, more frequently mowed plots than in the upper unmowed plots (Figure 5) . It encourages an increase of invasive cover in the lower, more frequently mowed plots. Some of the invasives found are ones that can quickly spread, such as *Lonicera japonica, Prunella vulgaris*, and *Microstegium vimineum.* In particular, *Lonicera japonica* was the fastest spreading invasive out of six in a study by Merriam (2003). The spread of these species into surrounding woodland may be detrimental to native forest plant communities as they take over habitats and resources. Mowing also inhibits the encroachment of woody species, which can be a favorable, as woody species grow larger and tend to limit visibility. Woody species were limited even in plots that were only mowed once per year, suggesting that frequent mowing is not needed to control them. Overall, less frequent mowing along the roads would most likely foster a greater diversity and frequency of wildflowers.

In turn, mowing along roadsides is necessary for the safety of drivers and promotes greater visibility. Thus, it is suggested that instead of mowing five times per year, perhaps roads could be mowed twice, once in spring and once in fall. This should allow more wildflowers species to grow and bloom during the summer months when there are more tourists in Highlands. More wildflowers should be capable of flowering if mowing took place only during the periods before April and after September. In this way wildflower populations would be able to prosper, yet sufficient mowing could still take place, just less frequently.

Further research should be done to determine the effects of mowing throughout the seasons. If a study was pursued starting at the beginning of the flowering time, such as during the month of March, then these effects could be observed in their entirety throughout the year. For instance, in the summer there may be a different set of species prominent along the roadside that were not apparent during this fall sampling period. During a year-long time frame, exact frequencies and intensities of mowing can be recorded as well. Furthermore, a more detailed survey could be conducted, including an examination of the slope between the plots and what effect, if any, this might have on the types of species along the roadsides. There are many factors that have been overlooked for in this study, including whether there was a difference in the moisture gradient between mowed and unmowed plots. If there is a difference, then this could have caused a difference in plant species between the two plots.

In conclusion, by encouraging a different, less frequent and intense mowing regime to be implemented by the NC DOT, the wildflower populations would likely benefit. A collaboration between the NC DOT and residents of the Highlands Plateau would be necessary for this to be successful. A regime like this would in turn beautify the roadsides around Highlands and contribute to the charm of the area for both tourists and residents alike. It would also lighten the workload of the NC DOT and cut labor costs. Invasive and weedy species would be less likely to invade vulnerable roadsides and abundant native wildflowers would offer travelers a more scenic driving experience.

References Cited

Radford, A.E., H.E. Ahles, and , C. R. Bell. 1968. *Manual of the Vascular Flora of the Carolinas*. The University of North Carolina Press. Chapel Hill, North Carolina.

Fernald, M.L.1950. Gray's Manual of Botany, Eighth Edition. American Book Company, New York.

Forman, R.T.T. and L.E. Alexander. 1998. "Roads and their ecological effects." Annual Review of Ecology and Systematics. 29: 207-231.

Maron J.L and Jefferies R.L. 2001. Restoring enriched grasslands: effects of mowing on species richness, productivity, and nitrogen retention. Ecological Applications. 11(4): 1088-1100.

Merriam, R. W. 2003. The abundance, distribution and edge associations of six non-indigenous, harmful plants across North Carolina. Journal of the Torrey Botanical Society. 130 (4): 283-290.

Neigebauer, A. L. 2000. Shoot and root characterization of Rudbekia hirta L. mowed at different heights. Hortscience 35(7): 1247-1248.

SAS Institute. 2000. Cary NC.

Sperling, Forman et. al. 2003. Road Ecology: Science and Solutions. Island Press, Washington DC.

Spellerberg, I. F. 2002. Ecological Effects of Roads. Science Publishers, Inc. Enfield, NH.

Schippers, P. and W. Joenje. 2001. Modeling the effect of fertilizer, mowing, disturbance and width of the biodiversity of plant communities of field boundaries. 2001. Agriculture, Ecosystems and the Environment. 351-365.

Tucker, Robert. 2004. Personal communication.

Wallace, Jim. 2004. Personal communication.

www.ncdot.org/planning/tbp/gis/DataDist/GISTrafSurvMaps.html. 2004. North Carolina Department of Transportation.

Zahner, Glenda. 2004. Personal communication.

Appendix

Table 1: Latitude and longitude for the starting point of each transect along Horse Cove Road

PLOT	Latitude and Longitude
HC1	N 35.04591; W 083.17735
HC2	N 35.04683; W 083.17840
HC3	N 35.04788; W 083.17789
HC4	N 35.05566; W 083.17723

Table 2: Mean cover values for all species found along Horse Cove Road for both mowed and unmowed plots. Plots where species did not occur received a zero cover value and these were averaged into the mean. Mean cover values, standard error (SE), and P-values were calculated with SAS proc GLM.

SPECIES NAME	Mean mowed	SE Mowed	Mean un-mowed	SE Unmowed	P-value
Acer rubrum L.	0.76	0.28	1.17	0.44	0.44
Agrostis hyemalis (Walter) BSP.	0.05	0.03	0.00	0.00	0.15
Alternate blade grass	0.02	0.02	0.00	0.00	0.32
Alternate leaf	0.00	0.00	0.02	0.02	0.32
Amphicarpa bracteata (L.) Fernald	2.55	1.13	2.64	1.16	0.95
Aster cordifolius L.	0.00	0.00	2.02	1.78	0.26
Aster curtisii T. & G.	0.26	0.18	2.50	1.77	0.22
Aster divaricatus L.	0.58	0.19	2.64	1.79	0.26
Aster sagittifolius Wedem. ex Willld.	0.00	0.00	0.07	0.07	0.32
Aster surculosus Michaux	0.00	0.02	0.02	0.00	0.32
Aster vinmineus Lam.	0.02	0.02	0.17	0.17	0.39
Betula lenta L. (seedling)	0.15	0.08	0.10	0.07	0.63
Botrychium dissectum Sprengel	0.00	0.00	0.08	0.07	0.29
Campanula divaricata Michx.	0.05	0.03	0.07	0.07	0.76
Carya glabra (Mill.)	0.19	0.17	0.02	0.02	0.33
Chimaphila maculata Pursh.	0.02	0.02	0.05	0.03	0.56
Collinsonia verticillata Baldwin ex Ell.	0.50	0.27	0.07	0.07	0.14
Cysopteris protrusa (Weatherby) Blasdell	0.02	0.02	0.00	0.00	0.32
Dark Green Trifoliate	0.36	0.36	0.17	0.17	0.63
Dennstaedtia punctilobula (Michaux) Moore	0.83	0.83	1.79	1.79	0.63
Desmodium nudiflorum (L.) DC.	4.71	1.90	3.10	0.99	0.45
Desmodium paniculatum (L.) DC.	0.05	0.03	0.17	0.17	0.49

SPECIES NAME	Mean mowed	SE Mowed	Mean un-mowed	SE Unmowed	P-value
Panicum boscii Poiret	0.69	0.07	0.20	0.00	0.22
Panicum dichotomum L.	0.98	0.07	0.48	0.00	0.29
Dioscorea villosa L.	0.08	2.02	0.00	1.83	0.29
Drooping grass	0.07	0.02	0.00	0.00	0.32
Erigeron pulchellus Michaux	6.31	0.00	7.69	0.02	0.62
Eupatorium fistulosum Barratt	0.02	0.00	0.00	0.23	0.32
Eupatorium maculatum L.	0.00	0.17	0.02	0.07	0.32
Eupatorium rugosum Houttuyn.	0.00	0.00	0.33	0.07	0.15
Festuca elatior L.	0.19	0.00	0.07	0.36	0.52
Fraxinus americana L.	0.00	0.02	0.07	0.03	0.32
Gaylussacia ursina (M.A. Curtis) T.&G. ex Gray	0.00	0.00	0.38	0.17	0.29
Gentiana quinquefolia L.	0.02	0.02	0.05	0.00	0.56
Gillenia trifoliata (L.) Moench.	0.00	0.17	0.17	0.07	0.32
Glecoma hederacea L.	0.02	0.03	0.00	0.00	0.32
Goodyera pubescens (Willd.) R. Brown	0.17	0.17	0.08	0.07	0.62
Grassy grass	0.05	0.23	0.00	0.39	0.15
Hamamelis virginiana L.	0.33	0.03	0.52	0.07	0.67
Halesia carolina L.	0.17	0.36	0.07	1.78	0.60
Helianthus decapetalus L.	0.05	0.17	0.07	0.17	0.76
Hieracium paniculatum L.	0.40	0.02	2.19	0.89	0.33
Houstonia purpurea L.	0.31	0.02	0.40	0.00	0.70
Hydrangea arborescens L.	0.02	0.00	1.33	0.17	0.15
Lespedeza cuneata (Dumont) G. Don	0.02	0.23	0.00	0.52	0.32
Leucothoe axillaris (Lam.) D. Don	0.00	0.89	0.17	0.52	0.32
Liriodendron tulipifera L.	0.57	0.00	1.14	0.39	0.32
Lonicera japonicaThunberg	1.86	0.17	1.10	0.49	0.44
Magnolia fraseri Walter	0.00	1.97	0.60	0.19	0.13
Mediola virginiana L.	0.19	0.36	0.74	0.17	0.30
Microstegium vimineum (Trinus) A. Camus	5.60	0.00	0.43	0.02	0.01
Minty chordate	0.36	0.13	0.17	0.50	0.63
Mitchella repens L.	0.00	0.17	0.02	0.17	0.32
Monarda clinopoda L.	0.55	0.00	1.45	0.07	0.09
Narrow leaf grass w/ aruncles	0.17	0.02	0.17	0.00	1.00
Nyssa sylvatica Marshall	0.00	0.00	0.07	0.02	0.32
Opposite ovate	0.02	0.05	0.00	0.02	0.32
Ovate Arrowhead	0.00	0.00	0.02	0.83	0.32
Oxalis acetosella L.	0.15	0.39	0.02	0.10	0.02

SPECIES NAME	Mean mowed	SE Mowed	Mean un-mowed	SE Unmowed	P-value
Oxydenrum arboreum (L.) DC.	0.00	0.40	0.83	0.23	0.32
Parthenocissus quinquefolia (L.) Planchon	0.45	0.23	0.07	0.07	0.13
Phlox carolina L.	0.00	0.00	0.05	0.03	0.15
Pinnate venation	0.00	0.07	0.00	0.02	0.00
Pinus strobus L.	0.10	0.00	0.02	0.00	0.36
Polygonatum biflorum (Walter) Ell.	0.07	0.10	0.71	0.95	0.13
Polystichum acrostichoides (Michaux) Schott	0.19	0.07	2.55	0.41	0.02
Potentilla canadensis L.	4.02	1.32	0.79	0.27	0.02
Prenanthes trifoliolata (Cassini) Fernald	0.26	0.18	0.05	0.03	0.24
Prunella vulgaris L.	1.95	0.84	0.76	0.39	0.21
Quercus alba L.	0.14	0.10	0.07	0.07	0.56
Quercus prinus L.	0.48	0.19	0.33	0.19	0.60
Quercus rubra L.	1.02	0.38	1.93	0.88	0.35
Rhododendron catawbiense Michaux	0.00	0.00	0.83	0.83	0.32
Rhododendron maximum L.	0.31	0.19	6.79	2.94	0.03
Rhododendron minus Michaux	0.17	0.17	3.33	1.95	0.11
Robinia pseudo-acacia L.	0.00	0.00	0.52	0.27	0.06
Round leaf	0.00	0.00	0.02	0.02	0.32
Rubus cunefolius Pursh	0.81	0.27	1.76	0.93	0.33
Seeding spp?	0.00	0.00	0.00	0.00	0.00
Small green leaf	0.00	0.07	0.07	0.00	0.32
Smilacina racemosa (L.) Desf.	0.07	0.00	0.00	0.00	0.32
Smilax glauca (L.) Desf.	0.00	0.07	0.00	0.00	1.00
Snowy plant	1.79	1.79	0.00	0.00	0.32
Solidago curtisii T. & G.	3.17	1.76	1.76	0.44	0.44
Solidago erecta Pursh.	0.38	0.19	0.36	0.19	0.93
Tall grass	0.00	0.00	0.07	0.07	0.32
Thelypteris noveboracensis (L.) Nieuwland	1.74	0.65	7.29	2.48	0.04
Thick grass	0.17	0.17	0.00	0.00	0.32
Thin-leaf grass	0.07	0.07	0.00	0.00	0.32
Tsuga canadensis (L.) Carr.	0.19	0.10	1.10	0.44	0.05
Tsuga caroliniana Englem.	0.17	0.17	0.00	0.00	0.32
very thin grass	0.00	0.00	0.07	0.07	0.32
Viburnum prunifolium L.	0.02	0.02	0.00	0.00	0.32
Viola	0.05	0.03	0.07	0.04	0.64
Viola cuculatta Aiton.	0.00	0.00	0.07	0.07	0.32
Viola palmata var. sororia (Willd.) Pollard	0.62	0.23	0.19	0.08	0.08

SPECIES NAME	Mean mowed	SE Mowed	Mean un-mowed	SE Unmowed	P-value
Vitis baileyana Munson	0.00	0.00	1.00	0.84	0.24
Wide leaf grass	0.00	0.00	0.00	0.00	0.00
Degrees of Freedom for the GLM procedure is 21					

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