INTRODUCTION

The annual IE-HFS closing celebration that caps our students' semester-long southern Appalachian immersion is always bittersweet. Taking the pulse of the Plateau and beyond, they worked diligently not so much toward this culminating event of the semester, but toward a lifetime of discovering the unfamiliar in the familiar. Seeing the students assembled for their presentations brought James Lipton's minor classic *An Exultation of Larks* to mind. Lipton coined apt terms for groups of various kinds: an 'exultation' of larks, a 'parliament' of owls, a 'sneer' of butlers, a 'conglomerate' of geologists, and so on. Adding to the Lipton Lexicon, thinking about this group, shall we call them an 'anxious' of students? Soon to become a 'relief' of students?

* * *

Highlands Biological Station is a founding Field Site of the Institute for the Environment, a research and educational center of UNC Chapel Hill that began as the Carolina Environmental Program in 2001. Why here? It's clear: HBS is all about place-based teaching, learning, and research, which means we're all about immersion — creeks & coves, ridgetops & rivers, woodlands & wetlands, and from the Smokies to the Escarpment and beyond, our students have been immersed in this fabulous landscape classroom and laboratory.

A place-based program puts us in a mind to muse on just where the field site is...
We are sitting in a remarkable place — at 4,000 feet perched on the edge of the Blue Ridge Escarpment, straddling the continental divide — the Mississippi River basin where we sit — via the Cullasaja, Little Tennessee, Tennessee, and Ohio, and the Savannah River basin just up Horse Cove — via the Chattooga and Tugaloo Rivers. Ours is a waterworld, 2nd wettest region in North America, feeding headwaters of several river systems found in a modest radius of this place: Chattooga, Cullasaja, Tuckasegee, Horsepasture, Whitewater, Thompson...

We are fortunate that we live in the midst of a vast, reasonably intact and healthy forest ecosystem teeming with plant and animal life. And yet, for all that, we can't help but lament what has been lost. An opportunity was missed almost exactly 116 years ago to the day, when in December 1901 one James Wilson, Secretary of Agriculture in Theodore Roosevelt's administration, issued a report arguing for the preservation of the forests and rivers of these mountains:

> "The Southern Appalachian region embraces the highest peaks and largest mountain masses east of the Rockies..."
>
> "...Upon these mountains descends the heaviest rainfall of the United States, except that of the North Pacific coast..."
>
> "...The rivers which originate in the southern Appalachians flow through or along the edges of every State from Ohio to the Gulf and from the Atlantic to the Mississippi..."
>
> "...These are the heaviest and most beautiful hard-wood forests of the continent. In them species from east and west, from north and south, mingle in a growth of unparalleled richness and variety..."
>
> **"...the preservation of these forests is imperative..."**

Wilson's plea went largely unheard, and forty-eight years later Aldo Leopold, in his classic *Land Ethic*, rhetorically asked that when we "sing our love for and obligation to the land of the free and the home of the brave...just what and whom do we love?" He knew what we didn't appear to love:

> "Certainly not the soil, which we are sending helter-skelter downriver. Certainly not the waters, which we assume have no function except to turn turbines, float barges, and carry off sewage. Certainly not the plants, of which we exterminate whole communities without batting an eye. Certainly not the animals, of which we have already extirpated many of the largest and most beautiful species."

Although the environmental history of the ensuing century had its triumphs, our relationship with the natural world remains fraught. So, the good news is that Leopold's land ethic has been realized, even if only in part. The bad news is that it is under siege as never before. But the hopeful news is the young people seated before us — they are the future of the environmental ethic in this country!

Acknowledgements

We are very grateful to UNC-IE leadership and staff for their many and varied contributions to the success of this year's program, especially Jaye Cable, Tony Reevy, Greg Gangi, Geoff Bell, Violet Anderson, Evelyn Johann, Genesis Gibbs, Chrissie Greenberg, and Myra Walters Burke.

We deeply appreciate, too, the many local / regional individuals and organizations who have made this semester the successful, enriching experience it was for our students. Special thanks, first, to our mentors for working with the students on their internship projects this year:

Chris Jenkins, Orianne Society	Clint DeWitt, Kanuga Conference Center
Patrick Brannon, HBS	Buzz Williams & Nicole Hayler, Chattooga Conserv.
Ron Lance, NA Land Trust	Timm Muth, Jackson Co. Green Energy Park
Trip Krenz, WCU	Gary Wein, Highlands-Cashiers Land Trust

Warm thanks to Gary Wein (Highlands-Cashiers Land Trust), who has once again taught *Landscape Analysis* and tirelessly provided his GIS expertise for the students' projects; and Jason Meador (Mainspring Conservation Trust) and IE-HFS alum and Macroinvertebrate Maven Michelle Ruigrok, under whose joint expert guidance the Capstone project was conducted. And of course, we thank the many faculty and guest lecturers who shared their expertise with the students this year: Mark Lord, Tom Martin, & Jeremy Hyman of WCU, JJ Apodaca (Tangled Bank Conservation), Jeff Wilcox (UNC-Asheville), Brent Martin (Alarka Expeditions), Anya

Hinkle, Rich Baird (Mississippi State Univ.), Jennifer Knoepp (Coweeta Hydrologic Laboratory), Jana Polson (University of Tennessee), and Brooks & Kay Wade (Jocassee Lake Tours), and Julie Townsend, Susan Sachs and Paul Super (Great Smoky Mountains National Park).

Our friends in the Highlands community once again warmly embraced our students, and make them feel welcome in their home away from home away from home this term: many thanks to the staff of the Highlands Recreation Park and Hudson Library, with a special shout-out to Will Barclift and Grant Benoit at The Bascom.

Last but not least, here at the Station the students were in the capable hands of our wonderful staff, especially Katie Cooke, Mike McMahan, Patrick Brannon, Eliese Ronke, and Chan Chandler — many thanks to you all!

~ Jim Costa & Sarah Workman

Table of Contents

EFFECT OF ASPECT ON SALAMANDER DIVERSITY AT BIG RIDGE PROPERTY, JACKSON COUNTY, NC

GRACE BOWMAN AND JACKSON DENTON

Abstract. We conducted daytime area-constrained surveys of salamanders at 23 sites in the Big Ridge property in Jackson County, NC, to examine the effect of aspect and associated microhabitat on species diversity in the Southern Appalachian Mountains. At each site, we established a 30 x 30 m plot, collected data on environmental variables that serve as microhabitat for salamanders, and searched beneath logs and rocks to create inventories of species diversity and abundance. Spearman's Rank correlation analyses demonstrated a significant positive correlation between aspect and site moisture class, with more northerly-facing, mesic slopes resulting in greater salamander species richness. Both overall and terrestrial salamander diversity had significant positive correlation with moisture class and greater proportions of Class 4 and 5 coarse woody debris (CWD) at mesic northerly sites, but negative correlation with Class 3 CWD more common at drier, southerly sites. The presence of streams or seeps at each site significantly increased overall salamander diversity due to the inclusion of aquatic species, but had no effect on the diversity of terrestrial species. Few studies have been conducted of salamander diversity in direct relation to slope aspect, and this study provides a baseline set of data that may be useful in future studies of Southern Appalachian herpetofauna.

Key words: aspect; biological inventory; coarse woody debris; Desmognathus*;* Plethodon*; Southern Appalachians; species diversity*

INTRODUCTION

The topography, climate and geologic history of the Southern Appalachians all contribute to the rich biodiversity found within the region today. Cool, mesic conditions and abundance of woody vegetation, streams, and seeps provide habitat suitable for a high density of salamander populations. Mountaintop isolation between populations as well as glacial history has led to high rates of allopatric speciation over time (Petranka 1998).

The majority of salamanders found within the Southern Appalachians belong to the family Plethodontidae. These salamanders do not possess lungs, and require moist, cool microhabitats for cutaneous respiration and reproduction such as decaying logs (Whiles and Grubaugh 1993) and deep leaf litter (Ash 1995). Coarse woody debris (CWD), particularly in the later stages of decay, serve as daytime refuges for many species of terrestrial salamanders while other species, especially those with aquatic larvae, generally occupy habitats such as streams and seeps (Welsh and Droege 2001).

Microclimatic conditions can vary considerably across landscapes (Matlack 1993). In the southern Appalachians, moisture gradients created by slope and aspect can significantly influence the distribution of species (Whittaker 1956, Heatwole 1962, Brannon 2002). Duration and intensity of sunlight on northerly slopes are greatly reduced, resulting in cooler temperatures and higher levels of environmental moisture (Wales 1972) ideal for the survival and reproduction of salamanders (O'Donnell et al. 2014). Greater sunlight exposure on more south-facing slopes can result in xeric microhabitats including much drier leaf litter (Wales 1972) which is suboptimal for salamanders (Ash 1995) and restricts surface activity (Jaeger 1980, Keen 1984).

Big Ridge, located near Glenville, Jackson County, North Carolina, is a privately-owned preserve that encompasses approximately 1,214 square hectares of forested land. The main spine of the ridge runs approximately northwest to southeast. Aside from five knobs which were cleared for former development projects that were later abandoned, Big Ridge property is entirely forested.

The dominant forest type is acidic cove forest, followed by montane oak-hickory stands (Lance, pers. comm.). A network of roads within Big Ridge represents remnant logging roads and skid trails from previous timber harvests, as well as several newer roads installed during development activity. The majority of the tract appears to have been last cut for timber in the 1940s, although two sections were also clear-cut in the 1990s (Lance, pers. comm.).

A primary duty of the current land manager at Big Ridge is to maintain species inventories for forest management purposes and to monitor any rare species. Since 2011, several biological surveys have been conducted on the Big Ridge property. An inventory of vascular plants within Big Ridge began in 2013 and currently contains 595 species. Other biological surveys have been less comprehensive, making Big Ridge an important location for creating and inventory of herpetofauna (Lance, pers. comm.). Our objective was to conduct a baseline survey of salamanders at various locations throughout the Big Ridge property. As a part of our survey activities, a secondary goal was to examine salamander species diversity patterns in relation to slope aspect and any associated difference in microhabitat.

METHODS

Surveys were conducted at 23 localities throughout the Big Ridge property (Fig. 1). Survey areas were selected partially for convenience, and consequently were typically located close to the ridgeline along main roads accessible by motor vehicle or ATV. Sites were also selected to reflect a variety of forest habitat types and slope aspects. Forest type was determined through the identification of dominant tree and shrub species within each plot, and from a map of a previous vegetation survey of the Big Ridge property. We then assigned each plot a moisture classification on a 1-5 scale using the protocol of Ford et al. 2002b, ranging from most xeric to most mesic, respectively. GPS coordinates, aspect, and elevation were obtained for each plot, and any presence of a water source (i.e. stream or seep) within the plot was also recorded. Slope aspect was converted into a linear scale using the formula utilized by Ford et al. (2002a), by which aspect was ranked from most mesic (northeasterly) to most xeric (southwesterly).

At each site, we established a 30 x 30 meter plot along a slope. Each plot was transected by three parallel lines 10 meters apart, along which four measurements of leaf litter depth were taken at 10-m intervals for a total of 12 measurements. We obtained leaf litter depth by inserting a ruler through the leaf litter to the soil A-horizon, and report mean values (Brannon et al. 2014).

All coarse woody debris within each plot ≥ 10 cm in diameter were ranked by their state of decomposition according to five decay classes (Whiles and Grubaugh, 1993). "Usable" CWD for salamanders are logs that lay completely on the ground (Petranka et al. 1994), and range from Class 3 (partially decomposed) to Class 5 (heavily decomposed). Values were then converted to percentages for each decay class. For analyses, we chose to group Class 4 and Class 5 CWD, because they represent more optimal microhabitat for salamanders (O'Donnell et al. 2014). Both are heavily decayed and, as well as providing protective cover, also retain greater amounts of moisture than Class 3 logs due to their spongier texture (Whiles and Grubaugh, 1993).

Our surveys were conducted from September to November 2017. Due to accessibility constraints, sites were surveyed during only daylight hours, and each site was surveyed only once. All movable pieces of CWD, as well as all movable rocks in any streams and seeps, were overturned within each plot to search for salamanders. Any salamanders found underneath were then tallied and identified by species (Smith and Petranka 2000). We designated each salamander species as either "terrestrial" or "aquatic." Members of the genus *Plethodon* are strictly terrestrial, while members of *Desmognathus* generally occupy streams and seeps and develop from aquatic larvae. However, we chose to designate the Ocoee Salamander (*D. ocoee*), as well as the Two-lined Salamander (Eurycea wilderae) and the eft stage of the Eastern Newt (*Notophthalmus viridescens*) as terrestrial species because, although these species are aquatic breeders, adult individuals are often found far away from water sources in surrounding forest (Petranka 1998).

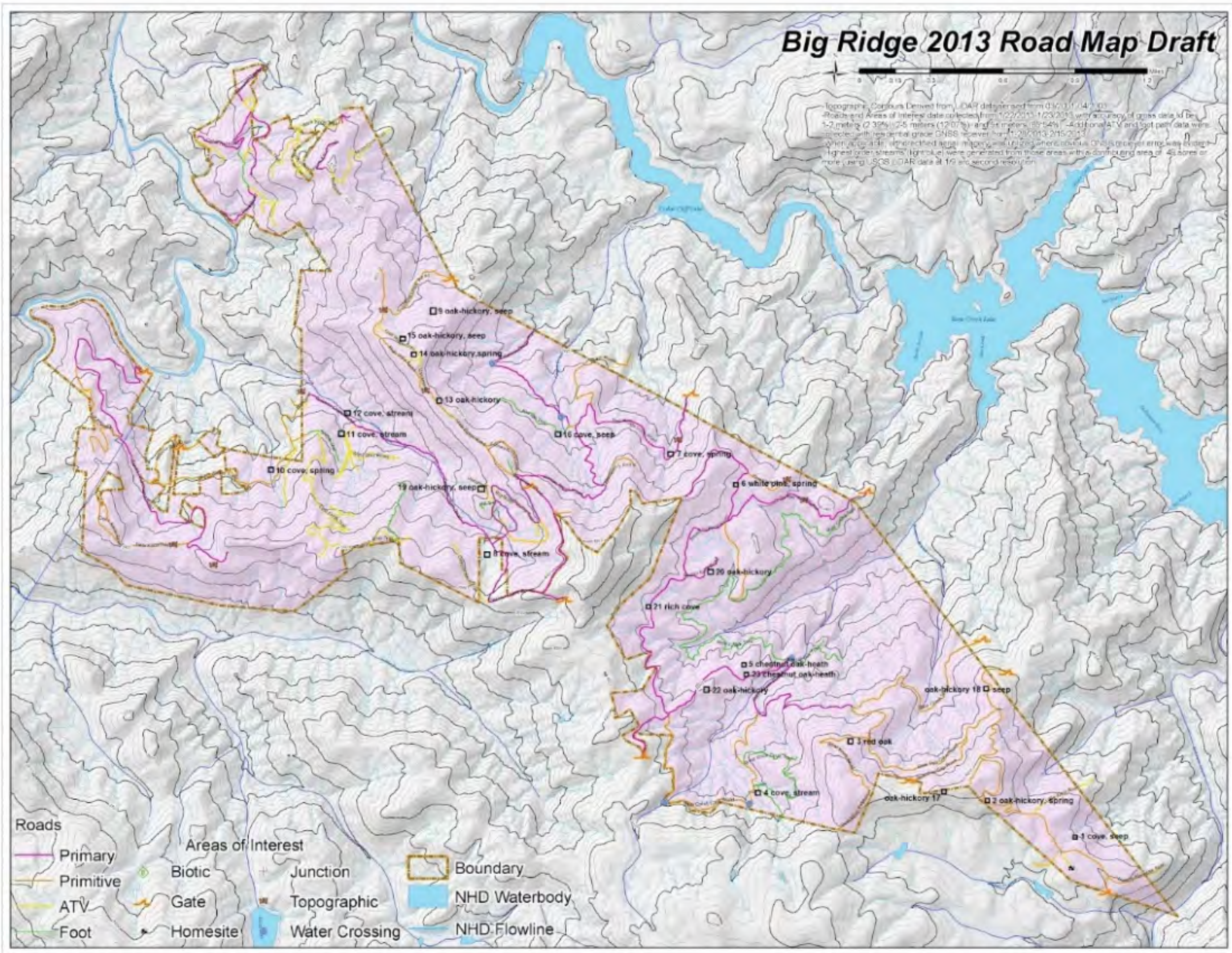

FIG 1. Map of Big Ridge Property and sites selected for herpetofaunal survey. Sites are represented by points, primary roads are represented by red lines and topographical contours are represented by dark blue lines.

Spearman's Rank Correlation was utilized to analyze the effect of aspect on each environmental variable measured. Diversity values for both overall salamander species and terrestrial salamander species were obtained using Shannon's Diversity Index. We also examined relationships between habitat variables and species diversity using Spearman's Rank Correlation.

RESULTS

We observed a total of 260 salamanders representing eight species. Of this total, we observed two species of aquatic salamanders (n=63) and five species of terrestrial salamanders (n=197; table 1). Of the 63 identified aquatic individuals, 19 were unidentified *Desmognathus* larvae. Most of these individuals had not yet fully developed and could not be reliably identified to species.

TABLE 1. Salamander species and number of individuals encountered during searches.

Aquatic Species	Common name	Total
Desmognathus monticola	Seal salamander	30
D. quadramaculatus	Black-bellied salamander	14
D. spp.	Unidentified *Desmognathus* larvae	19
Total Aquatic Individuals		63
Terrestrial Species		
D. ocoee	Ocoee salamander	124
Eurycea wilderae	Blue Ridge Two-Lined salamander	2
Notophthalmus viridescens (eft)	Eastern Newt (eft)	11
Plethodon metcalfi	S. Gray-cheeked salamander	50
P. serratus	S. Red-backed salamander	1
P. teyahalee	S. Appalachian salamander	9
Total Terrestrial Individuals		197

Aspect was found to be significantly correlated with moisture class in more mesic forests occurring on northerly-facing slopes (r_s=0.435, df=21, p=0.04, table 2). However, no other habitat variable was significantly correlated with aspect (table 2). Aspect was also significantly correlated with the number of salamander species (richness) at each site (r_s=0.407, df=21, p=0.05), but not with the number of individuals (table 2).

TABLE 2. Spearman's rank correlation coefficients (r_s) of habitat variables with aspect.

Variable	r_s
Moisture class	0.435*
Water source	0.094
Leaf litter depth	0.185
% CWD 3	-0.165
% CWD 4&5	0.165
Number of salamanders	0.272
Salamander species richness	0.402*

*p<0.05

Overall salamander species diversity (H') had significant positive correlation with the presence of a water source in the plot (r_s=0.804, df=21, p<0.001), but terrestrial diversity (H'_{terr}) was not (table 3). Overall diversity was also significantly positively correlated with forest moisture class (r_s=0.499, df=21, p=0.02), and % CWD 4&5 (r_s=0.659, df=21, p<0.001), but negatively associated with % CWD 3 (r_s=-0.659, df=21, p<0.001; Table 3). Similarly, terrestrial salamander species diversity was significantly positively correlated with moisture class (r_s=0.542, df=21, p=0.01) and % CWD 4&5 (p= .0113), and negatively correlated with % CWD 3 (r_s= -0.518 p= .0113). Unlike overall diversity, there was a significant relationship between terrestrial salamanders and aspect, with diversity greater on more northerly, mesic slopes (r_s=0.433, df=21, p=0.04; table 3).

TABLE 3. Spearman's rank correlation coefficients (r_s) of habitat variables with overall salamander diversity and terrestrial salamander diversity.

Variable	Overall r_s	Terrestrial r_s
Aspect	0.342	0.433*
Water source	0.804**	0.353
Moisture class	0.499*	0.542*
Leaf litter depth	0.085	0.104
% CWD 3	-0.659**	-0.518*
% CWD 4&5	0.659**	0.518*

*p<0.05; **p<0.001

DISCUSSION

A positive correlation between slope aspect and moisture class can be explained by the general characteristics of north-facing slopes; these slopes are generally cooler and wetter, which result in mesic forest types (Whittaker 1956, Wales 1972). Slopes facing the north generally receive lesser amounts of sunlight, contributing to lower temperatures and higher moisture retention (Brannon 2002). The favorable mesic microclimates of more northerly-facing slopes can support a larger abundance and variety of salamander species than more xeric slopes (Harper and Guynn 1999, Ford et al. 2002b) because of the ecological and physiological requirements of salamanders (Feder 1983, Keen 1984). Mesic conditions result in a greater proportion of heavily decomposed logs which are optimal microhabitats for salamanders (Petranka et al. 1994).

Although overall salamander diversity, which includes both the number of species and individuals, was positively correlated with forest moisture class, our analysis showed no correlation with aspect. The presence of a stream or seep at a site is independent of slope aspect, and adjacent areas can often provide isolated patches of moisture in these otherwise dry environments (Brannon 2002). Water sources can result in a larger number of species found at the site as a whole with the inclusion of aquatic species in addition to any terrestrial salamanders (Ford et al. 2002b). Conversely, terrestrial salamander diversity was significantly associated with aspect and moisture class, as terrestrial species are more reliant upon environmental moisture on the forest floor for cutaneous respiration and egg deposition (Feder 1983), rather than the presence of a stream or seep like those species with aquatic larvae (Ford et al. 2002a).

While vegetation type can vary depending on the aspect of a slope (Wales 1972), the depth of associated leaf litter in our study appeared to be fairly consistent among sites. Though leaf litter can be an important habitat feature for harboring abundant invertebrate prey and for providing sufficient cover from predators, litter moisture may be a more critical factor in determining the diversity and abundance salamanders since it allows for greater mobility (Jaeger 1980, Keen 1984), especially after recent rainfall (O'Donnell et al. 2014). Under dry conditions, salamanders are restricted to moist patches in an environment, such as beneath or inside Class 4 and 5 logs (Jaeger 1980, Whiles and Grubaugh 1993, Grover 1998). Although Class 3 logs are considered to be usable habitat for salamanders, they serve mainly as temporary refugia and not as long-term microhabitat, and likely cannot support as high of a diversity as areas with greater proportions of moister, "spongy" CWD like those in later stages of decay (Petranka et al. 1994, Whiles and Grubaugh 1993).

The data collected during this survey can be used as supplemental information for any future studies relating to salamanders or forest ecology conducted in the Big Ridge property, with additional applications for other areas in the southern Appalachians. The biological inventory created, as well as environmental data recorded, may serve as a baseline for research focusing on salamander distribution patterns or the effects of various environmental variables on salamander diversity and abundance. Despite a wealth of past studies that have demonstrated the relationship between Plethodontid salamander distributions and environmental moisture (e.g. Hairston 1949, Petranka et al. 1994, Grover 2000, Ford et al. 2002a, O'Donnell et al. 2014), this is one of the few (Harper and Guynn 1999, Ford et al. 2002b, Peterman and Semlitsch 2013) that has directly examined the effects of slope aspect on salamander diversity, albeit with some limitations (Dodd and Dorazio 2004, Strojny and Hunter 2009, O'Donnell and Semlitsch 2015). Additional studies that incorporate more sites, a longer sampling period, samples of additional microhabitat features, along with repeated surveys at various times of day should provide a greater understanding of the effects of aspect and associated microclimate on southern Appalachian salamander diversity.

ACKNOWLEDGEMENTS

We would like to thank Patrick Brannon for mentoring us on this project. We would also like to thank Ron Lance, the Land Manager of Big Ridge, for his help in selecting and surveying each site and navigating the property, as well as his dog Ruby for her unwavering companionship and morale boost throughout the study.

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CONSIDERATIONS FOR BOG TURTLE (*GLYPTEMYS MUHLENBERGII*) INTRODUCTION AT FALLS CREEK, OCONEE CO., SOUTH CAROLINA

AIDAN BUIE AND ISHIKA KUMBHAKAR

Abstract. The beaver ponds blockading Falls Creek, Oconee County, South Carolina provide an interesting and unique habitat. Remote sensing and a general vegetative survey of the site provided comprehensive information of the location and its habitat features. The results suggest that this could be a good opportunity for the possible introduction of the bog turtle *(Glyptemys muhlenbergii*). However, the small size of key habitat for the bog turtle in the site may be a potential issue for introduction. This, along with the issue that much of the site is open water due to beaver dams, means that any turtles would likely be confined to the small region of ideal habitat in the eastern area of the site. Bog turtles benefit from disturbance, as it reduces cover from forested areas and allows sunlight penetration. Therefore, opening land adjacent to the pond for cattle or horse pasture may improve the quality of the proposed reintroduction site.

Key words: bog turtle; habitat; habitat alteration; population size; wetland conservation.

INTRODUCTION

Habitat loss is one of the greatest threats that face animal and plant species. The modification of land for agricultural or other developmental purposes, as well as the alteration of ecological system components, has transformed the environment so drastically and rapidly that many species struggle to adapt. Bog turtles (*Glyptemys muhlenbergii*) are one such species. This critically endangered species was historically found throughout the southeastern region of the United States. The species is split into two disjunct populations - the northern range and the southern range, with the southern occurring largely in the Appalachian region. There is some variation in scientific naming due to taxonomic discussion regarding the genetic differences between the two allopatric populations. The bog turtle was reclassified from the *Testudo* genus into *Clemmys* in 1835, and was subsequently synonymized with genus *Glyptemys* in 2001. The southern population was originally described as *Clemmys nuchalis*, but subsequent study showed that there is insufficient genetic divergence between the two populations to warrant taxonomically separating them (Macey 2015). Due to habitat loss and degradation, as well as removal by pet dealers over the past few decades, their population has been in severe decline (Sirois et al. 2014). Much of direct wetland destruction has ceased or at least has been reduced by improved state and federal regulations; however, invasive species encroachment and succession threaten the habitat of bog turtles. The increase of woody vegetation in wetland areas decreases the amount of sunny and open areas required for bog turtles to thrive (Carter et a. 1999). An additional issue concerns the fact that the majority of land in the southern Appalachian region is privately-owned. This means that the decisions of individual private landowners are crucial to the health of wetlands and regional ecosystems.

Many wetlands that are privately-owned in the southern Appalachian region have not been assessed as potential bog turtle habitat or had bog turtle presence/absence studies conducted. It is necessary to have a better understanding of what potential habitats may exist in order to predict the extent of their regional population. It is also necessary to consider that the distances that bog turtles may migrate and the required area for a viable population are not definitively known, which further complicates studying the species (Sirois et al. 2014).

Therefore, conservation of bog turtle habitats or populations largely relies on the maintenance of wetlands. Many states have lost wetlands in attempts to convert that land into drier land more suitable for agriculture. After recognizing the impact that these conversions had on wetlands, the Food Security Act of 1985 discouraged this practice (Sucik and Marks 2010). In terms of private land, securing the longevity of a wetland is often done by creating an easement. A conservation easement consists of a voluntary legal agreement between the landowner and a larger agency (e.g., government agency or land trust) to limit the land's use and preserve its ecological value. For physical treatments of wetlands that are threatened by encroachment of woody vegetation, it is often useful to introduce browsing animals in order to cut back on the growth. Land historically classified as wetland that has been altered or converted can utilize prescribed burning or deep burning around the base of larger trees within the area to open up the canopy cover, which aids in restoring wetland features. Additionally, any rerouting or draining of water in the area should be returned to its original state.

The goal of this study was to assess a privately-owned wetland based on preferred bog turtle habitat characteristics as a potential reintroduction site. Specifically, we sought to designate the amount of viable habitat area for bog turtles within the wetland studied. Based on the assessment, we discuss possible options for treatment and maintenance of the wetland.

METHODS

This study focused on the upper section of the beaver pond found along Falls Creek in Oconee County, South Carolina, during the fall of 2017. This bog was chosen based on its relatively large size and availability for study. Both owners of the properties gave us the permission necessary to perform this study on their land. Most of our focus is on the Perrin Property (Parcel number 114-00-03-040), though it should be noted that the pond extends all the way down Chattooga Ridge Road into the George Property (Parcel number 114-00-03-073). The bog is also part of a conservation easement on the George Property, but there is no easement on the Perrin property. We chose the site because it had no known presence of bog turtles and therefore is a possible site for reintroduction of a new population. Finding a suitable wetland for this study was confounded by the fact that not all areas are catalogued; bog habitats on private property may exist, possibly without public knowledge. Furthermore, wetland drainage is a constant threat to bog habitats from humans looking to alter the landscape for developmental needs. For the purpose of this study, the area of the wetland is as delineated by property lines (fig. 1).

In order to gather the required data, we did both a visual survey of vegetation and gathered GPS points following a 20.3 cm (eight-inch) water depth around the Perrin property. For the visual survey, we sampled random plants around the wetland as we walked the boundary. We also focused on identifying all of the plant species we found in the specific area that was best suited for bog turtle habitat (the mapped area). Vegetation types can be used to extrapolate habitat aspects such as water quality, soil type, soil saturation, etc. with the knowledge that specific plants require different requirements to thrive. The plants were assessed as indicator species, specifically in terms of soil saturation levels and pH ranges. Bog turtles do not rely on specific plant species in terms of their habitat, therefore we did not consider it necessary to do a more specific survey type.

The focused study area consists of the area delimited by the points taken from the dry edge of the wetland habitat inwards until we reached areas of 20.3 cm in depth. This depth is considered to be the maximum that bog turtles will tolerate, as they are not aquatic and do not swim (M. Knoerr, pers. comm.). The points were taken using a Trimble handheld GNSS system and antenna, which consisted of the unit and a receiver on a pole. One person walked out into the bog, measured water depth, and flagged the points where the water was at the critical depth. The other person followed with the Trimble unit and took measurements of the GPS locations at the flagged points. The resulting points were then downloaded via Windows Mobile and mapped in ArcGIS ArcMap® 10.5.1. The points were joined into a polygon in order to better visualize the viable habitat. This polygon, overlaid onto base maps of different landscape aspects, was used to determine the specific habitat qualities and historical vegetative cover of the area.

A large feature of this study was the development of maps in order to spatially analyze the study area. Both elevation data and aerial imagery were found via the The National Map of the USGS. Elevation datasets were DEM one-third arcsecond resolution, acquired from the USGS National Elevation Dataset (USGS 2013). Land cover datasets of the National Land Cover Database (NLCD 2011) was acquired from the Multi-Resolution Land Characteristics Consortium. Aerial images were two combined 3.75 arcminute square images from the National Agriculture Imagery Program (USDA-FSA 2015a, USDA-FSA 2015b).

RESULTS

Habitat Mapping

We sampled and mapped the periphery of a privately-owned wetland as well a focused study area where there was viable bog turtle habitat (fig. 1).

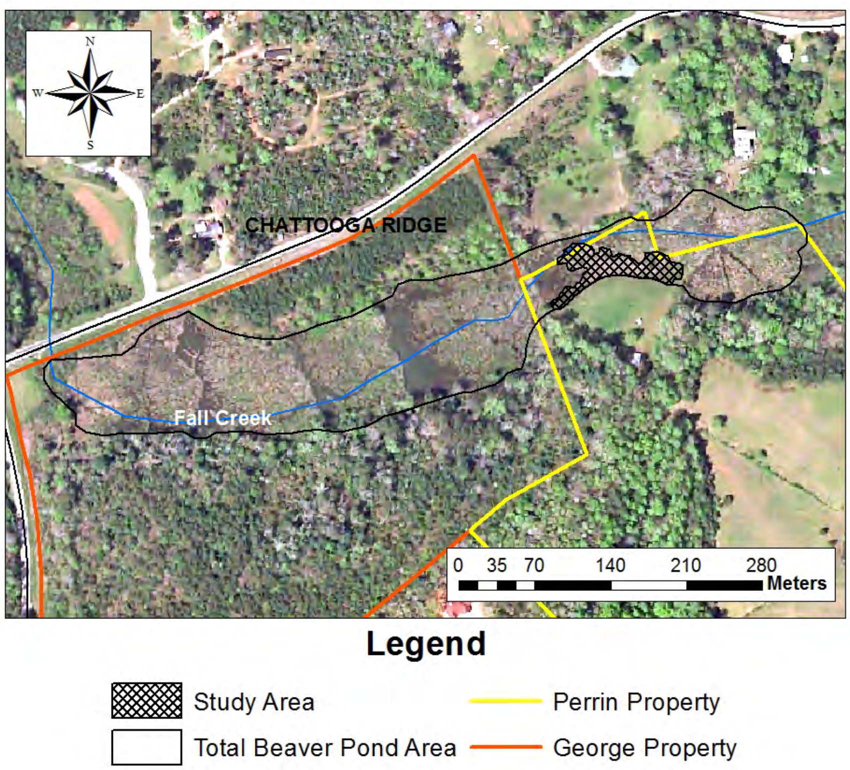
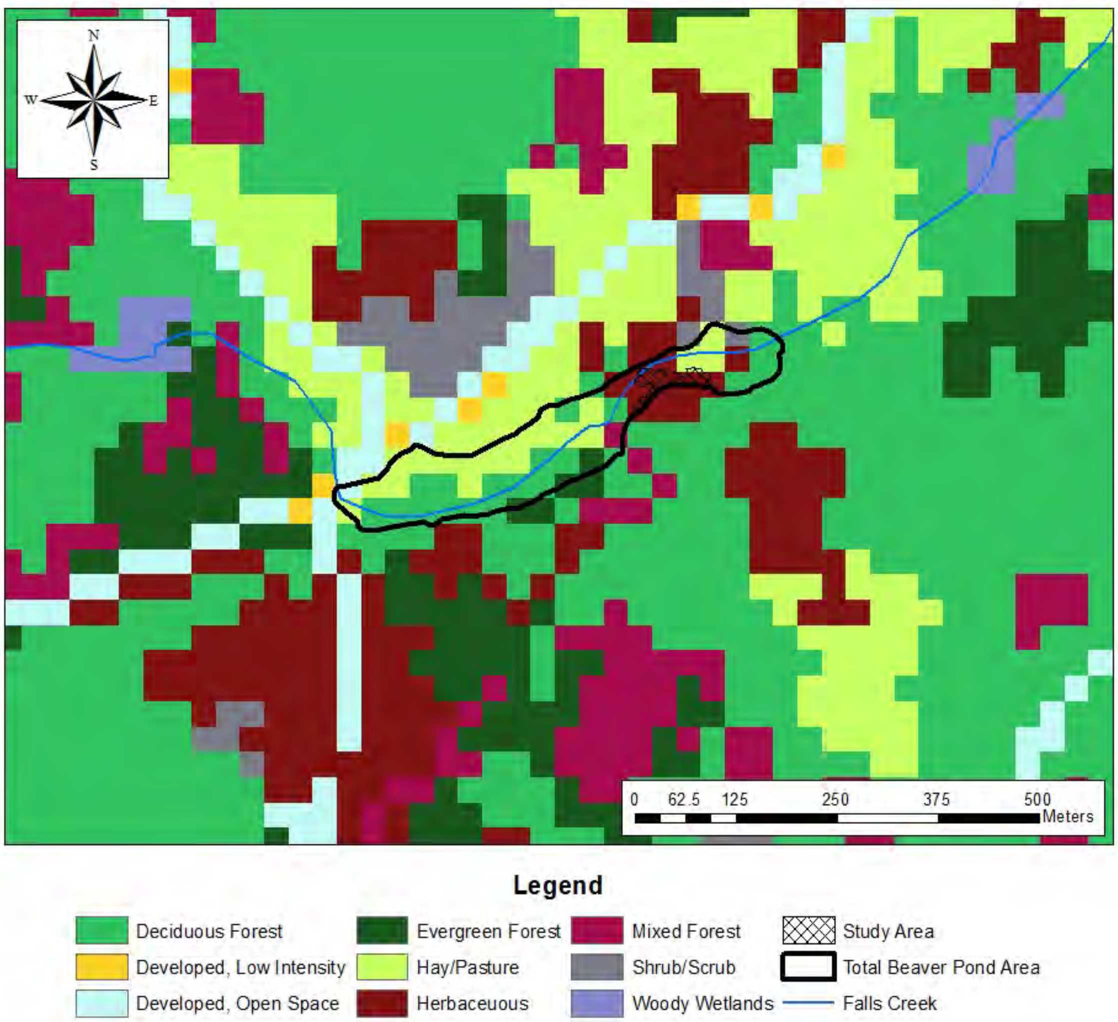


FIG. 1. A map of the overall study area, which is the total beaver pond area, and the focused study area. The yellow line delineates the Perrin property, and the red line delineates the George property.

The total area of the beaver pond area (total wetland area) is 5.95 hectares (59,536.80 m²). The area of the study area was 0.28 hectares (2780.16 m²), or 4.67% of the total beaver pond area.

FIG. 2. A 30-meter by 30-meter resolution raster image from NLCD 2011. This was the latest land cover dataset available.

The land cover data from 2011 shows that the wetland area as recently as six years ago consisted of hay/pasture, deciduous forest, herbaceous cover, some shrub cover, and some mixed forest. The study area in 2011 consisted almost entirely of herbaceous cover, with a small part being hay/pasture. The total wetland area is now a series of four beaver dams, so it is evident that



beavers created dams in the wetland after 2011, leading to the creation of this series of ponds at least approximately 2-3 meters deep.

Vegetation Sampling

Lists of sampled vegetation in the focused study area are represented in table 1 and in the total beaver pond area in table 2.

TABLE 1. Vegetation encountered in the study area, as well as the plant's moisture requirement and habitat pH range (USDA-NRCS 2017).

Common Name	Scientific Name	Moisture Use	pH Range
Woolgrass/Marsh Bulrush	*Scirpus cypurinus*	High	4.8-7.2
False Nettle	*Boehmeria cylindrica*	Medium	5.1-7.0
Meadow Beauty	*Rhexia virginica*	High	-
Bur-reed	*Sparaganium americanum*	High	4.9-7.3
Cat-tail	*Typha latifolia*	High	5.5-8.7
Ironweed	*Vernonia glauca*	-	-
Juncus	*Juncus* sp.	-	5.5-7.0
Aster	*Aster* sp.	-	-
Joe Pye Weed	*Eutrochium fistulosum*	-	4.5-7.0
Yellowroot	*Xanthorrhiza simplicissima*	-	-
Red Maple	*Acer rubrum*	High	4.7-7.3
Black Willow	*Salix nigra*	High	4.8-8.0
Tag Alder	*Alnus serrulata*	High	5.0-7.0

TABLE 2. Vegetation encountered in total beaver pond area, as well as the plant's moisture requirement and habitat pH range (USDA-NRCS 2017).

Common Name	Scientific Name	Moisture Use	pH Range
Switch Cane	*Arundinaria tecta*	-	-
Red Maple	*Acer rubrum*	High	4.7-7.3
Tag Alder	*Alnus serrulata*	High	6.8-7.2
Jewelweed	*Impatiens capensis*	Medium	6.4-7.4
Sedge	*Carex* sp.	-	-
Rush	*Scirpus* sp.	-	5.5-7.0
Turtle-head	*Chelone* sp.	-	-
Duck potato	*Sagittaria latifolia*	High	4.7-8.9
Sphagnum moss	*Sphagnum* sp.	High	-
Sourwood	*Oxydendrum arboreum*	Low	4.0-6.5
Bone-set	*Eupatorium perfoliatum*	-	-

Dog-hobble	*Leucothoe fontanesiana*	Medium	4.5-6.0
Arrowroot	*Maranta arundinacea*	-	-
Lily	*Lilium* sp.	-	-

Although some vegetation was common to both the study site and the rest of the beaver pond area, there were significant differences. Species such as sphagnum moss (*Sphagnum* sp.), turtle-head (*Chelone* sp.) and duck potato (*Sagittaria latifolia*) are common in bog habitats (Carter et al. 1999), but were not found in the study area. The vegetation in the focal study area indicated a pH range that is slightly acidic to neutral, while vegetation in the overall beaver pond area varied widely indicating slightly acidic to slightly basic pH.

Discussion

Despite the common name of bog turtle, which implies that the animal would be found in bogs, they are found in spring-fed fens. Carter et al. (1999) found that the primary habitat preference for bog turtles are wet meadows, bulrush or alder edge, and where tag alders grow. In terms of hydrology, these turtles prefer shallow rivulets and soft/saturated soils (Stratmann et al. 2016). They also require open-canopy areas that allow for access to sunlight, essentially meaning low vegetation such as herbaceous plants, shrubs or sparse small trees, and/or sedges. Some of the species that are commonly found in these habitats (sphagnum moss, turtle-head, duck potato) were found in the beaver pond area but not the study area (tables 1 and 2). However, the occurrences of these plants were only in small patches around the area, meaning that they would probably not provide enough total habitat area for a functioning bog turtle population. The moisture use and pH data available for the vegetation in the study area indicates that it requires high moisture, as expected, and prefers acidic to neutral pH. In comparison, the rest of the beaver pond habitat varies more in both categories, which allows us to infer that also varies in habitat type. This is yet another factor that implies that the beaver pond area is not be suitable as a whole, but may have small suitable patches.

The main bog turtle habitat issue in a fen is water depth. These animals are not aquatic and therefore cannot tolerate deep water. The study area we delineated using remote sensing was done based on the 20.3 cm maximum depth. It was also the largest area in which doing so was possible compared to the rest of the beaver pond area. Much of the wetland had a distinct shoreline, where the land cover went from dry land to open water without a saturated area in-between. Other parts, notably where we found sphagnum moss and other bog habitat indicator plants, were as aforementioned, small in size and therefore not viable as a successful bog turtle habitat.

Looking at the NLCD 2011 (fig. 2), we can infer that the beaver dams and resulting ponds are a fairly recent occurrence, having been created at some point in the past six years. When the NLCD 2016 is available, it would be interesting to see what changes have occurred in the study area. Bog turtles benefit from disturbance, as it opens up forested areas. Their preferred habitat includes open canopy areas where there is plenty of sunlight. In the past, large ungulates such as elk and bison would have created openings in the forest along creeks, fens, or wetlands (Davis 2000). With the absence of these species, beavers or ranchers may clear trees. Bog turtles can additionally often be found in pastures with cows or horses, because they create muddy areas by tramping around the water. The study area (fig. 1) is along a pasture where horses were once kept. Studies show that wet pastures where large animals graze are often suitable candidates for bog turtle habitats (Tesauro and Ehrenfeld 2007). This is because hooves leave indents in the soft, muddy ground that can serve as ideal microhabitats for the turtles. The vegetation of the study area also indicates saturated soils with an acidic to neutral pH. These habitat characteristics suggest that the study area is a realistic area to either find or reintroduce bog turtles.

Habitat size is an area of concern for possible turtle reintroduction. Carter et al. (1999) found that the home range of bog turtles was 0.15 hectares via cluster analysis and 0.52 hectares via minimum convex polygon analysis. The turtle population studied in the Carter et al. (1999) study was composed of 13 males and 12 females across three sites. Although these sites varied in area by greater than 400 percent, the home ranges did not differ (Carter et al. 1999). A study conducted in western North Carolina found the female home range to be 0.6064 hectares and male to be 0.4458 hectares (McCoy 2016). The same study found average daily movement rates for female bog turtles to be 8.34 meters per day and males to be 5.0 meters per day (McCoy 2016).

When considering that the entire beaver pond has an area of 5.95 hectares (fig. 1), this is certainly large enough for a sizable turtle population. Unfortunately, it is nearly entirely open water (fig. 1) and would require drainage and likely the removal of beavers to be fully utilized by the turtles. The small study area has a more favorable habitat of wet meadow, rush, and alder, but is only 4.67% of the total habitat at 0.28 hectares. Fortunately, the McCoy study found bog turtles at a variety of water and mud depths, so our core study area habitat is likely larger than we measured it to be. McCoy (2017). found bog turtles out of the water (0.0 cm water depth), at a maximum of 50.1 cm, and at an average of 15.8 cm deep.

It is difficult to say if the study area would be large enough to house a viable bog turtle population. A study done in Maryland found that turtle density varied from 7 to 213 turtles per hectare (Chase et al. 1989). This is obviously quite a large range, and populations from Maryland likely vary in many aspects from populations in western North Carolina. Gibbs and Shoemaker (2013) suggest that nearest-neighbor (< 2 km distance) populations of bog turtles should be managed as interconnected units. These total areas also include 'stepping stone' areas that may not be viable for long-term populations, but are necessary for connectivity of habitats (Gibbs and Shoemaker 2013). Bog turtles can also be found outside of open-canopy areas, and may additionally be found in scrub/shrub areas (McCoy 2016). Conserving larger tracts of wetland that may not necessarily be the most suitable habitat is therefore required for habitat management (McCoy 2016).

CONCLUSION

There are several factors that must be considered before making any management decisions about maintenance of habitat for bog turtles. One of these factors is adequate habitat size. The beaver pond area is not currently suitable for bog turtles, but after the beavers leave their dams it may return to a more favorable habitat, as seen in the NLCD 2011 (fig. 2). Managing Fall Creek to remove channelization and create overland flow may be a possibility for the future and should be considered if bog turtles were to be introduced here. This would help encourage ideal wetland conditions throughout the beaver pond area.

As the beaver ponds themselves are a relatively recent development, it may be valuable to place wells in multiple spots around the wetland and monitor trends in water levels over several seasons. Therefore, future studies should include temporary or permanent wells over a period of time of a year or longer in order to have a better understanding of the wetland hydrology. Similarly, the points taken at water depths of eight inches should be taken multiple times in a season in order to create a more accurate boundary of the area of viable habitat. Taking a greater amount of data over time would both provide insight into how or if the habitat is changing and also reduce random error. Obtaining a permit for a presence of absence study it in the study area would be necessary before any movement to conserve the habitat or introduce bog turtles. Communication between land trusts and other acknowledged habitats of bog turtles would be vital to a more regional approach to management.

The study area may be suitable for some bog turtles in its current state. However, it simply may not be large enough for a strong, reproducing population. Given the time limitations for this study and existing knowledge of bog turtle genetic diversity in isolated populations, it is very difficult to say with surety whether the focal study habitat would be viable for bog turtles. Gibbs and Shoemaker (2013) suggest that nearest-neighbor (< 2 km distance) populations of bog turtles should be managed as interconnected units. These total areas also include 'stepping stone' areas that may not be viable for long-term populations, but are necessary for connectivity of habitats (Gibbs and Shoemaker 2013). Bog turtles can also be found outside of open-canopy areas, and may additionally be found in scrub/shrub areas (McCoy 2016). Conserving larger tracts of wetland that may not necessarily be the most suitable habitat is therefore required for habitat management (McCoy 2016).

Another factor to consider is the potential for human-led alteration of the landscape to better suit the turtles. Opening more area up for pasture, and then grazing horses or cattle in the area could potentially promote excellent habitat for a bog turtle reintroduction. Extending further inland onto the Perrin property (fig. 1) is far more pasture for cattle to graze on, if the landowners were willing to allow this land use. One primary step would be allowing the wetland to become an easement. Creating a conservation easement would ensure the longevity of the wetland and protect any species that inhabit it.

Much still remains to be known and understood about the behavior of these rare creatures. The characterization from this study enlarges what we do know about them at this site in South Carolina. With this wetland vegetation and landscape data, the landowners working with the Chattooga Conservancy can examine issues that may motivate them to conduct a bog turtle presence-absence study or reintroduce this rare species in the area.

Acknowledgements

We would like to thank Dr. Costa and Dr. Workman for their unshakeable guidance throughout this semester. We would also like to thank Nicole Hayler and Taylor Howard for their assistance and helpfulness. We thank Dr. Wein and the Highlands-Cashiers Land Trust for his help with all things GPS and GIS related. Our fellow classmates deserve recognition for their hard work and positive energy this semester. And of course, last but not least, we thank Buzz Williams for his wonderful mentorship and willingness to learn along with us.

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POST FIRE SOIL CO_2 EFFLUX IN SOUTHERN APPALACHIAN HARDWOOD FORESTS

ANNA KELLY

Abstract. In fall 2016 over 50,000 acres of forest in western North Carolina were burned in arson fires, providing a unique opportunity for research. This study seeks to examine the response of forest soils to large disturbance events such as fire. As a proxy for biotic activity of soil microflora and –fauna, soil CO_2 efflux was measured using the LiCor LI-8100A Soil Gas Flux System. In addition to efflux, soil temperature and moisture were investigated as causal factors in post-fire shift in functional forest structure and dynamics. Data was collected at 4 sampling plots, two of each burned and unburned, with 9 sampling points, during a 10-week period. We found that soil CO_2 efflux was significantly higher in the unburned sampling sites than the burned sites, which may be attributed to differences in temperature, moisture, and other factors that were not measured; however, due to a limited data set, conclusive trends could not be identified.

Key words: burn; carbon dioxide; efflux; fire; soil; soil respiration; southern Appalachian hardwoods

INTRODUCTION

As effects of global climate changes have become reflected in present day ecological shifts, interest in studying relationships among natural systems of greenhouse gas regulation has grown. Earth's soils are a major component of nutrient and gas cycles, especially carbon dioxide (CO_2) via soil respiration (Bond-Lamberty and Thomson 2010). One of the key influential factors in climate change, CO_2 is a frequent focus of study, but popular discussion is often related to atmospheric phenomena. In actuality, soil carbon stores are greater than those in the atmosphere, so understanding complexities of carbon cycling between soils and atmosphere is important in mitigating and preparing for changes due to global climate change (Zhang et al. 2013). Understanding the factors that influence soil respiration rates is a point of relevance in studying the role soils fulfill in the carbon cycle.

Soil respiration occurs both above ground and below the soil surface through autotrophic and heterotrophic processes. Production of CO_2 below the soil surface originates from roots of vegetation, as well as heterotrophs such as animals, bacteria, and mycorrhizae. Aboveground, CO_2 is largely produced by litter decomposition (Zhang et al. 2013). These two factors combine to deliver the total soil respiration (Raich and Schlesinger 1992). As an essential part of ecosystem function, trace gases produced in respiration are both stored in soil pores and diffused into the atmosphere (Weil 2017). This research explores effects of a large disturbance event, forest fire, on the CO_2 flux of forest soils.

In some hardwood forests, soil efflux, the outflow of gases from soil pores, accounts for 60-80% of the total ecosystem respiration (Davidson et al. 1998). Efflux of carbon dioxide by soil is primarily dependent upon characteristics of the physical environment (e.g., temperature, rainfall) and soil composition (e.g., nutrient availability, chemical properties) (Gathany 2011). Past research demonstrates significant relationships among soil efflux, moisture, and temperature. Fire can drastically influence these parameters by eliciting changes in the structural and functional attributes of forest ecosystems (Neary et al. 1999). As above ground vegetation is burned, nutrient input in the form of leaf litter and dead vegetation is altered, modifying the nitrogen to carbon ratio in soil, thus affecting responses of flora and micro-fauna. Both burning and decreased input of leaf litter reduce organic matter at the soil surface, consequently augmenting solar heating (Hamman et al. 2007). Loss of vegetation also results in changing rates of evapotranspiration, shifting the water budget and altering moisture levels in the soil (Neary et al. 1999). Ultimately, due to the effects of the heat of the burn along with changes in moisture and temperature, fire causes a number of changes in the physical (e.g., bulk density) and chemical (e.g., nutrient availability) properties of soil (Certini 2005).

Considering fire-induced changes in soils, this study documents changes in CO_2 flux in recently burned forested areas. During fall 2016, roughly 55,300 acres of forest in Western North Carolina were burned as a result of over 30 wildfires (Off 2016). This presented a unique opportunity to study responses of forest-ecosystem soils to fire disturbance. We conducted this study at Cliffside Lake Recreation Area, Highlands NC—a Montane Oak-Hickory and acidic cove forest—in both burned and un-burned plots. Monitoring the changes in soil CO_2 efflux post-fire in this instance, will explicate our understanding of the relationships between disturbance events, forest responses, and carbon cycling as a whole. This preliminary research is intended to generate data that will inform future hypothesis-driven studies that will be aimed at identifying causal mechanisms driving variations in CO_2 efflux among sites (e.g, microbial activity, vegetative productivity, species succession).

METHODS

Site Selection

In April 2017, we chose two sites within the Nantahala National Forest to study changes in structural and functional attributes of forest ecosystems post wild-fire. The areas were part of the fall 2016 Camp Branch wildfires that covered approximately 45 hectacres (Fig. 1, 2). Burned and unburned sites, accessed at Cliffside Recreational Area, were chosen because they are both representative of typical vegetation in the area and are easily accessible. Four 30 X 30 m sampling sites were established, two in the burned area and two in the unburned area. Visual assessment of the sites showed clear evidence of the fire as canopy cover and vegetation density were noticeably less in the burned areas (Table 1).

TABLE 1: Mean canopy cover data in burned and unburned sites sampled by Kelder Monar.

Treatment	**Canopy Cover (% openness)**
Unburned	13.2725
Burned	20.4871

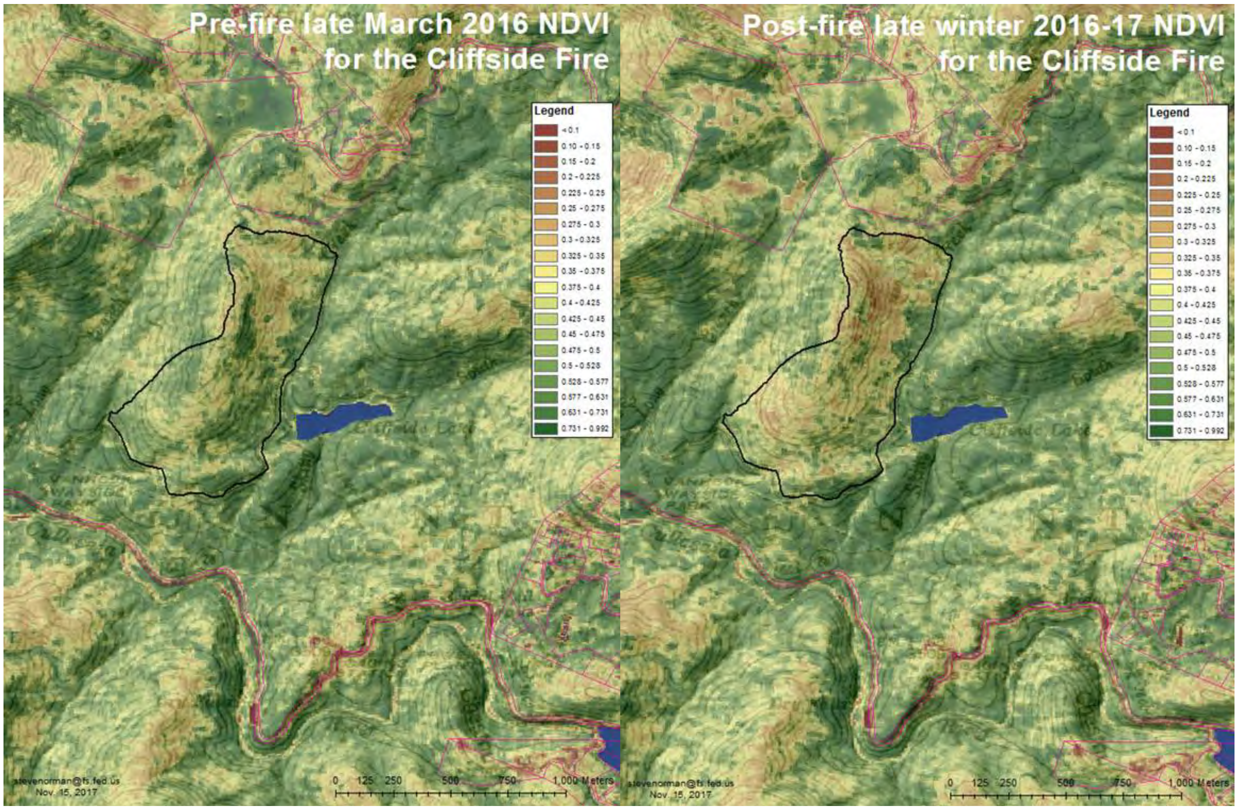

FIGURE 1: Comparison of NDVI maps pre- and post-fire. Maps produced by Steve Norman of the USDA Forest Service.

Soil Respiration

Soil respiration rates were measured on the burned and unburned plots were taken using the LiCor LI-8100A Soil Gas Flux System. This system measures the CO_2 flux of the soil, an indicator of soil properties including microbial activity (Weil and Nyle 2017). In each sample plot, six cross sections of PVC pipe, 10 cm in diameter by 10 cm in height, were inserted 4-6 cm into the ground to serve as respiration sampling points. The PVC collars served to ensure no gas leaked from the chamber as the LiCor measured atmospheric CO_2 concentrations at the soil surface (Kanemasu, Powers, & Sij, 1974). We began weekly data collection at these 24 points (six sampling points at each of four plots) in August 2017 and continued for 10 weeks. Sampling was typically conducted in the late morning to early afternoon – and was postponed during rainfall events. Due to ease of access, we visited the unburned sites first, and the burned sites thereafter.

In addition to the six PVC collars installed at each plot, in September 2017, three trenched sampling points were installed on each plot. These consisted of a PVC collar inserted into the ground surrounded by a square, 1 m^2 trench with a depth of 10-cm. These points were added to provide a corrected value for CO_2 concentration that excluded respiration by fine roots. Removing soil and roots from around the PVC collar causes fine roots inside the collar to decay, after which respiration values can be used to determine an average percentage of gas flux that is attributed to roots. This corrective metric can then be used to distinguish CO_2 efflux attributed to microbial activity from that which is produced in vegetative respiration (Díaz-Pinés et al. 2008).

Soil Moisture

The LiCor LI-8100A also measures soil temperature and moisture. These values were collected at each sampling point weekly using the LiCor temperature and moisture probes. Each probe was inserted into the soil 5 cm from the PVC collar, and collected measurements for the duration of the CO_2 flux measurement.

Temperature

Thermocron DS1921G-F5# iButton® temperature data loggers were deployed at each sampling point (6 sampling points X 4 plots). Each iButton was placed 5 cm from the PVC collar just above the soil surface. To discourage interference by investigatory animals, sensors were housed in PVC casing. iButtons collected temperature data points every 30 minutes for the duration of their deployment.

RESULTS

A total of 153 measurements of CO_2 efflux were taken during the 10-week sampling period. Due to logistics and equipment malfunctions, data from four weeks were not included in this analysis. Additionally, for the date of September 30 (sample 3), data from the burned sites were not available. In total, 149 measurements of CO_2 efflux and soil moisture spanning a 6-week period were included in analysis (Table 2).

Analysis of variance of burning and date as factors affecting soil CO_2 efflux showed both burn treatment ($p = 0.0299$) and day of collection ($p < 0.0001$) had a significant effect on the soil CO_2 efflux. While both burn treatment and day of collection affect soil CO_2 efflux, there was a statistically significant interaction between the two factors ($p = 0.0290$), thus there is no clear trend in soil CO_2 efflux relative to burn treatment and sample date.

TABLE 2: Mean soil CO_2 efflux and surface moisture by date of sample collection and burn treatment. On each sample date, 12 measurements of each metric, soil CO_2 efflux and surface moisture, were taken at both burned and unburned sites.

	Soil CO_2 Efflux (μmol/m^2)		Soil Surface Moisture (% H_2O)	
Sample Date	Unburned	Burned	Unburned	Burned
15-Sep	3.15583	2.741	0.66367	0.5554
21-Sep	3.265	3.10167	0.61542	0.49708
30-Sep	3.56722	*	0.71922	*
5-Oct	1.50438	2.04813	0.82506	0.65894
13-Oct	3.90889	3.00278	0.69067	0.59106
26-Oct	1.29667	1.45455	0.94158	0.789

* Data unavailable

The overall mean soil CO_2 efflux was higher in the unburned treatment than the burned (Table 3), with the mean efflux higher in the unburned sites on sample days one, two, three and five. On the fourth and sixth sample dates, mean soil CO_2 efflux was higher in the burned sites (Fig. 2).

TABLE 3: Mean soil CO_2 efflux and moisture over duration of sampling period.

Treatment	n	Flux (µmol/m^2)	Moisture (% H_2O)
Burned	67	2.49925	0.61761
Unburned	79	2.73506	0.74697

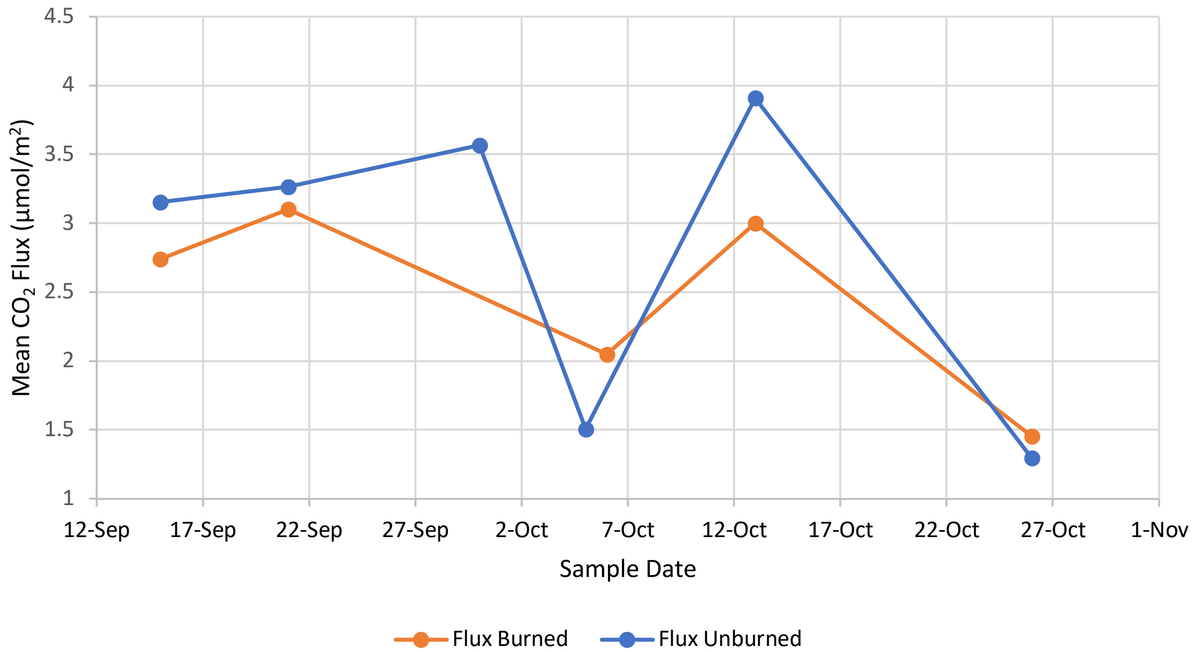

FIGURE 2: Mean soil CO_2 efflux was greater in the unburned sites on sample days one through three and five, while efflux was higher in the burned sites on days four and six. There are points of interaction between mean soil CO_2 efflux of the burn treatments through the fourth to sixth sampling days.

Simple linear regression analysis was conducted to determine the relationship between soil CO_2 efflux and soil moisture. In analysis of the burned treatment sites, 67 sample points were used to determine an inverse relationship between efflux and soil moisture. The relationship was significant ($F = 39.8$, $p = 0.04$) and 38% of our variation in flux was explained by variation in soil moisture ($R^2 = 0.3798$) (Fig. 3). For the unburned treatment, 79 sample points were used to also find an inverse relationship between efflux and soil moisture. While this relationship was present, it was weaker than that of the burned sites with 18% of the variation in efflux attributed to changes in moisture ($R^2 = 0.1774$) (Fig. 4).

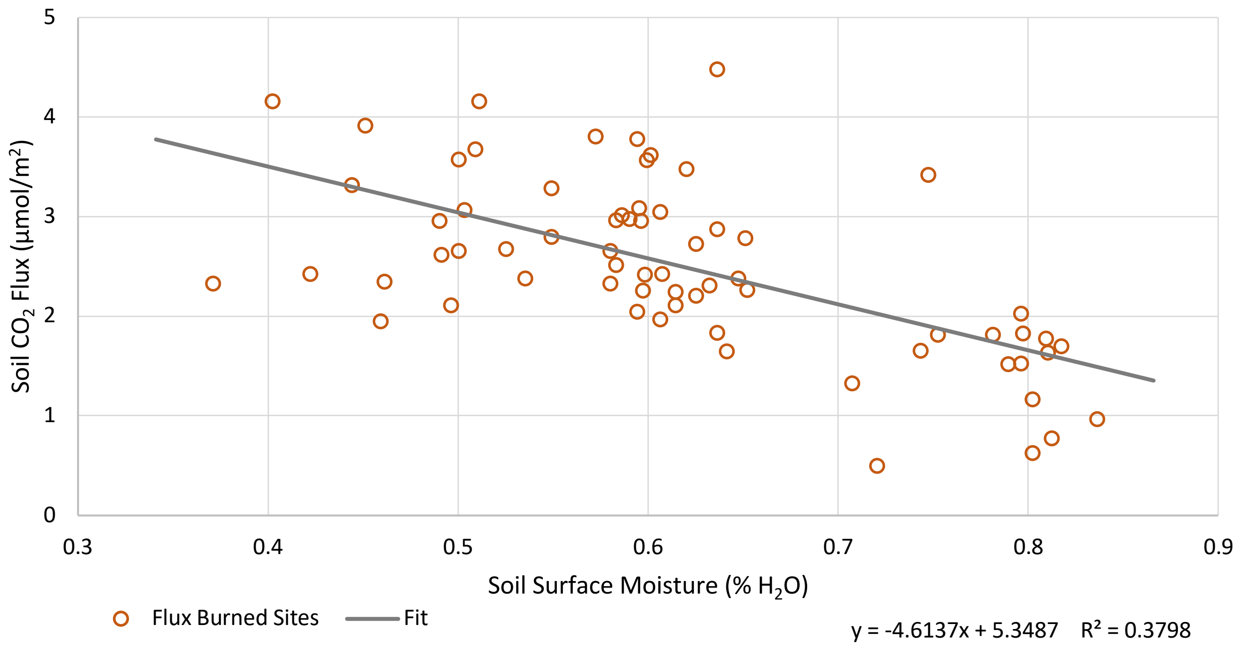

FIGURE. 3: Linear regression plot for $C0_2$ efflux dependent on soil moisture levels in the burned plots.

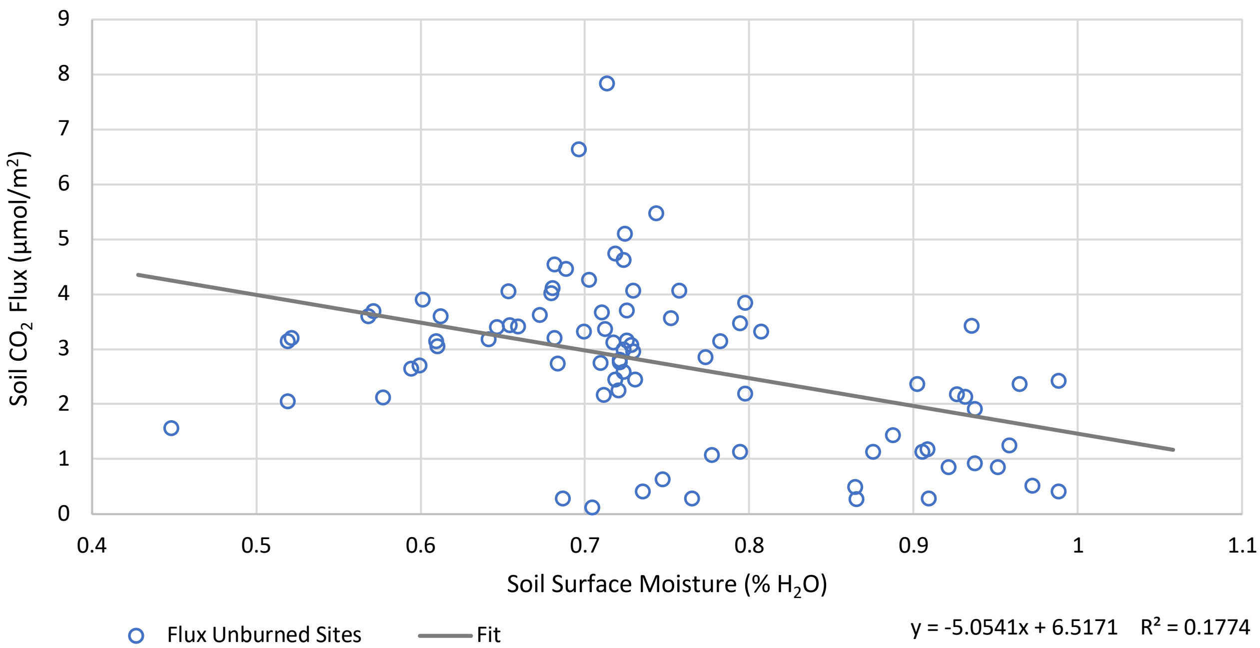

FIGURE 4: Linear fit plot for efflux dependent on soil moisture levels for unburned treatment.

Over the entire sample period, the unburned sites had a higher moisture content than the burned sites (Fig 5). In an assessment of the variation of soil CO_2 efflux with soil moisture, each treatment demonstrated a brief increase of efflux with moisture increase; however, past that point, efflux subsequently dropped sharply with increase in moisture (Fig. 6). In the unburned plots, reduction in CO_2 efflux showed minimal correlation with spikes in soil moisture (Fig. 7).

The burned sampling sites demonstrated a stronger relationship between soil CO_2 efflux and soil moisture as increases in soil moisture were reflected in declines in efflux (Fig. 8).

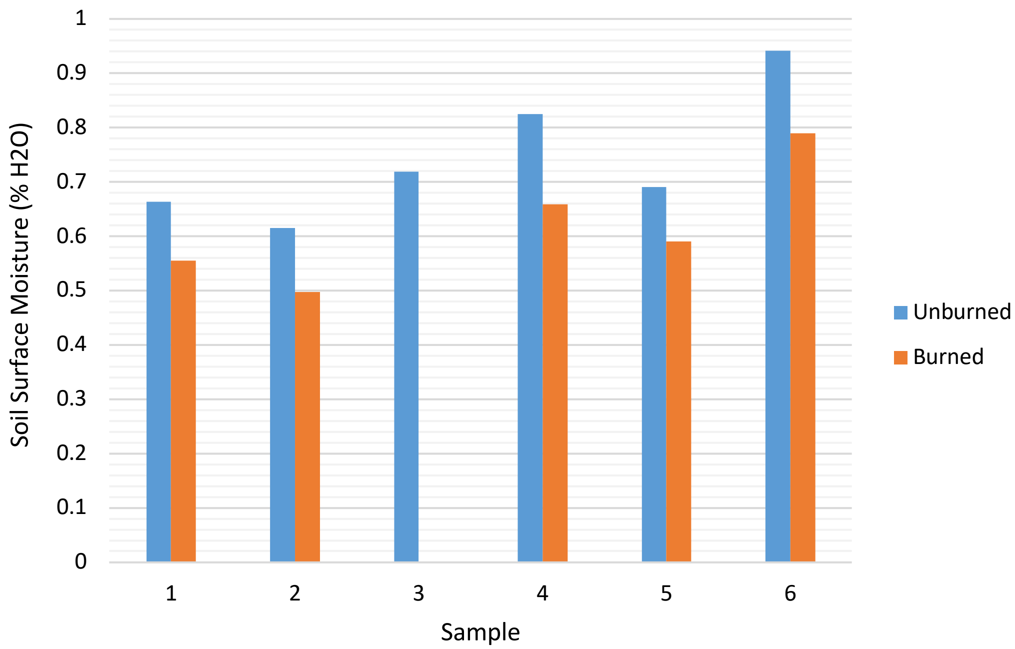

FIGURE 5: Mean soil surface moisture across 6 sampling events on burned and unburned plots at Cliffside site.

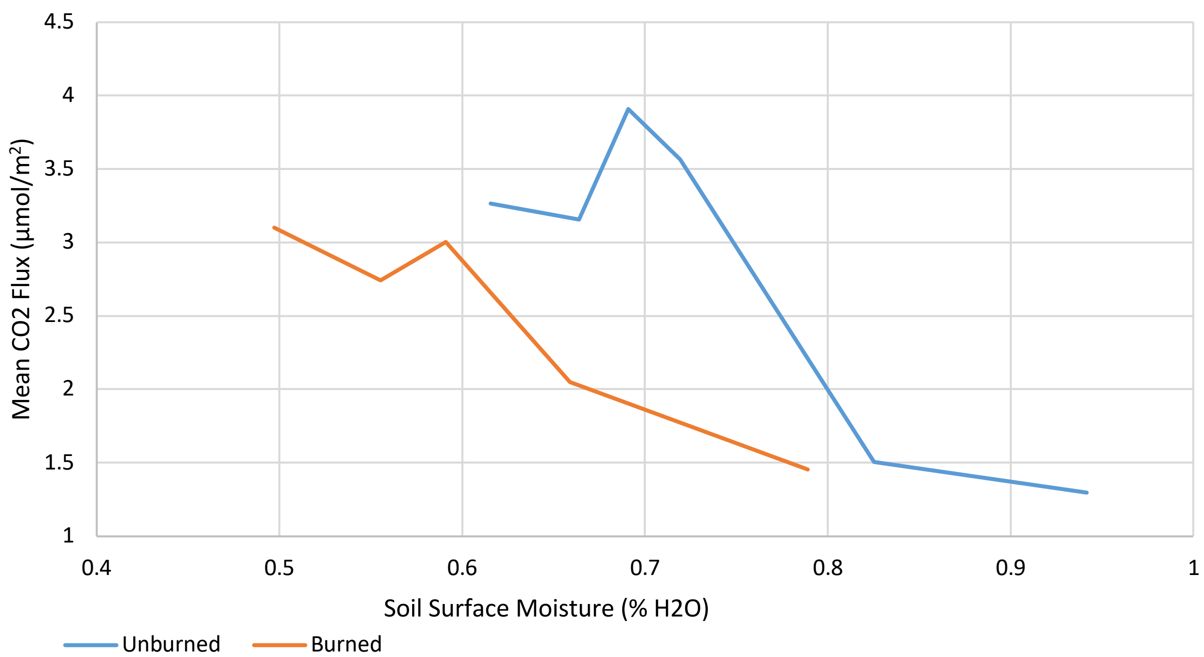

FIGURE 6: Mean soil CO_2 flux in µmol/m^2 in relation to percent soil moisture on burned and unburned plots at Cliffside site.

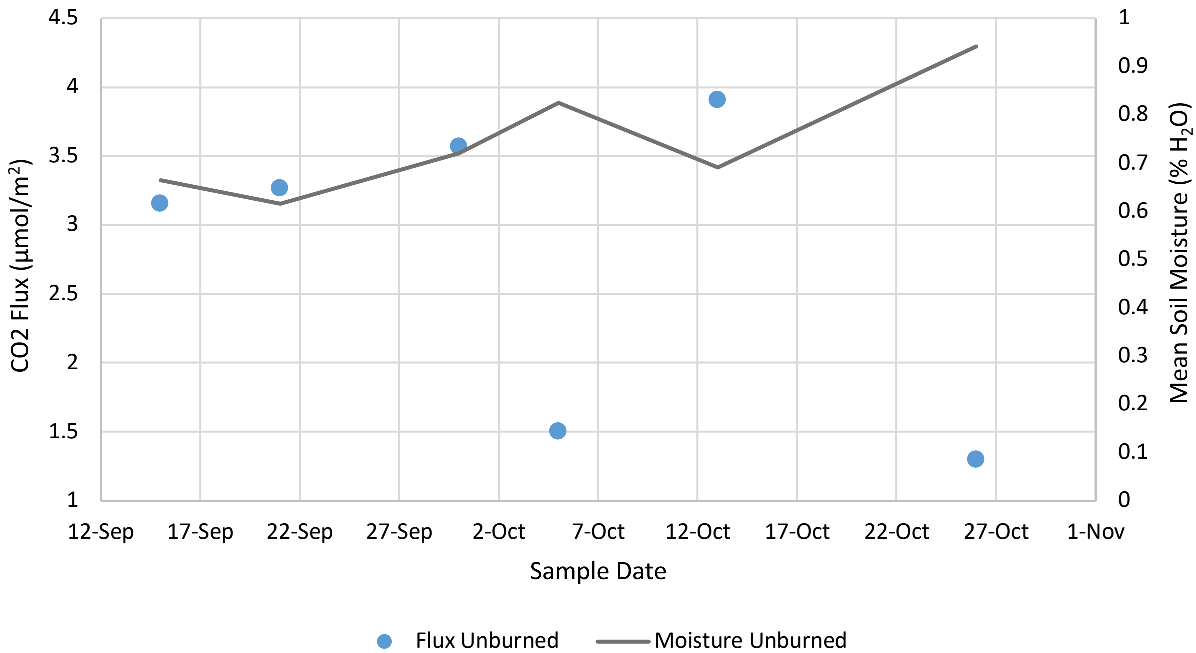

FIGURE 7: Variation of soil CO_2 flux with soil surface moisture measurements taken on each sampling date at unburned sites.

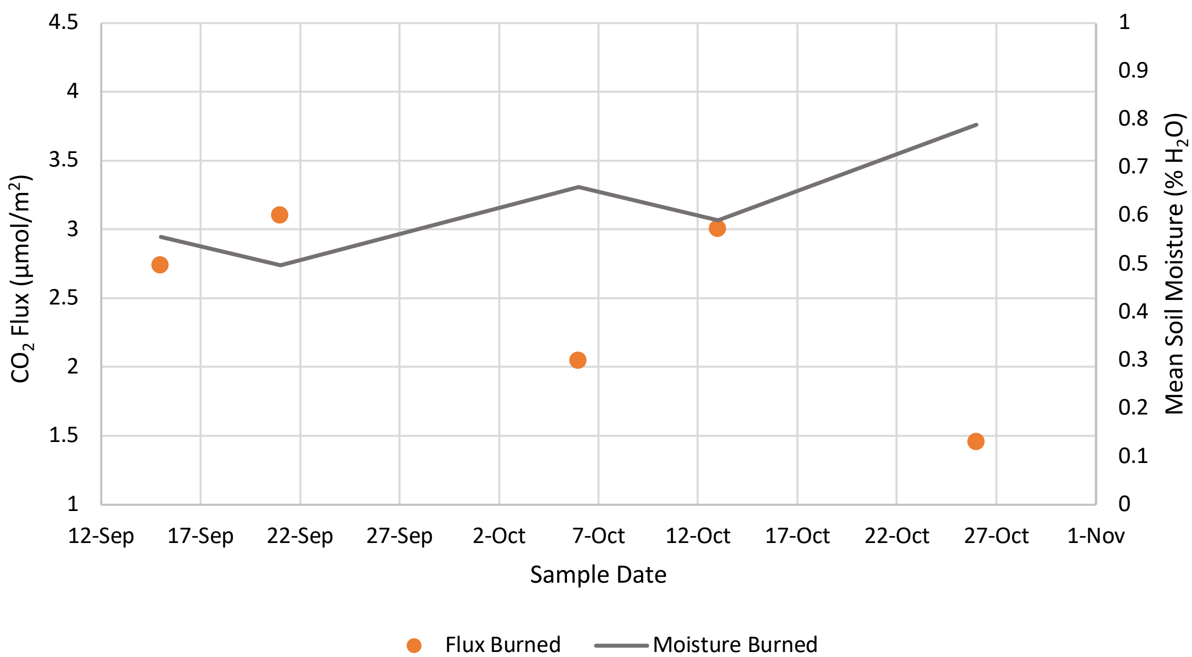

FIGURE 8: Variation of soil CO_2 flux with soil surface moisture measurements taken on each sampling date at burned sites.

Over the data collection period, air temperatures at time of efflux measurement were consistently lower in the unburned sites than in the burned sites (Fig. 9). In the unburned sites, regression analysis described a very strong relationship between temperature and efflux, with 30% of variation in temperature associated with variation in temperature ($F = 18.7904$, $p < 0.0001$). In the same analysis of the burned treatment, only 11% of the changes in efflux were attributed to variation of temperature ($F = 4.7528$, $p = 0.03519$).

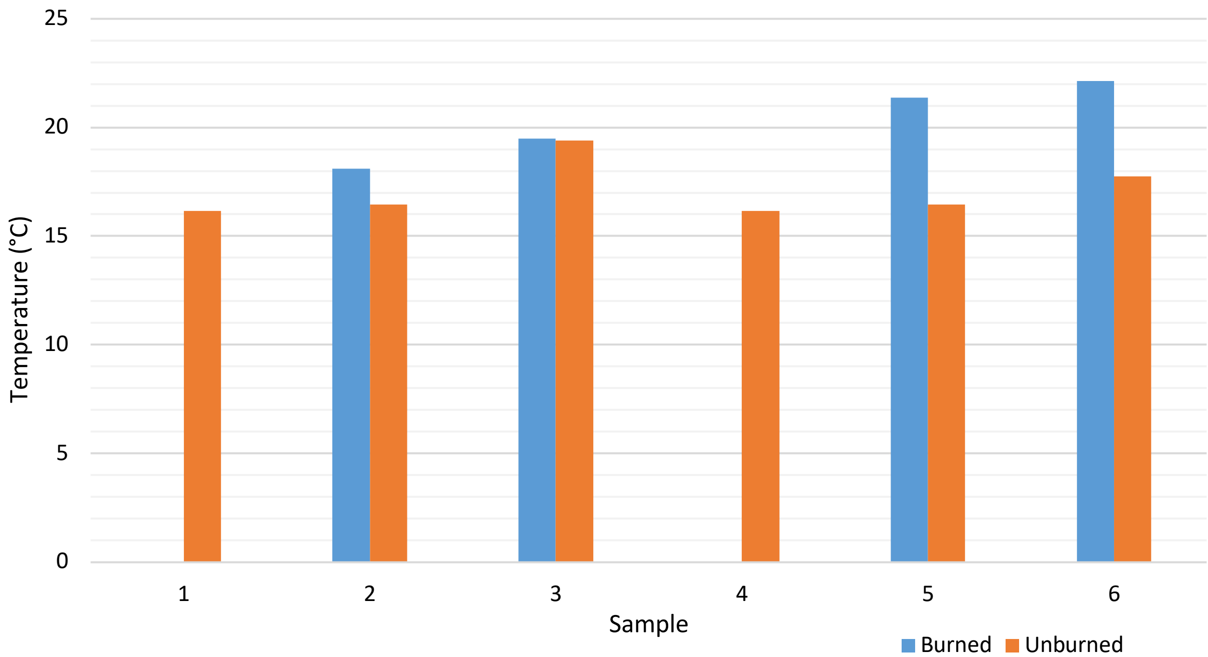

FIGURE 9: Mean temperatures at burned and unburned sites at time of CO_2 efflux measurement.

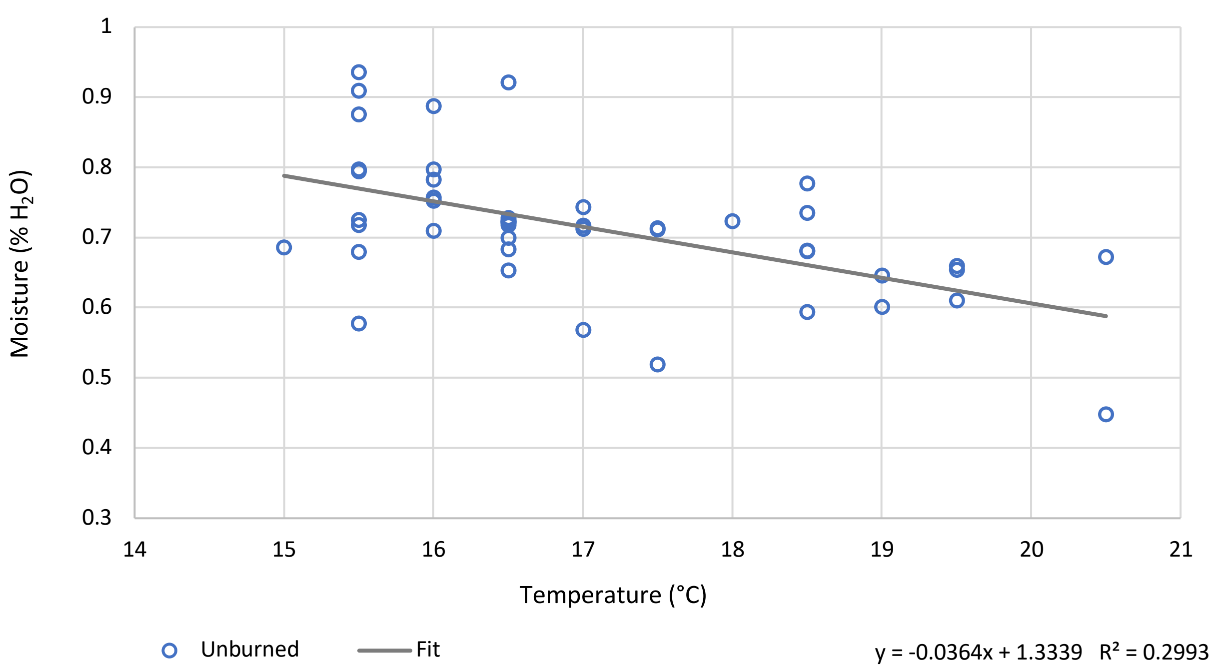

FIGURE 10: Variation of soil surface moisture with temperature in unburned sites.

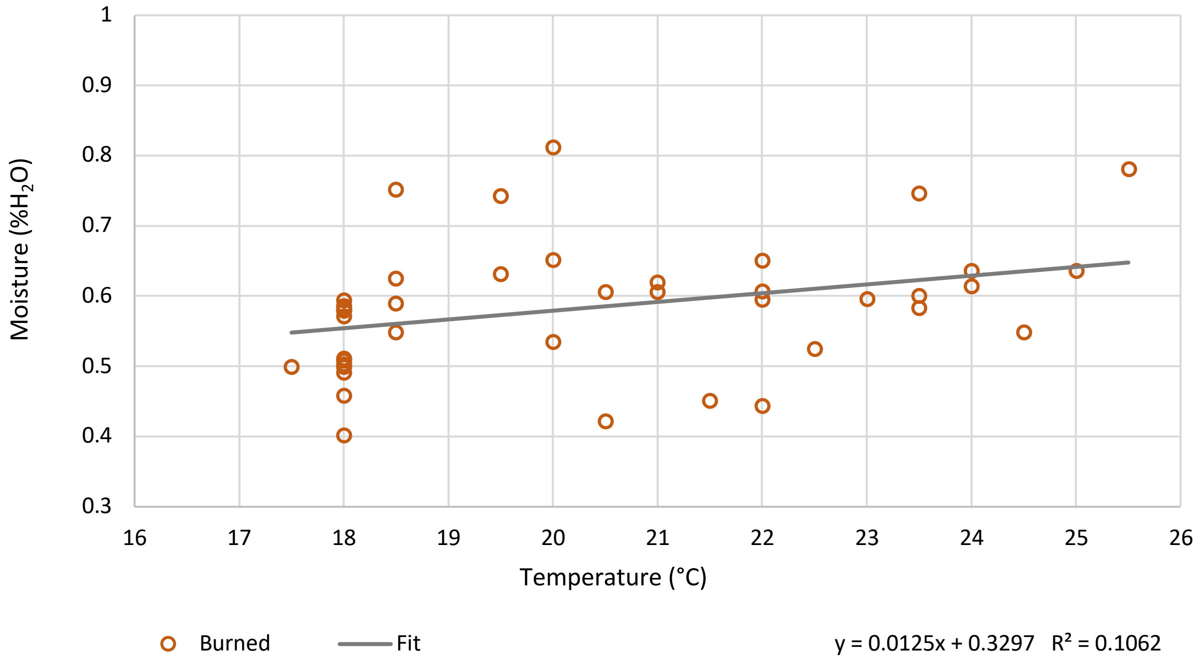

FIGURE 11: Variation of soil surface moisture with temperature in burned sites.

DISCUSSION

With a limited data set, conclusive trends in causal factors post-fire and CO_2 efflux cannot be determined. We can say, however, that efflux was more dependent on soil moisture in burned sampling sites relative to unburned sites. Past studies have frequently found soil moisture to be positively correlated with CO_2 efflux; consequently, water availability is often considered to be a limiting factor in CO_2 efflux and biotic activity (Cook 2009). Conversely, we observed an inverse relationship between soil moisture and CO_2 efflux, as peaks in soil moisture above a certain threshold (approximately 0.6% in the burned sites and approximately 0.7% in the unburned sites) corresponded with sharp decreases in efflux. This supports a hypothesis that soil moisture has an optimum level, after which saturation of pore space becomes too great, ultimately restricting biotic activity. It is likely that soil moisture exceeded the range ideal for CO_2 efflux on some data collection dates, contributing to restricted levels of efflux on days proximate to rainfall events during the sampling period.

In interest of an explanation for sample days when CO_2 efflux was higher in the burned plots, we can look at the soil moisture levels on those dates. Relative to the remaining sampling days, on collection days 4 and 6, when CO_2 efflux was greater in the burned plots than in the unburned, soil moisture levels significantly increased in both treatment sites. Supporting the observed relationship between variation in CO_2 efflux and changes in soil moisture, peak points in soil moisture are mirrored in drops in efflux in the burned sites. This relationship was not evident in the unburned sites, as soil moisture and CO_2 efflux do not demonstrate the same relationship.

While it appears that soil CO_2 efflux in the unburned sites is dependent on some dominant factor (or factors) other than soil moisture, our observations unexpectedly show that temperature is not directly influential on efflux either. Other studies have found temperature to have some degree of influence on CO_2 efflux (Barron-Gafford 2011), but we found there to be no relationship between temperature at time of data collection and soil CO_2 efflux in neither the unburned sites nor the burned sites. There was a relationship found between moisture in temperature, a trend that was expected. What was unforeseen, however, was the strength of the variation in the two treatments: the variation in moisture due to temperature in the unburned sites was almost 3 times that of the burned sites. We would have anticipated the inverse of this outcome, considering the greater number of factors that would stabilize moisture in the unburned sites. We conjecture that moisture may be less dependent on temperature in the burned sites due to the lack of organic matter at the soil surface within the A horizon. Consequently, the efficacy of soil to retain moisture may be reduced, as moisture drains faster through the upper horizons of the soils in the burned plots (Krenz pers. comm.). It is also possible, given the small data set, that the results are simply erroneous.

After a forest is burned, many of the factors that stabilize the physical environment of forest soils are affected. Soil temperature and atmospheric temperature, for example, are typically less stable post-fire due to several factors: reduced leaf litter, decreased canopy cover, and lower albedo. Fire can diminish or completely remove the layer of organic matter at the soil surface depending on the severity of the burn, the role of which is to act as an insulator, contributing to the soils ability to retain both heat and moisture. Additionally, reduction of canopy cover has a compounding effect of increasing exposure of soils to direct sunlight, and thinning dense canopies which block rainfall and lose water to evapotranspiration. The blackening of surfaces due to burning also decreases albedo, the ability to reflect sunlight, of forest surfaces. This may also contribute to fluctuations in temperature post-fire (Gathany 2011).

Due to the limitations of our data set and the experimental design, it's difficult to determine the extent to which soil CO_2 respiration has fluctuated after the forest fires at these sampling sites. Unfortunately this study was limited in that the area burned in the fall 2016 fires are ecologically different from those that remain unburned. For one, the two treatment sites are located on different slopes, each with a different aspect and gradient. We know that slope aspect can greatly influence the composition of a forest community, especially the moisture; north-facing slopes tend to be wetter, while south facing sloped tend to be drier. In this case, however, fire may also be a causal factor in moisture fluctuation, so statistically eliminating this factor would confound analysis of burn effects. The sampling sites also were of different forest community types in which the dominant vegetation types likely also affect the soil respiration (Krenz pers. comm). Ideally we would have been able to use the data collected from the trenched plots to control for vegetative differences to an extent, but due to length of time necessary for efflux to reflect the removal of fine roots, it was not included it analysis (Díaz-Pinés et al. 2008).

Ultimately, experimental design restricts the certainty with which we can explain the trends observed. The true metric we are seeking to measure through CO_2 efflux is biotic activity, however it is difficult to identify changes in the soil microbiome when we are observing it indirectly. This experiment provides insight into areas for future study: microbial activity, organic matter levels, and fungal communities. An analysis of these factors can complete our picture of the post-burn changes to forest functional dynamics and supplement our knowledge of forest carbon cycling as a whole.

ACKNOWLEDGMENTS

I would like to thank Dr. Trip Krenz for his guidance on this project, and Dr. Beverly Collins for her continuous technical support and assistance. Dr. Sarah Workman was also an integral part of this project through her patience in late evening office discussion and her coordination of the Highlands Biological Station's Instititue for the Environment field site program. Also, I cannot forget to include Dr. Jim Costa for his direction of the program and continual positivity.

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ECOLOGY OF POSTPARTUM FEMALE TIMBER RATTLESNAKES (*CROTALUS HORRIDUS)* IN THE SOUTHERN APPALACHIAN MOUNTAINS

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Abstract. Timber rattlesnakes (*Crotalus horridus*) are not extensively researched in the Southern Appalachian Mountains, resulting in a deficit in knowledge about their ecology. Due to differences in elevation and climate, there is an expected distinction from other populations in the United States (Northern and Western), which is the basis of our study. Here we discuss the ecology of two postpartum adults, affectionately called Irma and Orbit. As they moved from initial gestation site to overwintering site, the pair were monitored using a radio transmitter externally attached to the rear scales. Radio telemetry was utilized to track the snakes weekly over the course of three months. Both snakes were located, tagged, and tracked in the Warwoman Wildlife Management Area in Rabun County, Georgia between August and November 2017. The two snakes traveled down the mountainside to their primary foraging range, where they alternated between resting and hunting before migrating to an overwintering den (hibernacula). Their overwintering habitat and overall foraging range was analyzed using a minimum convex polygon. Our findings serve as a pilot study in examining the ecology of the Southern Appalachian timber rattlesnakes.

Key words: Crotalus horridus, *hibernacula, overwintering, postpartum, Southern Appalachian Mountains; timber rattlesnake*

INTRODUCTION

Timber rattlesnakes (*Crotalus horridus*) are members of the pit viper subfamily of vipers (*Viperidae: Crotalinae*), distinguishable by a distinctive facial sensory pit organ. The timber rattlesnakes are found on the eastern side of the United States, with a known range spanning from Southern Maine to Northern Florida, and extending to the Midwestern states as far as eastern Oklahoma and Kansas (Schmidt and Davis 1941). They are characterized by dark colored V-shaped bands on a skin color ranging from yellow to dark brown. Occasionally, individuals are very dark, sometimes black (Conant 1975). The Canebrake (*C. horridus atricaudatus*) was placed as a junior synonym of *C. horridus* in the Integrated Taxonomic Information System in 1801 (Latreille and Sonnini). In a study conducted by Pisani et al. (1973), it was determined that no designated subspecies could be determined. A study focusing on the phylogeography of timber rattlesnakes using mtDNA sequences from the northern, southern, and western regions of the United States confirmed that there are no evolutionary separations that would support a subspecies (Clark et al. 2003).

There are a myriad of factors that may affect the timing of fertilization and the actual birthing in the timber rattlesnake reproduction process. Females reach sexual maturity around seven to eleven years and males between four and six years (Martin 1993; Orianne Society 2017). Due to the high energy cost in reproduction, females tend to reproduce once every four or five years in the north, and every two or three years in the south (Martin 1993). Compared to males, females appear to be passive participants in the process of searching for a mate (Reinert and Zappalorti 1988). The mating process occurs in the late summer (Martin 1993), but the females can carry sperm and delay fertilization until the following early summer months. During this time, the gravid females search for ideal gestating habitats in the form of rock balds or outcrops. All rattlesnakes are ovoviviparous, meaning that the egg is retained within the mother and the young are birthed fully functional without a shell. The gravid females typically give birth in the late summer and early autumn months to 4-11 young (Orianne Society 2017). They stay with their young until their first shed, in which the young disperse to hunt before migrating to an overwintering den.

According to a study conducted by Reinert et al. (2011) on Northeastern timber rattlesnake diets, food habits varied widely within relatively small geographic distributions, and may not reflect prey species' availability. Variation in diet and behavior among the snakes can be due to their geographic location, their size, and age. The snakes that are found further south and higher in elevation in the United States tend to eat three types of mammals most frequently: New world mice (*Rodentia cricetidae*), cotton rat (*Rodentia Sigmodon hirsutus*), and cottontail rabbits (*Sylvilagus Oryctolagus cuniculus*) had the highest percentage of occurrence. The diet of juveniles tended to center around the consumption of shrews (family Soricidae), whereas the adult's dietary habits showed higher consumption rates of New world mice (Clark 2002). According to a study conducted in 2002 (Clark), timber rattlesnakes tend to focus primarily on endotherms for consumption. This study also observed that *C. horridus* opportunistically feeds on whatever small mammals are available.

Timber rattlesnakes are found in all types of habitat ranging from the Coastal Plain to the forested regions of the mountains; however, their populations are severely reduced in areas of urbanization and agriculture, and therefore tend to be found in mountainous forests (Mohr 2010; Sealy 2002). Our study focuses on the northeastern mountains of Georgia, where hardwood and pine forests are the dominant vegetation type, and rocky balds exist on the sides of the mountains.

Balds typically comprised of mostly gneiss and schist with exposed rocky terrain, are ideal sites for gravid timber rattlesnakes. The shallow hollows underneath the rocks have a reduced temperature fluctuation with mean temperature of around 26 °C which is optimal for birthing (Orianne Society 2017). Gravid females are semi-stationary once they find a birthing site, and stay within the area with their young for about ten days, or until the first shed of the young. Male and nongravid female timber rattlesnakes have a large range, some territories spanning around 200 hectares of land.

According to a study conducted by Adams in the forested mountains of West Virginia, overwintering habitats include crevices and ledges of rock outcrops (2005). Additionally, these hibernacula locations occurred below the frost line to allow for overwinter survival, which is further supported by research in New York (Adams 2005)**.** A hibernaculum is a place in which an animal seeks refuge to overwinter. Research in the northern United States on overwintering habitats states that dens are typically on "open, steep, south-facing slopes with rock fissures or talus surrounded by hardwood forests" (New York DEC 2000). For the Southern Appalachian Mountains, there is no information about timber rattlesnake overwintering habitat types, time of hibernation, or overwintering location. Because the Southern Appalachians comprise a large portion of timber rattlesnake's southern range, this information can assist in the conservation of the species.

The overall goal of this project was to determine the location of overwintering sites relative to gestation sites and the size of and foraging ranges of postpartum female timber rattlesnakes in the Southern Appalachian Mountains to better understand their movement and ecology. A secondary objective was an evaluation in the efficiency of the commercially available transparent film dressing, Tegaderm, in the external application of radio transmitters on the snakes. An assessment of the Tegaderm's effectiveness in this preliminary study may lead to its use in future projects.

METHODS

Study Location

We sampled in Warwoman Wildlife Management Area in Rabun County, Georgia between August and November. No specific coordinates are given for the safety of the rattlesnakes. This area consists of roughly 15,800 acres (6394 ha) of protected public land. The habitat was primarily mixed, relatively new-growth deciduous and coniferous forest with interspersed rocky outcrops. Only a handful of trees appeared to be old-growth, indicative of prior logging. There were significant rhododendron (*Rhododendron catawbiense)* and buckberry (*Vaccinium stamineum)* thickets, as well as ample amounts of leaf litter and rotting logs throughout the sites.

Gestation Site Identification

Using aerial imagery during dormant months, rocky balds were identified throughout the study area by Orianne Society staff. These sites were subsequently surveyed for the presence of gravid females from 2012 to 2017. A subset of these sites were selected as long-term monitoring sites. In August 2017, we captured one gravid female each from two of these monitoring sites. Initially, we planned to study twelve snakes, but a hurricane impeded the number of sites that were accessible.

Radio Telemetry

We attached transmitters to two adult females at their birthing sites and tracked them using radio telemetry. Gravid females were targeted because they were expected to be in the pre-scouted gestation sites, but one male was also tagged opportunistically. However, his transmitter failed within one week, and we were unable to track him to his overwintering site. We then tracked them throughout their journey in the Warwoman Wildlife Preserve. As we were monitoring their movements for only a few months, we decided to attach transmitters externally, as opposed to surgically implementing them, which is often utilized for long-term studies (Mohr 2010). After the initial capture of each snake, the head was put into an open plastic tube. The top scales on a lower section of the snake was cleaned using cotton swabs and rubbing alcohol (fig 1). A transparent film dressing (Tegaderm 3M corp.) was cut and applied to the cleaned area (fig 1). After orienting the radio transmitter so the antenna pointed towards the tail, another cut film dressing was applied, ensuring the transmitter was smoothly and completely covered. The snake was then released back where it was found. GPS coordinates were taken at each site using a Garmin GPSMAP 64st.

Every few days, we tracked the snakes using a radio receiver and an amplification antenna. We entered the unique frequency code of each snake into the receiver and increased the gain, the measure of the receiving range. As per conventional practice, we began on the highest value and decreased the gain as we approached the transmitter. We then determined the direction from which the transmitters were broadcasting by the clarity and volume of the beeps (e.g., the louder the beeps are indicative of a stronger signal). We then hiked in the direction indicated by the transmitters, adjusting gain and direction as required. When the gain was reduced, it helped determine the origin of the loudest beeps. A higher gain, or a broader receiving range, would begin to pick up the transmitting signal from all directions as we neared the transmitting location. A lower gain or smaller receiving range would limit the excess noise and indicate the correct location of the snake.

When on the lower receiving range, we would being to scan the ground for the snake. Once we located the snake, we recorded the GPS coordinates.

Analysis

We collected GPS data points at the initial sites at which we attached radio transmitters to the snakes, each time we tracked and located the snakes, and at their overwintering sites. We used the snakes' location to define the postpartum foraging range for each snake using minimum convex polygons, which are the smallest polygons that encompass all of the snake's locations. Within ArcGIS 10.5, we analyzed the distances traveled using elevation masks. We also used ArcGIS 10.5 to create a map that displays the location of gestation and overwintering sites, foraging ranges (minimum convex polygons), and snake locations. Within our map, we used Universal Transverse Mercator mapping units on a Mercator projection in the North American Datum of 1983 for the Georgia State Plane.

RESULTS

The snakes were captured within the Warwoman Wildlife Management Area above 3,000 feet in elevation (fig. 2). We found that both snakes left their gestation sites and moved to lower elevation sites for foraging (table 1). The size of each foraging ranges were 10,558 m^2 and 13,243 m^2, respectively (table 1). Both snakes had similar movement patterns, but Irma traveled greater distances (table 2, fig. 3, and fig. 4). The snakes, when found during tracking, were typically in either a foraging position (fig. 5) or sunning themselves.

FIG. 1. A photograph of Orbit with a transmitter attached using Tegaderm. Taken on 5 Oct 2017.

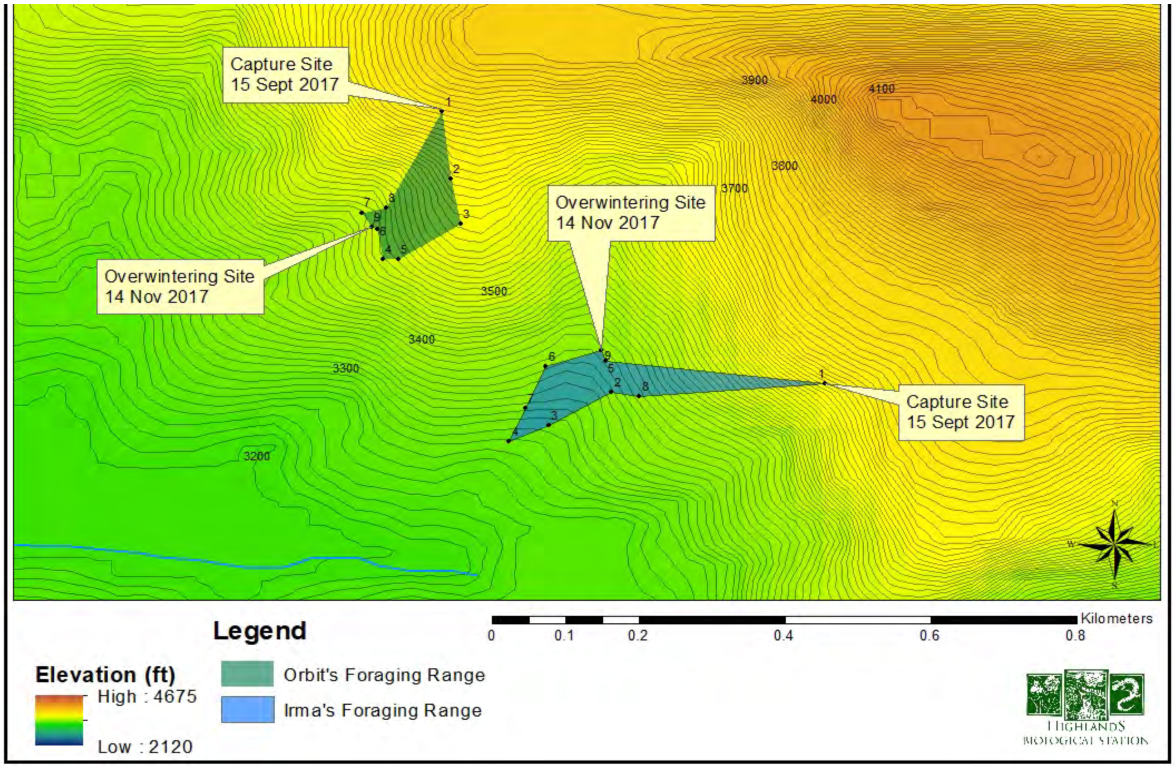

FIG. 2. A minimum convex polygon analysis of movement patterns of two timber rattlesnakes, tracked in Rabun County, GA.

TABLE 1. A summary of the movement of two timber rattlesnake (*C. horridus*) females tracked in the Warwoman Wildlife Management Area (Rabun Co., Georgia) between August and November 2017.

	Irma	Orbit
Elevation at gestation (m)	1,093	1,124
Elevation at overwintering (m)	1,023	1,049
Gestation to overwintering distance (m)	70.1	75.0
Total distance moved (m)	292.0	202.1
Polygon area (m^2)	10,558	13,243

TABLE 2. A summary of the change of movement per day of two timber rattlesnake (*C. horridus*) females tracked in the Warwoman Wildlife Management Area (Rabun Co., Georgia) between August and November 2017.

Date	Irma (m/day)	Orbit (m/day)
9/15	0.0	0.0
9/22	34.3	13.9
9/26	81.4	16.2
10/2	62.6	16.9
10/4	266.6	9.0
10/10	100.1	8.1
10/12	332.3	14.7
10/17	158.9	5.5
11/14	31.2	1.1
Average	133.4	10.7

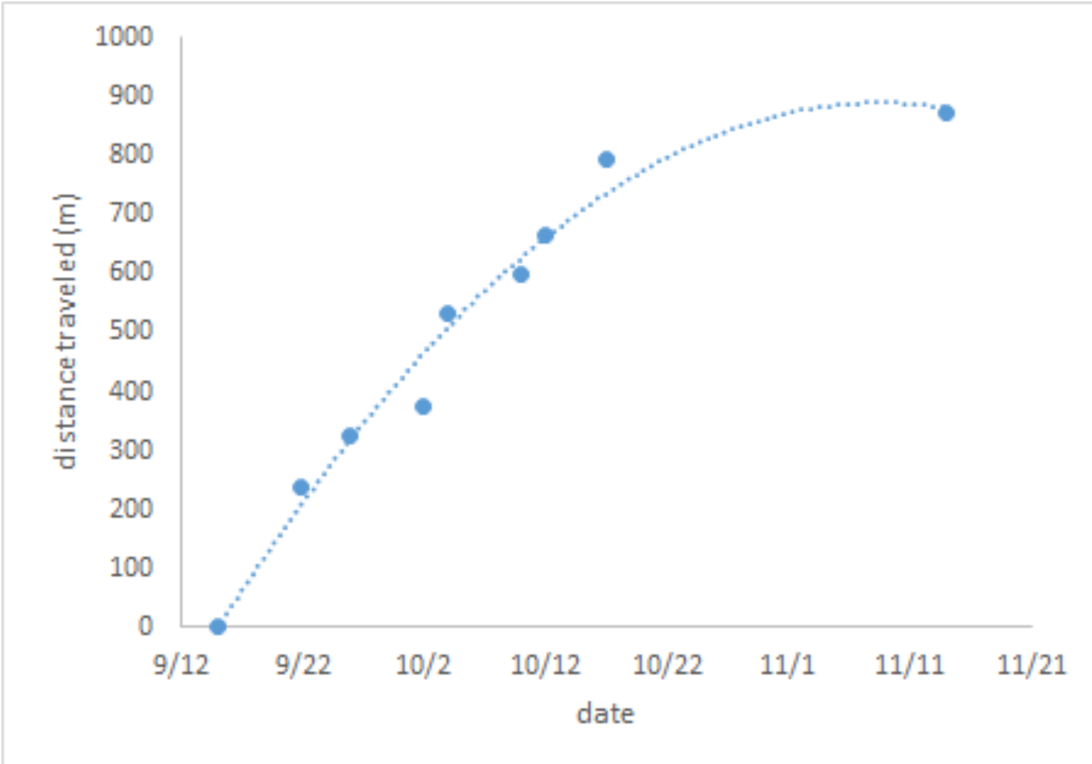

FIG. 3. A graphical representation of Irma's movement throughout the study.

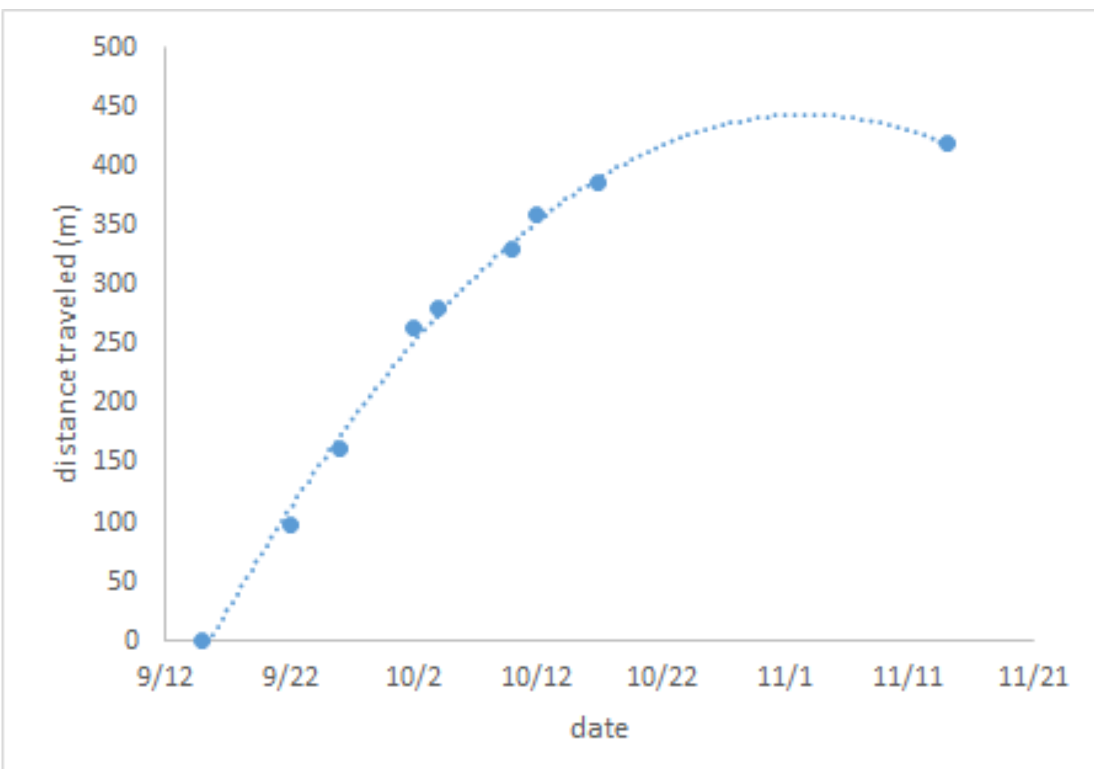

FIG. 4. A graphical representation of Orbit's movement throughout the study.

Fig. 5 A photograph of Irma in a foraging position on a log, adjacent to a stream. Taken on 13 Oct 2017.

Hibernacula Habitat

The hibernacula dens used by the rattlesnakes in this study appear to be burrows constructed by local animals (fig. 6 and 7). The final overwintering site for both snakes was dominated by deciduous trees with a sparse dispersal of conifers. Common tree species found at both sites included: *Quercus alba, Q. rubra, Q. coccinea, Q. prinus, Acer rubrum, A. pennsylvanicum, Betula alleghanensis,* and sparse *Pinus strobus*. There was also an ample amount of leaf litter and numerous rotting logs throughout the sites. According to a study conducted at lower elevations in South Carolina, timber rattlesnakes utilize solo hibernacula burrows, which often consists of rotten tree stumps, small rocky areas, or burrows constructed by gopher tortoises or mammals (Morh 2010). Our study found similar hibernacula habitat, resulting in an interesting connection.

DISCUSSION

We initially planned to locate and attach radio transmitters to at least ten adult timber rattlesnakes, of either sex. However our sampling group was limited to select snakes that were not about to shed and/or had already given birth. We focused on postpartum females for this study because they would likely be in predictable locations such as rocky balds. Having a targeted area led to higher chance of encountering snakes, in comparison to searching the forest. These sites were isolated from main highways, forestry roads, and marked trails. Due to several large storms, access to most of these sites became severely restricted. As a result, we were unable visit most of the planned sites in time to capture the postpartum females, which remain with their young for only a few days after their birth. In future studies, there should be better preparation for intermittent weather (i.e.; bringing equipment for debris removal).

FIG. 6. A photograph of Orbit's overwintering den. Taken on 14 Nov 2017.

FIG. 7. A photograph of Irma's overwintering den. Taken on 14 Nov 2017.

Early in our sampling, we attached a radio transmitter to an adult male and later two postpartum females. The transmitters purchased had an expected life of around twelve weeks. The transmitter attached to the male had a malfunction after two weeks, and his location was lost. The transmitters on the two females began to fail within the first month due to an unknown manufacturer error. Fortunately, we were able to track the snakes' weak signals, and attach an additional transmitter on each snake. To avoid future transmitter malfunction, there should be extensive testing before utilization in the field.

The movements of the two females were successfully monitored from gestation sites to overwintering sites, as shown in figure 2. Both of the snakes lost about 60 m in elevation compared to their gestation sites, and foraged within a 10,500-13,250 m^2 area (table 1). The loss in elevation is most likely due to locating an overwintering den below the frost line (Adams 2005). Once the snakes were within their optimal foraging range, they meandered in search of food sources in an overall circular formation (fig. 2), which supports findings conducted by several researchers on timber rattlesnake movement (Reinert and Zappalorti 1988; Mohr 2010). While it is uncertain why the snakes appear to follow a circular path from gestation to overwintering sites, however, this pattern may be due to conspecific chemical signals for mating purposes or avoidance of intraspecific contact and competition (Reinert and Zappalorti 1988).

The overwintering sites for both snakes were adjacent to streams, thus it is likely sites are chosen for proximity to food and water sources. Both sites were on south southwest facing slopes, and had an abundant amount of sunlight. Orbit's final site was on a steep gradient, in what seemed to be a dried up stream bed. Irma, on the other hand, chose an old tree stump, also in a stream bed, as an overwintering location. The locations also had larger boulders and fallen trees sporadically through the landscape. Orbit's final overwintering elevation was 1049 m, and Irma's was 1023 m (table 1)**.** As shown in figures 3 and 4, both snakes had similar movement patterns; however, Irma was shown to move greater distances per day (table 2). This portrays that the snakes moved rapidly down the mountain, and slowed as they reached their respective foraging and overwintering sites. This could be due to a greater resource availability at lower elevations. The snakes appeared to be hibernating independently, but in consideration of past studies, it is possible that other snakes remain undetected. It has been found in studies conducted in the Northeastern United States that timber rattlesnakes are often found in communal dens (Mohr 2010; Reinert and Zappalorti 1988). According to herpetology specialist Dr. Jenkins, Southern Appalachian timber rattlesnakes have been found in such communal dens in previous years, co-habited by other timber rattlesnakes and copperheads (*Agkistrodon contortrix*) (Jenkins, pers. comm.). However, it is unlikely that Irma and Orbit are using communal dens based off their stream bed location as the dens appeared shallow.

When initially approached and observed, the snakes were quite docile. Most often, the snakes were in either resting or foraging position, and made no movement to strike when approached. They remained stationary even when we were within a few feet of them to record an accurate GPS coordinate. When attempting to capture the snakes, their first reaction to move away rather than strike. Only when they were put into small plastic tubes did they begin to strike. This pattern of non-aggressive confrontations was mimicked each time we encountered a timber rattlesnake, male or female. Although not conclusive, we have observed the timber rattlesnake's non-aggressive nature, and their preference to flee when disturbed.

This project serves as a pilot study for the effectiveness of transparent film dressing, Tegaderm, in the external application of the radio transmitters along with the use of radio telemetry on Southern Appalachian timber rattlesnakes. The film held throughout the study and was a simple and effective resource to use in the field. It is a short term tracking method that is expected to fall off after the snakes shed. Tegaderm will likely be used in future studies.

In the future, the Orianne Society further plans to continue research on the winter hibernacula of these rattlesnakes, with the goal of obtaining larger sampling groups in order to collect more data on where the snakes go for winter, how long they hibernate, and to monitor these individuals over a longer period of time.

ACKNOWLEDGEMENTS

We would like to thank Dr. Chris Jenkins from the Orianne Society for hosting this internship and for teaching and guiding us through this project. His knowledge and experience was invaluable, as was his assistance in scouting the sites ahead of time. We would also like to thank Dr. James Costa and Dr. Sarah Workman at the Highlands Biological Station for their continued support throughout this study.

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NATIVE BEES OF HIGHLANDS: DIVERSITY AND CONSERVATION

LEAH PURVIS

Abstract. Native bees are a keystone ecological group, supporting native plant populations and pollinating many crops that honey bees do not. Researchers have reached a consensus that native bees are declining at global, regional, and local scales. The continued loss of species could result in not only a loss of taxonomic diversity, but also cause negative impacts to native plant diversity as well as agricultural crop quality, yields, and diversity. This project investigates the diversity of native bees in Highlands, North Carolina, conservation strategies for native bees, and ways homeowners and small farms can provide foraging and nesting habitats to support local populations of native bees.

Key words: agriculture; bee nesting; conservation; diversity; habitat; Highlands, NC; land use; native bees; native plants; North Carolina; pollinator decline; pollination; wildflowers.

INTRODUCTION

In recent decades, increasing declines in honey bee populations have raised concerns about the future of insect pollination of major crops. While the main focus has been on learning more about the causes of honey bee declines, the plight of native bees (Hymenoptera: Apoidea) has not received as much attention. The importance of native bees as pollinators and as keystone species in ecosystems is also less understood, partly due to the lack of spatial and temporal data for the majority of the estimated 4,337 species of native bees in North America and Hawaii (Ollerton 2017, Kopec and Burd 2017). Based on the considerable research that has been conducted, it is apparent that native bees are critically important pollinators, and can be more efficient pollinators than European honey bees, which were introduced to U.S. in the 1600s (Brittain et al. 2013, Wilson and Messinger Carril 2016, Winfree 2010, Garibaldi et al. 2013). Native plant diversity and pollinator diversity are often strongly mutually dependent (Potts et al. 2010, Burkle et al. 2013, Ollerton 2017), and over half of native bee species appear to be in decline in diversity and abundance (Cameron et al. 2011, Burkle et al. 2013, Kopec and Burd 2017). Given these trends, it is necessary that our agricultural system and land use practices be transformed in order to protect and support native bee abundance and diversity.

The main causes of declines in native bee populations are habitat loss, habitat fragmentation and degradation, climate change, urbanization, pesticide use, competition with introduced species, and introduced pathogens (Hopwood 2008, Cameron et al. 2011, Burkle et al. 2013, Goulson et al. 2015, Kopec and Burd 2017). The most important of these are habitat loss and degradation, largely due to agricultural intensification and urbanization. These lead to the loss of foraging and nesting sites and exposure to insecticides (Potts et al. 2010, Rundlöf et al. 2015).

Native bees are best adapted to pollinate native plants, though some are generalists (forage on a wide variety of flower types and species) and can efficiently pollinate introduced species. Many wild bees pollinate plants that honey bees do not, including many fruit and vegetable crops (berries, melons, apples, tomatoes, etc.) that without native bees would decrease in both quality and yields (Garibaldi et al. 2013, Ollerton 2017). It is estimated that the fruit-pollination services of native bees in crop production amounts to $3 billion a year in the U.S. (Kopec and Burd 2017). Continued declines in native bee populations and the subsequent loss of their pollination services would undoubtedly have negative impacts on wild plant diversity, ecological function, and food security (Potts et al. 2010, Brittain et al. 2013, Ollerton 2017).

This research project focused on how homeowners, gardeners, and small-scale farmers can help provide habitat for native bees to support local populations. The Highlands Biological Station (Macon, Co., North Carolina) has a wide variety of native plants and habitats to host the wide diversity of bees in this region. My project specifically looks at the native bees in Highlands and how the biological station can further serve as a model and provide educational opportunities for creating bee-friendly habitat and nest sites, particularly for solitary tunnel-nesting native bees.

There are an estimated 500 species of native bees in North Carolina, the vast majority of which are solitary bees, which nest in aggregations but do not nest cooperatively in colonies or produce honey as honey bees and bumble bees do (Adamson n.d.). Around 30% of native bees in North America are tunnel-nesting bees, which in natural habitats nest in hollow stems and in holes bored by other insects (Mader et al. 2009). The native bee "hotels" (also referred to as nest boxes or houses) are built to provide nesting and overwintering sites for such bees in order to support local populations. Along with the wide diversity of bees comes great variety in nesting preferences and habits—many other bees dig tunnels underground or require particular nesting materials (e.g. certain soil types for making mud and particular leaves for provisioning nests). I will recommend some general tips for providing these bees with nesting resources as well (with details on nest box construction and maintenance in Appendix).

My project consists of three parts: First, a survey of the representative bees of Highlands, collected largely during the month of September 2017, with specimens pinned and identified to genus and species where feasible; Second, an educational pamphlet and field guide for Botanical Garden/Station visitors to learn about the importance of native bees, with hand-drawn illustrations of 10 common bees or bee groups in Highlands; Third, construction of "Native Bee Hotels," 7 nest houses of varying shapes and sizes to provide nesting habitats for a variety of tunnel-nesting bees.

In sum, the purpose of this project is threefold: I aim to contribute to the Highlands Biological Station's native bee conservation efforts by (1) providing a general idea of the common bee genera and species in Highlands, (2) designing interpretive materials to educate Station visitors about how to support native bees, and (3) constructing and installing nest boxes around the Station and thereby support local bee populations.

Specimens collected in this project will also contribute to the teaching and research insect collection at HBS. Finally, there is an experimental aspect to the nest box component of the project. With assistance, I built a variety of shapes, sizes, and types of nest boxes and nesting materials that over time can demonstrate what specific nesting habitat certain bees prefer. Use and observation of these bee hotels can show how to improve nest box design in the future to optimize use by native bees, rather than introduced wasps and bees, which may also make use of the nest boxes (see MacIvor and Packer 2015).

METHODS

Survey of Highlands' Bees

The first component of this project was conducting a survey of the bees at the Highlands Biological Station and nearby areas within 10 miles of Highlands. I sampled from September 8th to October 5th 2017; the majority of my specimens were collected in September. The locations that I sampled from were the Station campus and Botanical Garden, Satulah Mountain summit, Cliffside Lake Recreation Area, and Glenda Zahner's property (Table 1). I visited sites representing a variety of habitats including urban/roadside, bog edge, cove forest, oak-hickory forest, and open field.

I collected bees twice weekly between mid-morning and mid-late afternoon. To quantify my sampling effort, I estimate that I spent about 1-3 hours every day that I collected bees (approximately 10 days total), usually spending between 5 and 15 minutes at a flower patch site.

I used a sweep net, aspirator, and kill jars charged with ethyl acetate to collect bee specimens. On days when the weather was suitable for bee activity (sunny and warm) I visited flower patches and caught foraging bees that I believed I had not previously collected in order to avoid over-collecting common species. This sampling technique was by design not quantitative or systematic, as the intention was to obtain a representative sample of native bees of Highlands and not to quantify overall bee diversity or relative abundance.

Once collected, I froze the specimens for at least an hour, sometimes overnight, to ensure that the larger specimens were dead before pinning. I pinned the specimens using standard techniques (pinning block and no. 2 insect pins). I used several resources to identify the specimens to genus and, where feasible, to species. These included DiscoverLife.org's "IDnature guides," digital guides by Sam Droege, and print resources such as Theodore Mitchell's *Bees of the Eastern United States* (Mitchell 1960, 1962) and *The Bees in Your Backyard: A Guide to North America's Bees* (Wilson and Messinger Carril 2016). To identify specimens I used a microscope (Olympus VMT 1x, 4x VM Lab Stereo Zoom Microscope) and a video microscope system (Motic SMZ-168-BH 35° Trinocular Stereomicroscope fitted with Canon Vixia HF S30 1080p HD Camcorder).

Each specimen was labeled with locality, date collected, genus/species, and supplemental information where applicable (e.g., sex, host flower). The total collection of 46 pinned specimens was assembled in a dedicated drawer and deposited in the HBS insect collection.

Nest Houses

Seven native bee houses were built, one a larger size (20" x 24" x 12") and six smaller houses to be distributed around HBS campus and botanical garden. I used Hollow-Stem Joe-Pye-Weed (*Eutrochium fistulosum*), small dry logs, untreated 2" x 4, 4" x 4" and 4" x 6" cedar and hemlock lumber, and bamboo as nesting materials to fill the boxes. Holes of varying sizes and depths were drilled into the boards and small logs. The smaller tunnels were drilled less than ¼" in diameter and were 3"-5" deep, and the larger holes were greater than ¼" diameter (most were 5/16" in diameter) and 5"-6" deep. Bundles of bamboo and hollow stems were placed inside cylindrical containers (PVC pipes) or between the drilled wood blocks to fill the majority of the space within a nesting box. Open ends of Joe-Pye-Weed stems were sealed with caulk (with one end open and one closed: open end facing outward), and pieces of newspaper were rolled into straws and inserted into roughly 40% of the drilled holes of the logs and lumber blocks. All of the nest houses were painted with latex paint, in the colors yellow, dark lavender, dark green, light green, and pale yellow.

Educational Pamphlet and Field Guide

To complement the nest boxes, I created an educational pamphlet about the importance of native bees and creating habitats and nesting sites for them, along with a field guide featuring 10 common bees found in the Highlands area. I created the final products digitally using Microsoft Word and created the field guide from my own hand-drawn illustrations. The drawings were primarily done in watercolor and color pencil on mixed media paper, and were scanned and digitized at 600 dpi using a scanner (Brother DCP-7065DN multi-function copier). I also included photographs of flowers and bees that I took during the semester.

RESULTS

Survey of Highlands' Bees

Table 1 lists the sites I visited to collect bees. The majority of my specimens came from the HBS campus and Botanical Garden. Table 2 lists flower types and species on which bees were found foraging. I did not take notes of the flowers I collected bees from for every specimen, but from the data I did collect and general observations, some interesting trends can be noted. From Table 2, it is apparent that flowers in the Asteraceae family were utilized the most. This is most likely accounted for by the phenologies and availability of flowers. The majority of bee specimens for which I collected host flower information were found on Goldenrod (*Solidago* spp.), which was in bloom throughout the month of September, was very widespread, and often had a variety of bees foraging on them.

TABLE 1. Sites bees were collected from.

Cliffside Lake Recreation Area, Highlands, NC (trails and lake-side)
Flat Mountain Road (roadside vegetation of wildflowers along wet ditch at 4200 Flat Mountain Road)
Flower beds in front of Coker Hall and administration building, HBS
Glenda Zahner's property (open field of native vegetation, Bob Zahner Road; 35° 4'15"N, 83°12'37"W)
HBS Botanical Garden (Bog & Upper Lake Boardwalks)
HBS Cherokee garden and flower beds by New Duplex
HBS dam
Mary Enloe Memorial Garden, HBS Botanical Garden
Satulah Mountain summit (heath bald)

TABLE 2. Common host flowers on which bees were observed foraging.

Family	Genus	Species	Common name
Asteraceae	*Achillea*	*millefolium*	Common Yarrow
	Ageratina	*altissima*	Snakeroot
	Eupatorium	*perfoliatum*	Boneset
	Eutrochium	*fistulosum*	Hollow-Stem Joe-Pye-Weed
	Rudbeckia	*hirta*	Blackeyed Susan
	Rudbeckia	*laciniata*	Tall Coneflower
	Solidago	spp.	Goldenrod
	Symphyotrichum	*divaricatus*	White Wood Aster
	Symphyotrichum	*novae angliae*	New England Aster
	Symphyotrichum	spp.	(Other unidentified species)
	Verbesina	spp.	Wingstem
	Vernonia	*noveboracensis*	New York Ironweed
Balsaminaceae	*Impatiens*	*capensis*	Spotted Jewelweed
Campanulaceae	*Lobelia*	*siphilitica*	Great Blue Lobelia
Cucurbitaceae	*Curcurbita*	spp.	Squash
Hydrangeaceae	*Hydrangea*	*radiata*	Silverleaf Hydrangea

Collection Results

My pinned collection consists of 46 specimens of wild* bees, 22 of which I was able to identify to species; all are identified to genus. All the specimens were pinned and labeled with location information, taxonomic identification, date collected, and information about the specimen's sex, caste (for *Bombus impatiens*) and what flower it was collected from, where applicable. The entire collection is pictured in Figure 1.

*Because I didn't identify all to species, I cannot say all are native. Some *Megachile* Leafcutter bees (i.e. *M. rotundata*) in the region are introduced species.

TABLE 3. Summary of bees collected in Highlands, North Carolina, between September 8th and October 5th, 2017.

Family	Genus	Species	Common Name	# specimens
Apidae	*Xylocopa*	*virginica*	Eastern Carpenter Bee	1
	Bombus	*impatiens*	Common Eastern Bumble Bee	11
	Ceratina	*dupla*	Doubled Ceratina	1
	Ceratina	sp.	Small Carpenter Bee	1
Halictidae	*Augochlora*	*pura*	Pure Golden Green Sweat Bee	6
	Augochlorella	sp.	"Little Augochlora" Sweat Bee	3
	Lasioglossum	sp.	Sweat Bee	11
	Agapostemon	sp.	Metallic Green Bee	1
Colletidae	*Hylaeus*	sp.	Masked Bee	1
Megachilidae	*Megachile*	*texana*	Leafcutter Bee	2
	Megachile	sp.	Leafcutter Bee	3
	Coelioxys	*sayi*	Cuckoo Bee	1
Andrenidae	*Andrena*	sp.	Mining Bee	3

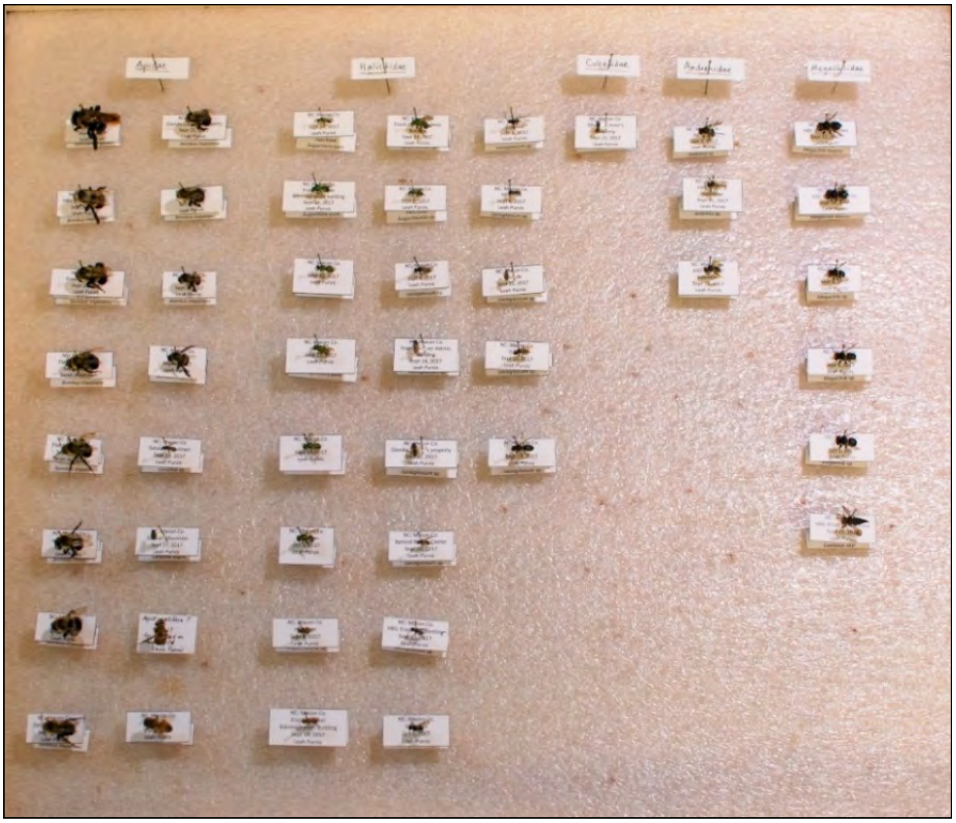
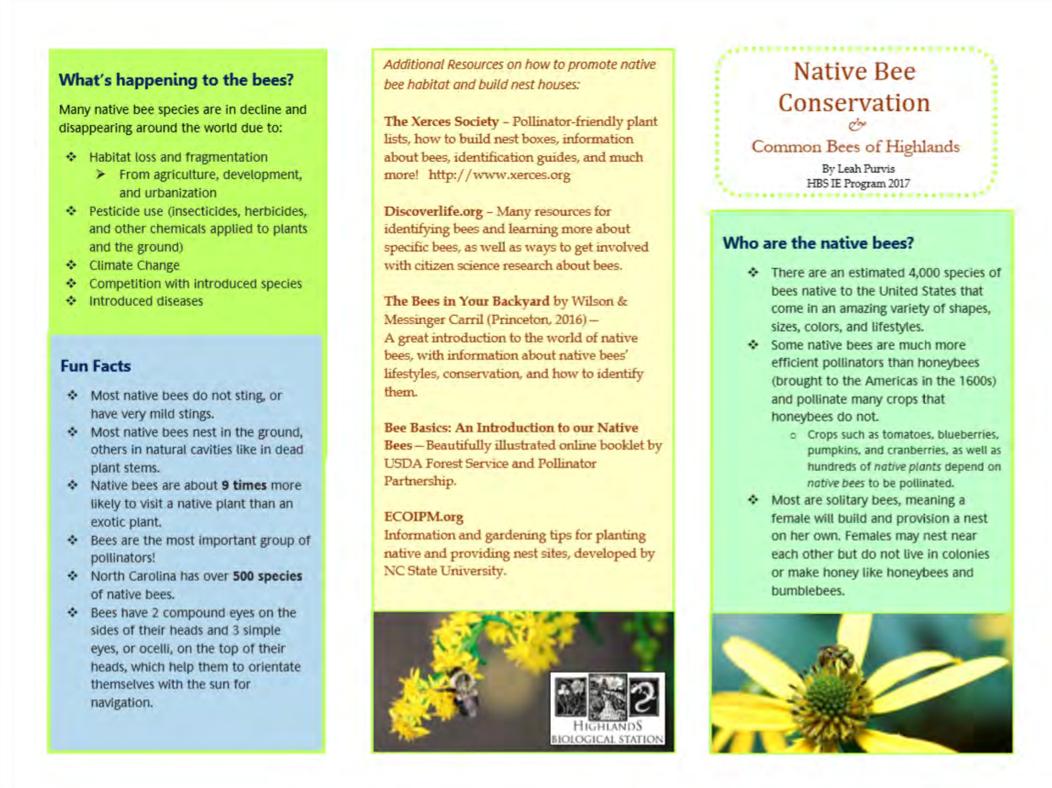


FIG. 1. Pinned specimen collection, organized in columns by family.

Pamphlet and field guide

The pamphlet and field guide (Fig. 2) will accompany the nest houses and will provide visitors with information about native bees, why they are in decline, and some simple recommendations for providing bee-friendly foraging and nesting habitat. The field guide includes 10 common bees found in Highlands, most of which are represented in the collection. Next to each scanned image of my illustrations is a brief description of each bee or bee group with defining characteristics or interesting natural history facts. Next to each drawing is an actual size bar, most of which were derived from approximate body length measurements of specimens in my collection.



What's happening to the bees?

Many native bee species are in decline and disappearing around the world due to:

- Habitat loss and fragmentation
  - From agriculture, development, and urbanization
- Pesticide use (insecticides, herbicides, and other chemicals applied to plants and the ground)
- Climate Change
- Competition with introduced species
- Introduced diseases

Fun Facts

- Most native bees do not sting, or have very mild stings.
- Most native bees nest in the ground, others in natural cavities like in dead plant stems.
- Native bees are about **9 times** more likely to visit a native plant than an exotic plant.
- Bees are the most important group of pollinators!
- North Carolina has over **500 species** of native bees.
- Bees have 2 compound eyes on the sides of their heads and 3 simple eyes, or ocelli, on the top of their heads, which help them to orientate themselves with the sun for navigation.

Additional Resources on how to promote native bee habitat and build nest houses:

The Xerces Society – Pollinator-friendly plant lists, how to build nest boxes, information about bees, identification guides, and much more! http://www.xerces.org

Discoverlife.org – Many resources for identifying bees and learning more about specific bees, as well as ways to get involved with citizen science research about bees.

The Bees in Your Backyard by Wilson & Messinger Carril (Princeton, 2016) – A great introduction to the world of native bees, with information about native bees' lifestyles, conservation, and how to identify them.

Bee Basics: An Introduction to our Native Bees – Beautifully illustrated online booklet by USDA Forest Service and Pollinator Partnership.

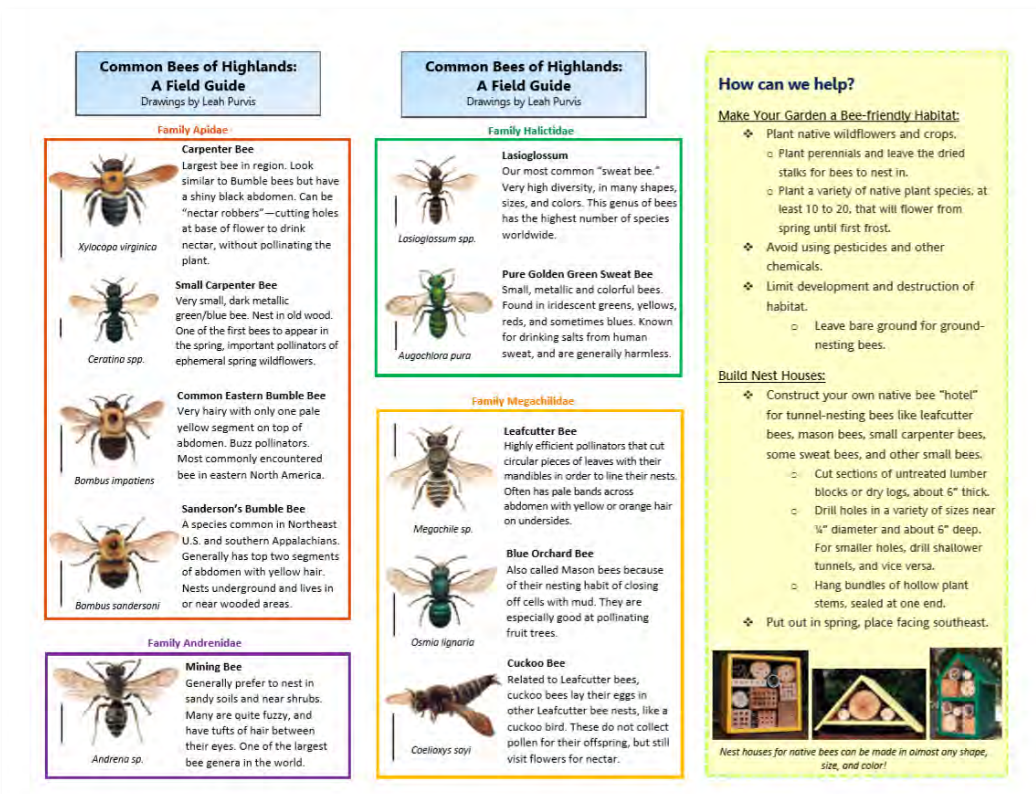
ECOIPM.org
Information and gardening tips for planting native and providing nest sites, developed by NC State University.

HIGHLANDS BIOLOGICAL STATION

Native Bee Conservation
&
Common Bees of Highlands
By Leah Purvis
HBS IE Program 2017

Who are the native bees?

- There are an estimated 4,000 species of bees native to the United States that come in an amazing variety of shapes, sizes, colors, and lifestyles.
- Some native bees are much more efficient pollinators than honeybees (brought to the Americas in the 1600s) and pollinate many crops that honeybees do not.
  - Crops such as tomatoes, blueberries, pumpkins, and cranberries, as well as hundreds of *native plants* depend on *native bees* to be pollinated.
- Most are solitary bees, meaning a female will build and provision a nest on her own. Females may nest near each other but do not live in colonies or make honey like honeybees and bumblebees.



Common Bees of Highlands: A Field Guide
Drawings by Leah Purvis

Family Apidae

Carpenter Bee (*Xylocopa virginica*)
Largest bee in region. Look similar to Bumble bees but have a shiny black abdomen. Can be "nectar robbers"—cutting holes at base of flower to drink nectar, without pollinating the plant.

Small Carpenter Bee (*Ceratina spp.*)
Very small, dark metallic green/blue bee. Nest in old wood. One of the first bees to appear in the spring, important pollinators of ephemeral spring wildflowers.

Common Eastern Bumble Bee (*Bombus impatiens*)
Very hairy with only one pale yellow segment on top of abdomen. Buzz pollinators. Most commonly encountered bee in eastern North America.

Sanderson's Bumble Bee (*Bombus sandersoni*)
A species common in Northeast U.S. and southern Appalachians. Generally has top two segments of abdomen with yellow hair. Nests underground and lives in or near wooded areas.

Family Andrenidae

Mining Bee (*Andrena sp.*)
Generally prefer to nest in sandy soils and near shrubs. Many are quite fuzzy, and have tufts of hair between their eyes. One of the largest bee genera in the world.

Common Bees of Highlands: A Field Guide
Drawings by Leah Purvis

Family Halictidae

Lasioglossum (*Lasioglossum spp.*)
Our most common "sweat bee." Very high diversity, in many shapes, sizes, and colors. This genus of bees has the highest number of species worldwide.

Pure Golden Green Sweat Bee (*Augochlora pura*)
Small, metallic and colorful bees. Found in iridescent greens, yellows, reds, and sometimes blues. Known for drinking salts from human sweat, and are generally harmless.

Family Megachilidae

Leafcutter Bee (*Megachile sp.*)
Highly efficient pollinators that cut circular pieces of leaves with their mandibles in order to line their nests. Often has pale bands across abdomen with yellow or orange hair on undersides.

Blue Orchard Bee (*Osmia lignaria*)
Also called Mason bees because of their nesting habit of closing off cells with mud. They are especially good at pollinating fruit trees.

Cuckoo Bee (*Coelioxys sayi*)
Related to Leafcutter bees, cuckoo bees lay their eggs in other Leafcutter bee nests, like a cuckoo bird. These do not collect pollen for their offspring, but still visit flowers for nectar.

How can we help?

Make Your Garden a Bee-friendly Habitat:

- Plant native wildflowers and crops.
  - Plant perennials and leave the dried stalks for bees to nest in.
  - Plant a variety of native plant species, at least 10 to 20, that will flower from spring until first frost.
- Avoid using pesticides and other chemicals.
- Limit development and destruction of habitat.
  - Leave bare ground for ground-nesting bees.

Build Nest Houses:

- Construct your own native bee "hotel" for tunnel-nesting bees like leafcutter bees, mason bees, small carpenter bees, some sweat bees, and other small bees.
  - Cut sections of untreated lumber blocks or dry logs, about 6" thick.
  - Drill holes in a variety of sizes near ¼" diameter and about 6" deep. For smaller holes, drill shallower tunnels, and vice versa.
  - Hang bundles of hollow plant stems, sealed at one end.
- Put out in spring, place facing southeast.

Nest houses for native bees can be made in almost any shape, size, and color!

FIG. 2. "Native Bee Conservation & Common Bees of Highlands" informational trifold pamphlet and field guide, with information about native bee natural history, causes of decline, and ways to provide foraging and nesting habitat for native bees.

Nest House Construction

With assistance from HBS Facilities Manager Mike McMahan, I built 7 nest houses of varying shapes, sizes, and colors. They were filled with nesting materials including stems of Hollow-Stem Joe-Pye-Weed (*Eutrochium fistulosum*), collected from the HBS grounds, and introduced bamboo (*Phyllostachys* sp.*)*, drilled wood blocks (cedar and hemlock), drilled logs (maple, fire cherry, white oak), and PVC pipes containing bundles of the hollow stems. Figure 3 (at bottom center) shows a nest house designed to open to permit viewing of nest tunnels in cross-section through a Plexiglass panel. The nest boxes will be installed at various locations on the HBS campus, including at the new pollinator garden when it is completed.

FIG. 3. Completed native bee nest houses, filled with drilled lumber blocks and logs, Hollow-Stem Joe-Pye-Weed, cinder blocks, and bamboo. Note: these images do not indicate relative sizes.

Discussion

Highlands' Native Bees

The specimens I collected represent the common groups of bees that were foraging in late summer and early fall (early September-early October), and include specimens from all of the families that would be expected for this region (Table 3). The most common specimens were the apid *Bombus impatiens* and the halictid *Lasioglossum (Dialictus)* spp. and species of the Augochlorini tribe: *Augochlora* and *Augochlorella.* Bees in the family Halictidae have the greatest representation in the collection, with 21 specimens total, and 14 in the family Apidae. The number of specimens of each family is not important, as I was only sampling for representative species, but it does give some indication of the relative abundances of each family, and corresponds to what I observed in the field as I collected. Halictids are very widespread and abundant, as are *Bombus* spp., though I am surprised by the lack of diversity of bumble bees I collected.

According to *Bumble Bees of the Eastern United States* (Colla et al. 2011), there are likely more than 12 species of bumble bees in the southern Appalachians based on historical range maps. I hoped to find Sanderson's Bumble Bee (*B. sandersoni*) (a northern species common in the southern Appalachians) at the summit of Satulah Mountain, but was unsuccessful. This is likely accounted for by their phenology, since their foraging period ends by the close of August. It is unclear why I did not collect any other bumble bee species, such as the widespread *B. bimaculatus B. griseocollis, B. perplexus, B. vagans, B. pensylvanicus,* and others. One possibility is that managed *B. impatiens* populations have caused declines in other bumble bee species from introduced pathogens from Europe (Cameron et al. 2011). This is only speculative, however; there are a number of other factors that could have affected my observation and collection of other bumble bee species, including sampling methods and sites visited. The time of the year likely had a significant effect; if I had sampled during the height of summer I would have likely gotten a much broader assemblage of species.

In retrospect I realize I could have collected more specimens, but I chose bees to collect based on whether I thought I had collected that species before in order to reduce redundancy in my collection and to avoid unnecessary loss of life. Nonetheless, I feel this collection is a good representation of the most common bees that were foraging at the time.

Brief natural histories of the 5 common bee families of Highlands

Leafcutter bees (genus *Megachile*) and Mason bees (genus *Osmia*) are two of the most frequent users of "native bee hotels," and both are common in the region. These bees belong to the family Megachilidae, a diverse group of solitary cavity-nesting bees. Many megachilids have special adaptations in the hairs on the face and abdomen for carrying large amounts of pollen, and thus are known as very efficient pollinators. In fact, many species are managed by farmers for major crops, like alfalfa (pollinated by the introduced *Megachile rotundata*) and fruit crops by the Orchard Mason Bee (*Osmia lignaria*) (Bambara 2002, Beaudette 2013). Bees in the genus *Megachile* are known as Leafcutter bees for their habit of cutting circular pieces of leaves with their mandibles and using them to line their nest cells. The family Megachilidae also includes cleptoparasitic bees like *Coelioxys*, which I collected, and are quite different from the aforementioned bees. *Coelioxys* is a cuckoo bee, which lays its eggs in other megachilid bees' nests, and takes advantage of the hosts' pollen and nectar loaf intended for the host offspring. They have an extremely pointed abdomen which they use to break into the sealed nest cells of other megachilid bees, particularly Leafcutter bees (Wilson and Messinger Carril 2016).

Halictidae are another diverse family, and are globally very widespread and often form dense populations. Most are quite small, metallic, and very colorful. This group of bees is unique in that not all are solitary; some species are primitively eusocial, others communal or semisocial. The degree of sociality varies among groups. Most halictidae are ground-nesting, though some species prefer to nest in rotting wood. Ground-nesting halictids often dig branched tunnels; their burrows can be recognized by the small mounds of dirt around the entrances. Like other ground-nesting solitary bees, halictids nest in areas of bare earth or sand and often in sunny locations, though the soil type and preference for shade cover vary by species. I collected many halictids, primarily of two kinds: species in the tribe Augochlorini, and unidentified *Lasioglossum* species. I observed that both were very abundant as I was collecting, and were often foraging on Asteraceae. *Lasioglossum* is the largest bee genus in the world, with about 1,800 species. Halictidae are known as extremely effective pollinators on account of their large population sizes and because most are generalists (Wilson and Messinger Carril 2016).

Andrenidae are also ground-nesting bees, sometimes called "Mining bees." This is the largest of the six bee families (the sixth, Melittidae, do not occur in the southern Appalachians). Andrenids line their nest tunnels with a waterproof substance made by the female bees that protects the developing young from pathogens in the soil. I found three andrenid bees, all in the genus *Andrena* (which includes about 1400 species—one of the largest bee genera globally). This genus is very widespread and abundant, and are often characterized by their especially hairy bodies, with patches of hair between the eyes (called facial fovea). These bees are important pollinators of commercial crops like blueberries, apples, and cranberries (Wilson and Messinger Carril 2016).

Colletidae, the fourth family common in the southeastern United States, only has 5 genera north of Mexico. Colletidae show a wide range of nesting habits; many are ground-nesters while others nest in twigs and hollow stems. Like Andrenids, they secrete a waterproof substance on the walls of nest tunnels to protect offspring from soil fungi and other pathogens as well as excess moisture. I found only one specimen in this family, of the genus *Hylaeus*, a widespread and common genus in North America that often nests in hollow stems of perennial bushes and other woody plants. They only nest in preexisting holes, and so could be benefitted by artificial bee nest houses (with a variety of hole diameter sizes and depths to accommodate the very small H*ylaeus*). Interestingly, they are often found in forests, where bees are not commonly found. They are also commonly found in suburban/urban areas likely due to their variability of nesting preferences. *Hylaeus* bees are very small and wasp-like, with very few hairs and a jet black body often with yellow and white markings on the face and legs. Since they don't have pollen-collecting hairs on their bodies, these bees instead collect pollen for their offspring by eating it and regurgitating later to provision their nests (Wilson and Messinger Carril 2016).

The final family with representation in my collection is Apidae, which includes *Bombus*, *Xylocopa*, *Ceratina*, and *Apis* spp. (bumble bees, carpenter bees, and honey bees), which are the genera I collected. The Apidae are the most commonly seen and well-known bees, though this family includes many more species (5,700 species total), with great variation in sizes, colors, and body shapes. Many bees in this family are specialists, and thus are important in pollinating specialty crops and maintaining native plant populations and diversity. Most Apidae nest in the ground, as many bumble bees do in abandoned rodent holes, but others, like many carpenter bees

(*Xylocopa* and *Ceratina*) nest in plant materials (stems and solid wood). A distinguishing character of this family is that it includes the only bees that are truly social (with a queen and worker castes)—honey bees and bumble bees. Native bumble bees are very important for agricultural crop production; bumble bees are buzz pollinators, giving them the ability to more effectively pollinate plants that honey bees cannot, like tomatoes, blueberries, peppers, and potatoes. (Buzz pollination involves a bumble bee intensely vibrating a flower's anthers to forcibly release pollen that otherwise would not be accessible.) In addition, bumble bees have the ability to regulate their body temperature by vibrating their flight muscles, which allows them to forage in temperatures most bees cannot, making them the bees with the longest foraging seasons—from early spring to late fall, dawn to dusk (Wilson and Messinger Carril 2016).

Threats and Conservation

Native bees worldwide face many threats. These include habitat loss and fragmentation, pesticide use, climate change, competition with introduced species, parasites and diseases, and loss of plant diversity (Winfree 2010, Cameron et al. 2011, Ollerton 2017, Goulson et al. 2015). Research by Burkle et al. (2013) found that "interaction network structure and function" has become degraded due to the loss of bee species, mismatches in phenologies between bees and their host plants due to increasing temperatures, and land use change as forests and prairies have been converted to agricultural landscapes (among other factors). It stands to reason that as bee species become locally extinct and global diversity decreases, the pollination services of bees will also decline, negatively affecting native plant populations and diversity (particularly those that rely on oligolectic (specialist) bee species), and lowering the quality and yields of agricultural crops (Potts et al. 2010).

With regard to pesticide use, though the exact effects of the many thousands of toxic chemicals in use are unknown, recent studies have found that neonicotinoids, one of the most commonly used groups of pesticides worldwide, is affecting both domesticated honey bees and native bees (Burkle et al. 2013, Rundlöf et al. 2015, Bush 2016). Neonicotinoids, and other pesticides, do not have to be directly lethal to negatively affect bee populations, though there is evidence that they directly reduce nesting populations, slow colony growth of bumble bees, and reduce the density of native bee populations (Rundlöf et al. 2015). The toxic compounds accumulate in a bee's body over time and can cause neurological damage, impairing the bee's ability to navigate and forage, and impair immune responses, making them more susceptible to disease and parasites (USGS 2017, Goulson et al. 2015, Rundlöf et al. 2015, Bush 2016). In addition, pesticides can also affect bees by reducing the diversity and availability of flowering plant species (Potts et al. 2010, Goulson et al. 2015). Recent studies also indicate that bees found outside of agricultural areas show less exposure to pesticides, and that untreated plant cover in surrounding edges of agricultural fields could be very important for reducing the impacts of agricultural pesticide use on bees (USGS 2017). Organic farming methods could help mitigate negative trends, such as reducing pesticide use and tillage, rotating crops, planting a diversity of native flowering cover crops, planting smaller fields, and planting native plants along field and forest edges (Hopwood 2008, Winfree 2010).

Connectivity between foraging and nesting habitats is an important factor in native bee population stability, along with landscape heterogeneity or patchiness. The restoration of agricultural land and roadsides can have significant positive impacts on bee diversity and abundance. Roadside and field edge restoration gives a number of benefits, including providing connectivity between habitat fragments, increasing potential nesting sites and foraging resources, and supporting other wildlife (Hopwood 2008). A 2007 study on how land use affects crop visitation of native bees found that increasing connectivity between small farms (small patches of natural habitat) had a greater effect on the number of bees visiting crops than land use type (Winfree et al. 2008). The study emphasizes the importance of maintaining a heterogeneous landscape with a variety of nesting substrates and native plant diversity. Notably, this study also found that wild bees visited more plants than honey bees, and were "responsible for 62% of the flower visits over the entire study system" (Winfree et al. 2008).

Providing Foraging and Nesting Habitats

Along with increasing connectivity and heterogeneity of landscapes, increasing the diversity of nesting substrate types and flowering plants in agricultural and non-agricultural contexts is essential for supporting a wide diversity of native bees. Floral and nesting requirements and preferences among bees is widely variable, and bees that specialize in utilizing particular nesting requirements and flower types will likely decline without them (Hopwood 2008). Therefore it is very important that a wide variety of native flower resources are available from spring to early autumn, in rural and suburban contexts. Agricultural and suburban landscapes can be very suitable habitats for native bees, so long as they are not too intensively managed (Winfree 2010). It is becoming increasingly important that the conservation efforts for native bees take advantage of their ability to thrive in human-modified landscapes, as there is much we can do to support them.

In addition to planting a diversity of native flowering plants that have overlapping blooming periods—providing floral resources throughout the foraging season—it is also important to provide nesting habitat. This includes leaving bare soil for ground-nesting bees, leaving patches of woody debris for cavity-nesting bees, providing mud sources for bees that use mud to partition their cells, and providing artificial nesting sites for tunnel-nesting bees (Wilson and Messinger Carril 2016). Studies have found that providing nesting sites does grow cavity-nesting bee populations (Winfree 2010). Artificial bee nesting houses were first developed in the 1960s by alfalfa farmers to manage alfalfa leafcutter bees (*Megachile rotundata*), and more recently for managing Orchard Mason Bees (*Osmia lignaria)* (Xerces Society n.d.). Nest houses can be constructed to attract other species of native bees like *Ceratina* spp., some Halictidae, *Hylaeus* spp., and other species of leafcutter and mason bees (Wilson and Messinger Carril 2016). Drilled blocks, bundles of hollow stems, adobe blocks, and wooden shelters filled with nesting material can all attract native bees. Nest houses can be constructed to be as simple or decorative as the owner likes, so long as certain requirements are met (see Appendix A).

Effective conservation strategies for native bees may require changes in agricultural landscapes to provide greater diversity of flower resources and nesting sites for native bees. Roughly a third of the world's land is used for agriculture, and because bees provide pollination services and managed honey bees are in decline, native bees may become more relied upon for pollination. Therefore there are numerous incentives for farmers to support native bee populations (Winfree 2010).

Long-term data on native bee population trends is scarce, and there is great need for consistent, standardized monitoring protocol that citizens and farmers can contribute to. The Xerces Society for Invertebrate Conservation has developed a streamlined monitoring procedure "to allow agency staff, land managers, farmers, and others to evaluate the performance of individual pollinator habitat plantings" (2014). This kind of monitoring protocol could be very useful in providing information about the land use practices and conservation tools that are most effective for augmenting native bee populations, and for engaging more people in native bee conservation efforts.

As more research is conducted, scientists are finding that native bees play vital and indispensable roles in ecosystems and agricultural systems, and that relying on managed honey bees for crop pollination and disregarding native bee conservation comes with great risk. Without native bees, it is very likely that the efficiency of crop pollination will decrease, as shown in a study by Garibaldi et al. (2013), who found that for 41 crop systems worldwide, wild bees played a larger role in the pollination of those crops than honey bees, were more efficient, and increased fruit set two times more than honey bees did. Other studies have similarly found that native bees contribute greatly to crop pollination and can be as effective as honey bees (Vaughan and Black 2007, Winfree et al. 2008). In addition, native bee diversity and native plant diversity are strongly codependent, with changes in either group causing a cascade of decreasing ecosystem diversity and function. Taken in context with the global patterns of dramatic declines in diversity of all insects (along with many other taxa), it is imperative that natural habitat be restored and protected, that more research and monitoring be done for native bees, and that land management practices change to increase heterogeneity in the landscape (Garibaldi et al. 2013, Caspar et al. 2017, Goulson et al. 2015, Winfree et al. 2008).

CONCLUSION

Maintaining native bee diversity and populations is crucial for native plant populations and overall ecosystem function, and will become more important for agricultural crop production as well if honey bee populations continue to decline. As agricultural demand grows with human population growth, more care will need to be taken to promote a landscape that supports a diversity of native pollinators. Decline in native bees will have consequences not only for cultivated crops, but also for the wild plants that native bees help to fertilize and the wildlife that depends on plant diversity. Considering the combination of present-day threats (habitat loss, pesticide use, introduced species and pathogens, decreasing plant diversity, etc.), along with the looming threat of climate change, it is clear that action needs to be taken now to support native bee populations. This includes federal programs as well as conservation initiatives by private and non-profit organizations, and actions taken by cities, small farms, homeowners, community gardens, and schools to create as much bee-friendly habitat as possible.

ACKNOWLEDGEMENTS

I would like to thank the following people for their contributions to this project: Dr. Costa–for assigning me this project and being my mentor, for introducing me to the marvelous world of bees, and for his guidance and assistance throughout the semester on the project (helping identify specimens, editing this paper, pointing me to helpful resources, and inspiring me to delve deeper into the study of insects). Many thanks to Mike McMahan for his tremendous assistance with building the nest houses, and his support, enthusiasm, and conversation. I also want to thank Leslie (Chan) Chandler—for his constant support and eagerness to help, Dr. Workman—for taking me places to look for bees, and for her assistance with various parts of the project, and Dr. James Pitts of Utah State University—for helping with the identifications of several challenging bee specimens.

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APPENDIX.

The following are general specifications for building nest houses, compiled from a variety of sources listed below.*

<u>For drilled wooden blocks:</u>

- Only use untreated lumber. 4" x 4" and 4" x 6" work well.
- Tunnels should be approximately ¼" diameter, though the hole diameter can vary to attract a variety of bee species. In general, tunnels less than ¼" diameter should be 3" to 5" deep, and those larger than ¼" should be 5" to 6" deep. (The diameter and depth of the tunnels affect the proportion of males to females in a population; larger and deeper tunnels produce more female bees.)
- Holes of 5/16" diameter and 6" deep are preferred by mason bees.
- It is recommended to drill holes of the same size in each block.
- Holes should be drilled ¾" apart from one another.
- Do not drill all the way through block, or bees will not use it. (Nor will they use open-ended hollow plant stems; caulk can be used to seal open ends.)
- The smoother the inside of the tunnel, the better; drill perpendicular to wood grain and use a sharp drill bit.
- For easier cleaning, use paper straws to line the tunnels (parchment paper and newspaper work).
- The exterior can be stained or painted, but do not use wood preservatives.
- Attach a downward slanting roof to divert water from the nest holes.
- Place nesting block on a sturdy post or on a wall, at least 3 feet above ground.
    - Facing southeast, so bees are warmed by morning sunlight.
- Place at a location within 300 feet of flowers (the foraging range of mason bees), and a mud source within about 50 feet of the nest block, as mason bees partition their cells with mud.

For hollow stem bundles:

- Joe-Pye Weed, reed, teasel, cup plant, and bamboo are suitable nesting materials.
- For stems with natural nodes, it is easiest to cut below the nodes to create straws with one open end. For stems like Joe-Pye weed, caulk can be used to close off one end.
- Bundles of 10 to 20 stems can be held together with string, wire, or tape, and should be either placed in a sheltered location (barn, shed, etc.) or should be stuffed inside a cylindrical container (like a short section of PVC pipe, a water bottle, plastic buckets, etc.).
- Like drilled blocks or larger bee houses, bundles should be placed facing south/southeast so that morning sunlight warms the nest
- Bundles of plant stems can also be placed between drilled blocks and logs to create a variety of nesting materials in a nest box. (Fig. 4)

FIG. 4. Examples of ways to incorporate different nesting materials into native bee nest house.

- Maintenance and sanitation:
    - To avoid buildup of disease and parasites, nesting materials should be replaced after two years of use.
    - Chalkbrood and pollen mites are the most common threats.
    - Once bee activity has stopped around first frost, move bee house into an unheated building. (Cool temperatures are essential for the larvae and overwintering adults to develop and hibernate properly.)
    - If paper straws were used, remove them from blocks and store them in a protected container. The wooden blocks should be submerged in a bleach solution for a few minutes to kill any possible pathogens.
    - In the spring, once temperatures consistently reach 55 degrees Fahrenheit, return the nest house to its outside location.
    - Building several, smaller nest houses or bundles and distributing them farther away from each other may better mimic natural nesting conditions, prevent build-up of disease and parasites, and are easier to replace and maintain.

(Note: Bee nest houses, paper tubes, stackable wood nesting trays, bundles of reeds, and other nesting materials are also commercially available and can be purchased online.)

*For additional information about native bees, building nest houses, and planting natives:

https://www.ces.ncsu.edu/depts/ent/notes/Other/note109/note109.html
(“How to raise and manage orchard mason bees for the home garden” –by Stephen Bambara, NCSU)

http://www.ecolandscaping.org/03/beneficials/attract-mason-bees-no-protective-gear-needed/
(“Attract Mason Bees – No Protective Gear Needed” – by Judy Beaudette, Ecological Landscape Alliance)

For information about all nest house types and details on how to maintain and clean bee houses:
http://www.xerces.org/wp-content/uploads/2009/11/tunnel-nest-management-xerces-society.pdf
(“Tunnel Nests for Native Bees: Nest Construction and Management—by Xerces Society)

For lists of native flowers by region by Xerces Society:
https://xerces.org/pollinator-conservation/plant-lists/

Top 25 Native Pollinator Plants for North Carolina:
https://growingsmallfarms.ces.ncsu.edu/wp-content/uploads/2012/08/Top-25-Plants-and-Suppliers-2.pdf?fwd=no
(by Debbie Roos, NC Cooperative Extension—Chatham County Center)

Clay Bolt's website on North American bees:
http://beautifulbees.org/

“Enhancing Nest Sites for Native Bee Crop Pollinators” by USDA National Agroforestry Center
https://www.plants.usda.gov/pollinators/Enhancing_Nest_Sites_For_Native_Bee_Crop_Pollinators.pdf

OLD GROWTH ACROSS THE CHATTOOGA RIVER WATERSHED: THE 1995 CARLSON MAPS, AND WHAT THEY MEAN

ERIC SCHWARTZ

Abstract. A history of industrial logging in the southern Appalachians has rendered old growth forest- tree stands of late-stage development relative to forest type and ecosystem structure- ostensibly scarce. In 1995, the U.S. Forest Service funded a study of old growth in the Chattooga River watershed. The resulting maps of old growth highlighted vast discrepancies between the Forest Service data and the realized landscape; an important issue, as the former is a basis for forest management decisions, including timber cuts. By digitizing the 1995 maps, this study aims to heighten awareness of documented old growth stands, thus allowing them to become players in forest management decisions. Further, a comparison of the 1995 field notes with Ellicott Rock Wilderness area Forest Service data highlights persisting misconceptions of forest type and structure in the landscape.

Key words: Carlson; Chattooga River watershed; Ellicott Rock Wilderness; forest ecology logging; old growth; timber; U.S. Forest Service.

"Every forest has a right to its ghost."
-*El Bosque Animado,* 1994

INTRODUCTION

Crossing the north branch of Little River, GA, and moving up a sloping bank, William Bartram finds himself among Black Oaks; the very scale of these trees, great trunks of 30-foot circumference, leave him in disbelief: he writes in *Travels,* "To keep within the bounds of truth and reality, in describing the magnitude and grandeur of these trees, would, I fear, fail of credibility…" (1998: 19). More than two and half centuries after Bartram set out from Philadelphia to explore the Southeast in 1761, an encounter with such a stand of forest might invoke even greater awe, as a mere one half of one percent of known primordial forest still stands in the Southeastern US (Sierra Club 2005). Indeed, of the estimated 24281138.5 hectares (60,000,000) acres of primary pine forests that once stretched over the region during the period of initial European settlement, it is believed a mere 4856.3 hectares (12,000 acres) still stand (Lockette et al 2004).

Engagement with the land precedes European agency. Native Americans are believed to have cleared parts of the original forest at least 12,000 years ago. They focused their settlements on the most easily accessible and fertile tracts of land in the region, the floodplains (Dickens 1976). Living in the valleys, and establishing 64,373,760 square meters (40,000 square miles) of settlement there- the Native Americans were nevertheless known by mountain peaks: the first white settlers called the Blue Ridge mountains "Cherokee mountains," according to Davis in *Where There Are Mountains* (2000: 61). The still pervasive forests of the region provided the Cherokee with the flora and fauna they subsisted on, including blackberry, strawberry, hickory nuts, chestnuts, herbs, black bear, and white-tailed deer (Davis 2000). According to Cherokee myth, "no hunter who has regard for his health ever fails to ask pardon of the deer for killing it" (2000: 61); a pathos that well represents the reciprocity of the Native American's relationship to the landscape. Scars of greater permanence and extent did not begin to appear across the southern Appalachian landscape until European expansion, and ultimately, the industrial logging epoch of the 1880's-1920's. It was in the late 19th century that an encroachment of railways into the southern Appalachians facilitated unprecedented clear-cuts (Davis 2000). These cuts catalyzed an environmental cascade, where unnaturally barren mountainsides channeled extreme levels of run-off- and erosion- into the headwaters, causing mass flood events (2000: 165). In an attempt to control the flooding, in addition to bringing rural electrification, large-scale dam construction also began in the 20th century, creating lakes where there were once valleys, and ultimately inundating “more than a million acres of mountain lands, [and submerging] over a thousand miles of natural flowing streams and rivers…” (2000: 191). Davis notes, “During the four-hundred-year history of land use that [*Where there are Mountains*] chronicles, the single greatest human activity to affect environmental and cultural change in the southern Appalachians is industrial logging” (2000: 166).

In response to the massive disappearance of American forests, The Forest Reserve Act of 1891 was passed, granting the president of the United States power to set aside forests as public domain. In 1905, the US Forest Service (USFS) became the controlling body of these public forests. Under the Forest Service’s protection– or as described by Leverett (1996), their “benign neglect”- secondary forests began to grow. Gifford Pinchot, the original chief of the Forest Service, considered forest conservation, "the art of producing from the forest whatever it can yield for the service of man." Thus, the Forest Service operates both as a means of forest preservation, but fundamentally, as an agency of resource extraction.

The 1990’s saw a burgeoning shift in management ethos, propagated by public support of forest preservation (Leverett 1996), away from traditional clear-cutting practices and towards “ecosystem management,” synonymous with more selective cuts (Wallace and Cortner 1996). During this heightened moment of ecosystem awareness, old growth trees became a more mainstream point of study. These trees- relics of the original landscape, often predating the American experiment itself- were described by the Forest Service as meeting the following parameters:

1. Large trees for the species and site.
2. Wide variation in tree sizes and spacing.
3. Accumulations of large-sized dead standing and fallen trees that are high relative to earlier stages.
4. Decadence in the form of broken or deformed tops or boles and root decay.
5. Multiple canopy layers.
6. Canopy gaps and understory patchiness.

In 1995 the U.S. Forest Service tapped forest ecologist Paul Carlson to study the remaining old growth in the Chattooga River Watershed. Between October of 1994 and March of 1995, Carlson led a survey across 4,9371 hectares (122,000 acres) of National Forest land in the Chattooga River watershed, looking for stands of old growth forest. According to the Forest Service (1997), “Old-growth communities are rare or largely absent in the southeastern forests of the United States” (p. 1). Indeed, Carlson (1995) used 150 years of age as the criterion for “old” in this study because, “this corresponds to a period (around 1844) when logging was limited near to early settlement sites” (Carlson 1995: 5). The study predicted old growth stands to exist in relatively inaccessible locations where logging was historically difficult or impossible due to steepness of terrain or other landscape factors. The Forest Service acknowledges a diversity of old growth value, including “biological diversity, wildlife habitat, recreation, esthetics, soil productivity, water quality, aquatic habitat, cultural values, and high-value timber products" (USFS 1997), although the last of these attributes seems incompatible with other ecosystem benefits promised by old growth. Given the malleable definitions concerning old growth, and the ambiguity of the concept, it is necessary from a conservation management perspective to identify stands based on the Forest Service's identification. This is part of why Carlson's work remains important 22 years later; as a Forest Service study, it provides a semblance of framework for old growth identification in the Chattooga River watershed.

The Carlson study resulted in eight maps delineating 156 old-growth (and potential old-growth restoration) stands, commanding 2304.7 hectares (5,695 acres) over three states- Georgia, South Carolina, North Carolina- and three National Forests: Chattahoochee, Sumter, and Nantahala. Yet, study is only a start on old growth identification in the Chattooga River watershed, as noted by Carlson (1995, p. 6): "[this study] does not represent all of the old growth forest present in the watershed, but rather a subsample of the total old growth resource."

As an agency mandated to cultivate the forests systems they manage, the Forest Service regularly engages in timber sales. The "Southside Project" is a currently proposed timber sale in the Chattooga River watershed, which potentially encroaches on stands of old growth identified by Carlson (USFS 2017), and serves as a case in point. Balancing the system-level interests of the landscape with the Forest Service's tendency to treat old growth as "high-value timber products" frames my study: A digitization of Carlson's maps, allowing known old growth stands to better inform forest management decisions; a comparison of Carlson's field notes with Forest Service data in areas of old growth; and modeling elevation, slope and aspect of old growth areas, to more accurately describe and predict the extent and environmental qualities where ancient forest can be found the Chattooga River watershed.

METHODS

Carlson's maps had previously existed exclusively as hand-annotated paper copies, with highlights and boundaries penned across eight hardcopy maps. Their lack of availability in electronic form limited their potential for use in management decisions. To make this valuable data more readily available, I digitized the Carlson maps and integrated them with Forest Service data. An additional goal was to provide an exacting representation of discrepancy between known old growth stands and Forest Service data of know old growth stand areas. (table 1). The impetus here is twofold: The Forest Service data currently lists forest-type, without mentioning old growth qualities at all, and digitizing these maps increases the likelihood of their recognition. Further, incongruities between Forest Service labels and true forest structure holds serious implications for management, i.e. if a mixed oak-dominated stand is labeled by the Forest Service as a white pine dominated stand, this gives them the basis to manage it as such: when they cut the forest, they will leave white pine trees standing. This way, the stand will be repopulated as white pine dominant. Because all trees are not created economically equal- white pine makes disproportionally lucrative timber (Williams personal communication. Interview. 2017.) - native forest composition and diversity must be recognized, and correctly catalogued.

Finally, I improve upon original elevation estimates through models with updated data (table 2), provide information on these stands' slope (table 3), and- based off of these known parameters, provide a predictive model of likely old growth area in the Chattooga River watershed (fig x). The minutia of forest labels- and the access to geospatial information on old growth stand- ultimately guide management action- action like the Southside Project.

Carlson denoted stands of old growth trees across maps of eight landscapes, separated as USGS topographic quadrats: Satulah, Tamassee, Rabun Bald, Rainy Mountain, Scaly Mountain, Whetstone, Cashiers, and Highlands. The old growth stands fell into three classifications:

> Class A: Old growth stands with no significant signs of human disturbance to the forest canopy or understory; a canopy dominated by trees of at least 150 years in age;
>
> Class B: Stands dominated by trees of at least 150 years in age, accompanied by signs of anthropogenic disturbance of at least 50 years of age- or stands of younger trees without sign of anthropogenic disturbance;
>
> Class C: Old growth stands with anthropogenic disturbance of any date, with mature trees in the canopy (potential old growth restoration sites).

Using ArcGis (Environmental Systems Research Institute (ESRI). 2017. ArcGIS Desktop: (10.5) Redlands, California, USA.) I digitized each of these quadrangles and their respective stands. The stands are colored in relation to class; Class A old growth stands are magenta, Class B stands are orange and Class C are Tarheel blue. Exemplars of each are provided in Figure 1.

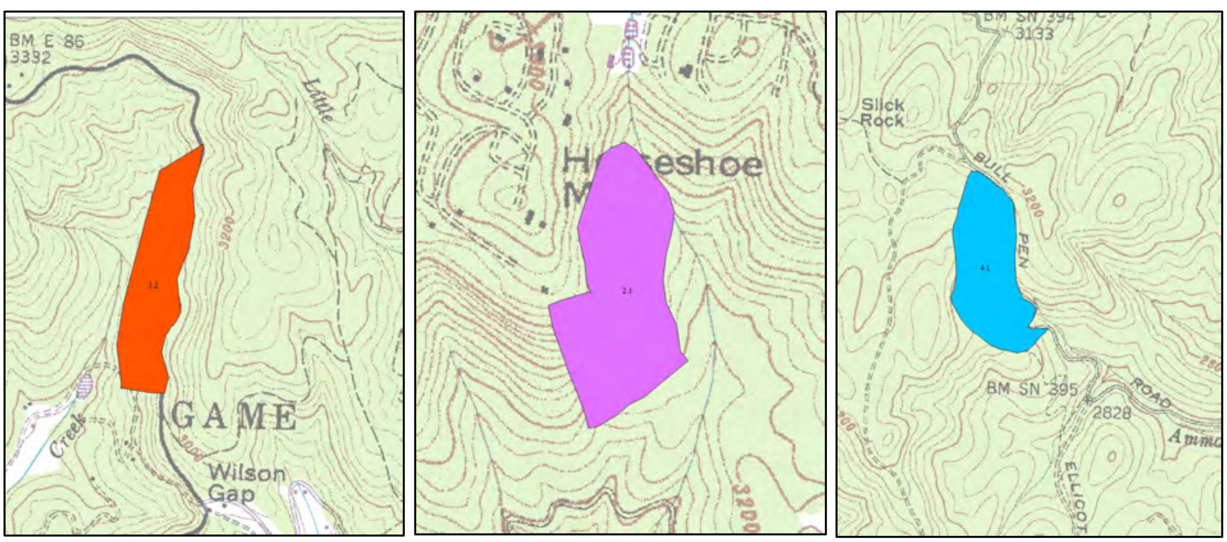

FIG. 1 Class B old growth stand north of Wilson Gap (left); class A old growth (middle) along Horseshoe Mountain's gradient; and class C old growth (right) hugging Bull Pen Road, between Ellicott Rock and Slick Rock.

To digitize the Carlson maps, I first scanned each hard copy map and downloaded the image onto my computer. Then I geo-referenced each image, a method of lining up an uncoordinated image with a coordinated layer. I used a USGS topographic map as my reference layer. Assigning the scanned Carlson maps coordinates allowed me to trace polygons over each stand of old growth. Finally, I populated the polygons with class information and ran elevation, aspect, and slope models for each stand class using a topographic layer from Landscape Fire and

Resource Management Planning Tools (LANDFIRE). Using data ranges from slope, elevation, soil information, input from Feature Analysis- an ArcGis tool that discerns spatial patterns- I created a model for likely old growth habitat.

RESULTS

Figure 2 outlines the 2304.7 hectares (5,965 acres) of old growth forest stands stretching across Macon and Jackson, Rabun and Oconee Counties, as identified by Carlson (1995). Figure 3 orients the landscape by the eight quadrats Carlson delineated in his study (figures 4-11). The elevation and slope (tables 2 and 3), in addition to soil information and spatial pattern were used as parameters in a predictive model of likely old growth across the Chattooga River watershed.

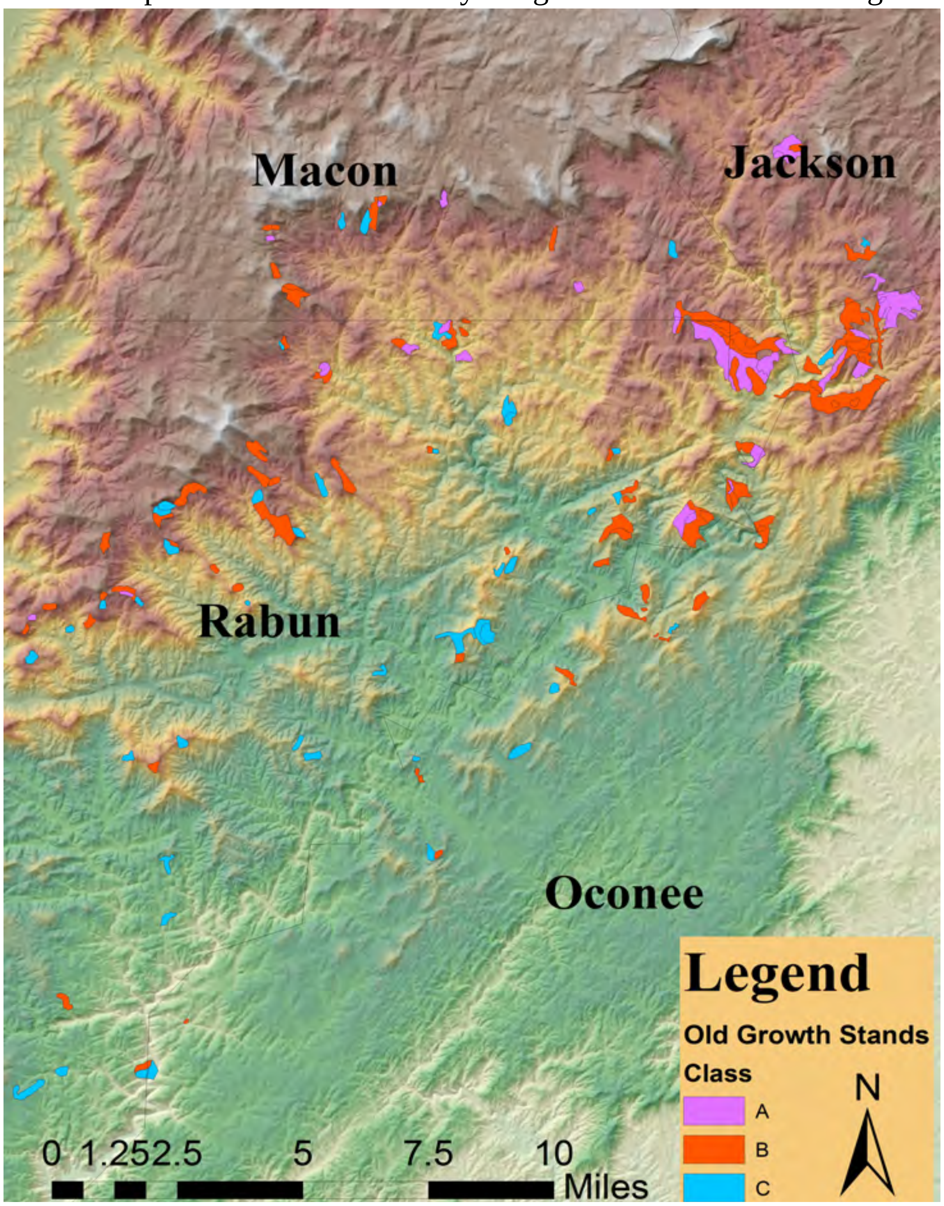

FIG 2. A mosaic of old growth as identified by Carlson, stretching across Macon, Jackson, Rabun and Oconee counties.

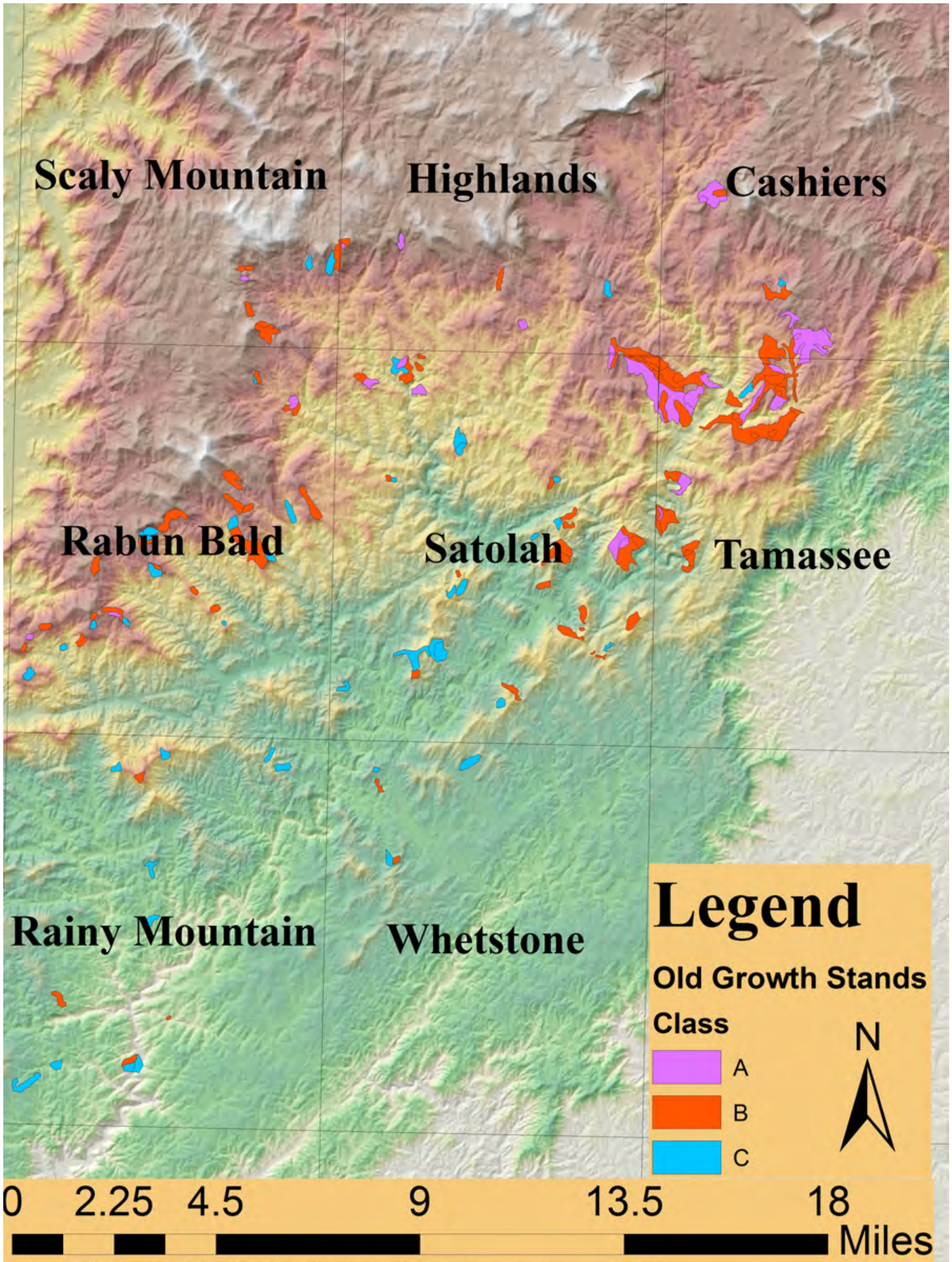

FIG 3. A mosaic of old growth as identified by Carlson, as delineated by quadrat.

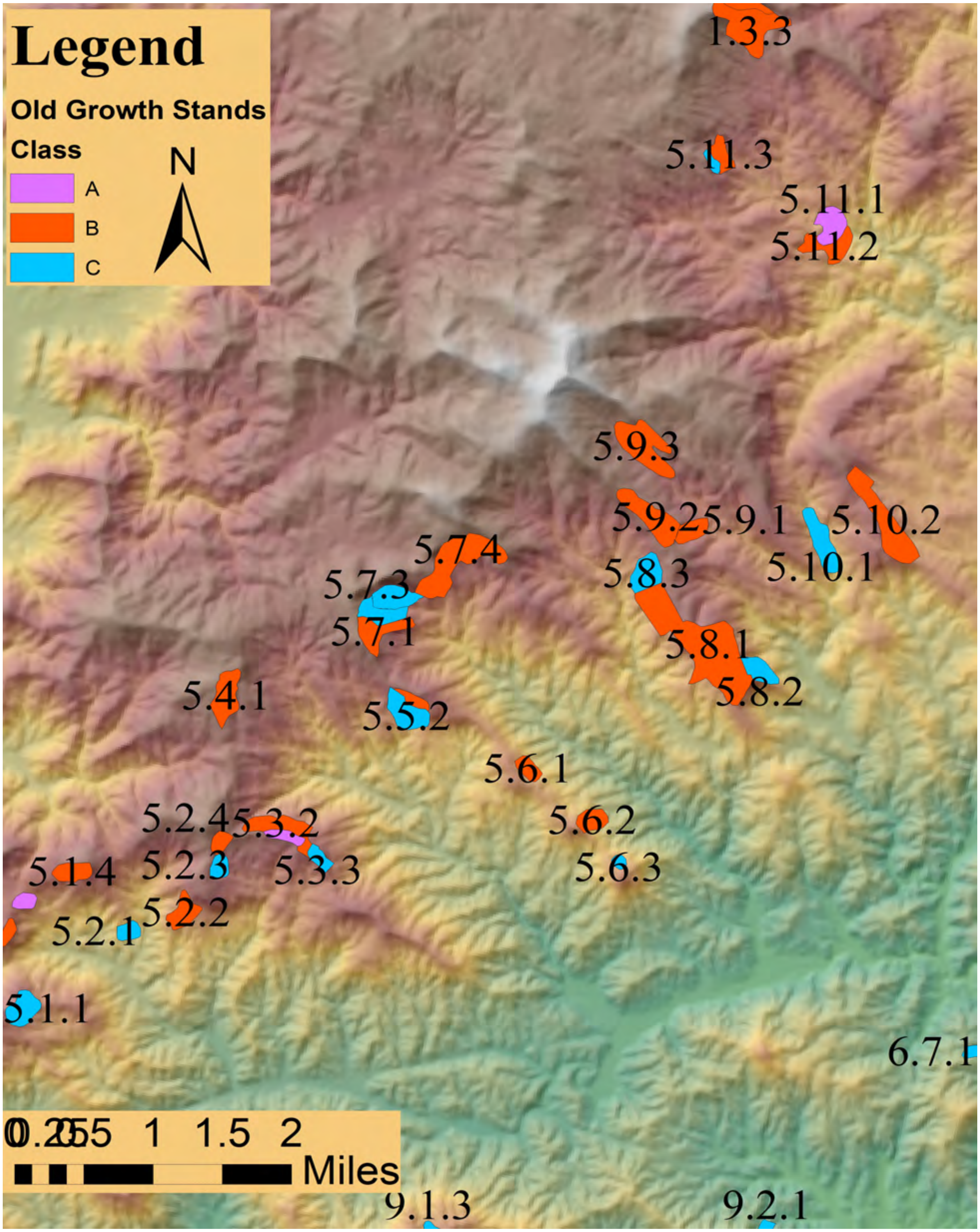

FIG 4. Old growth across the Rabun Bald quadrat.

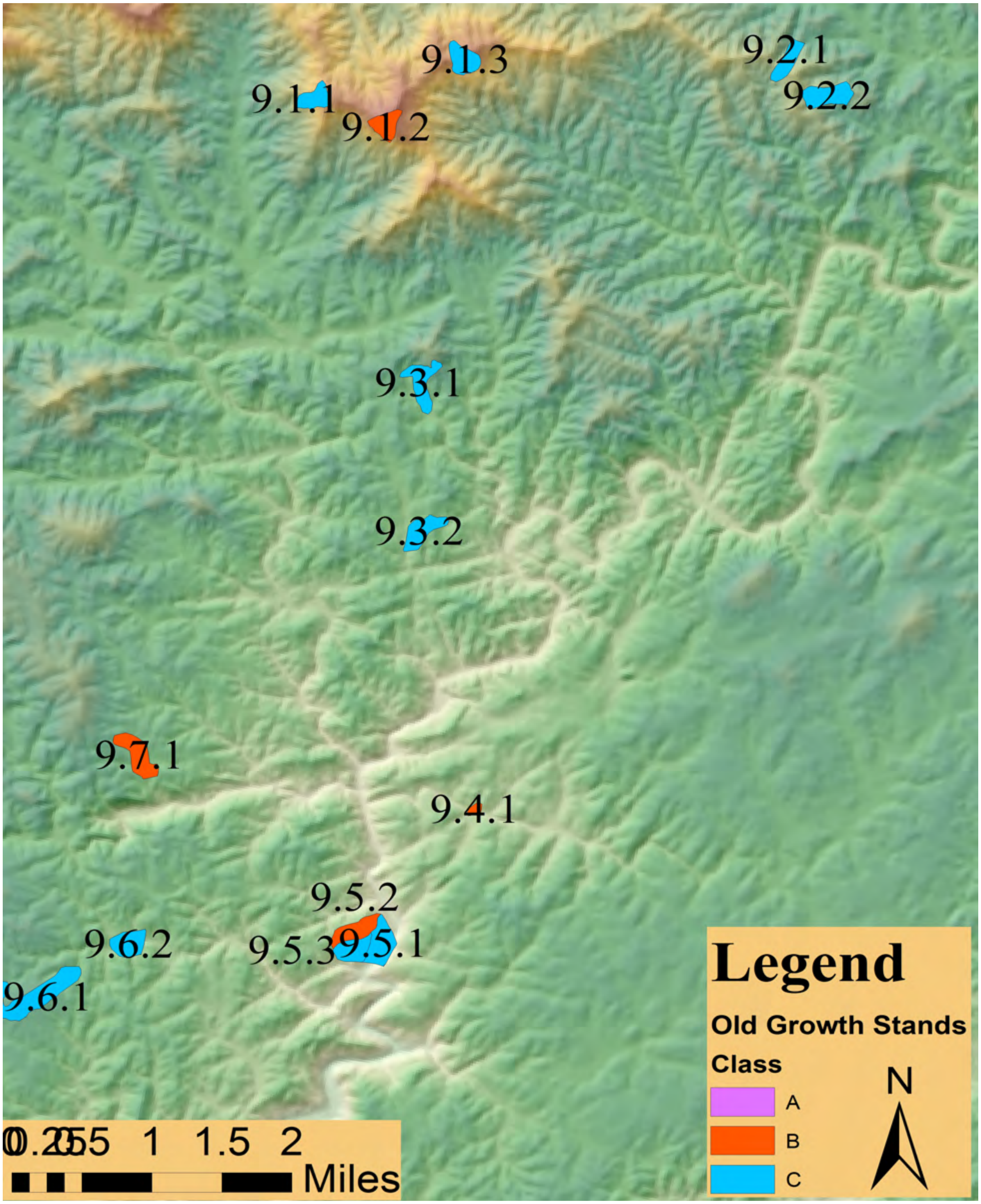

FIG 5. Old growth across the Rainy Mountain quadrat.

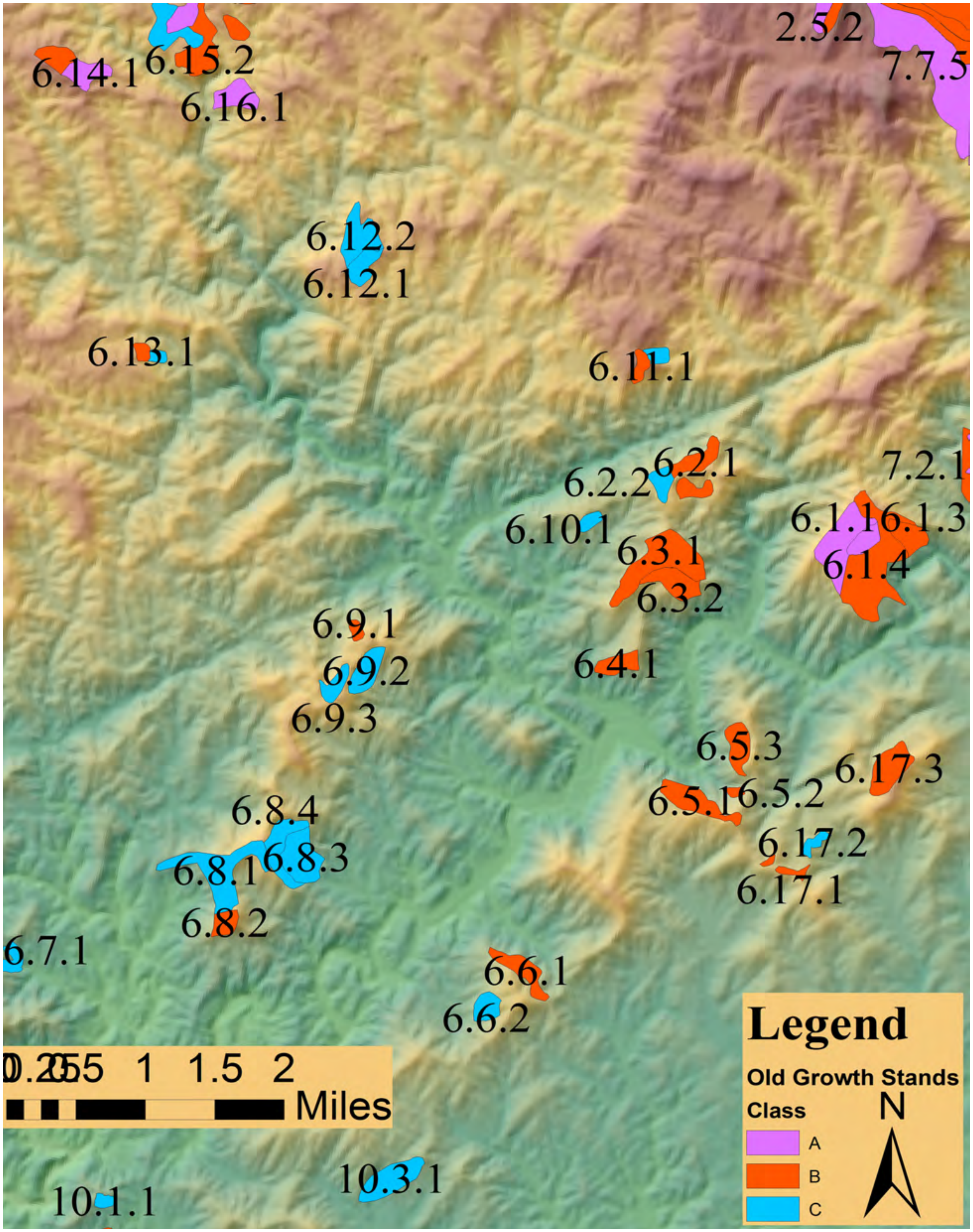

FIG 6. Old growth across the Satulah quadrat.

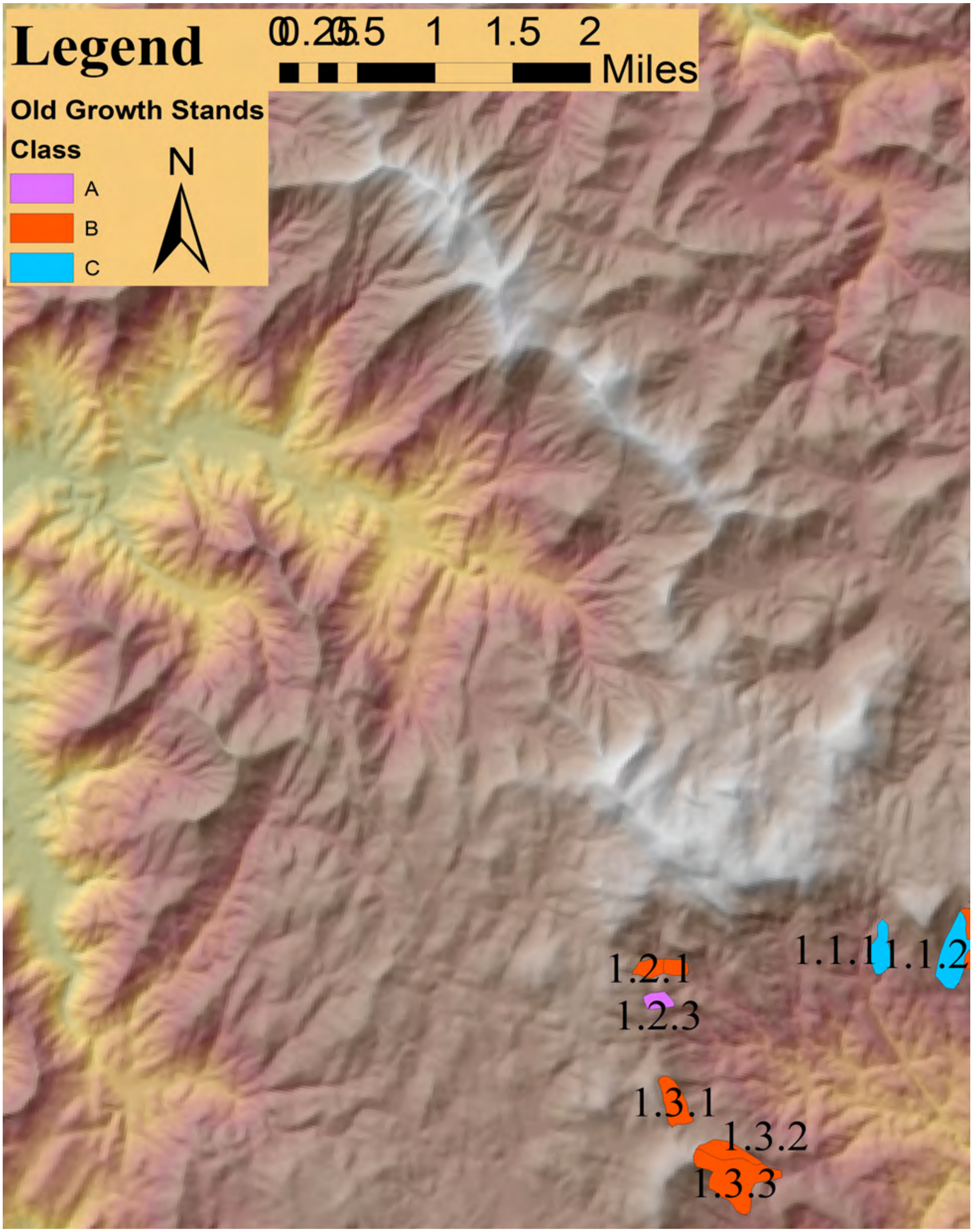

FIG 7. Old growth across the Scaly Mountain quadrat.

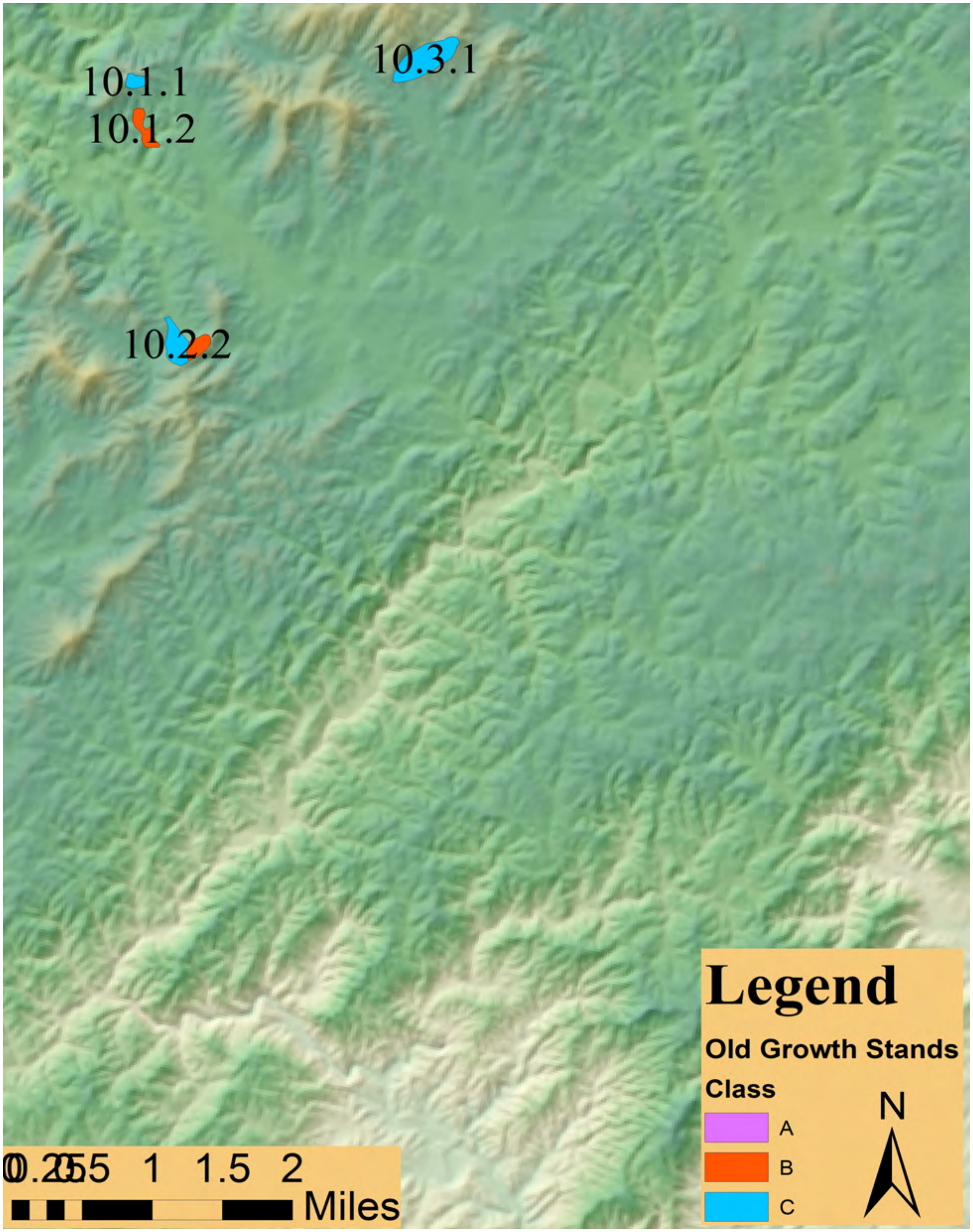

FIG 8. Old growth across the Whetstone quadrat.

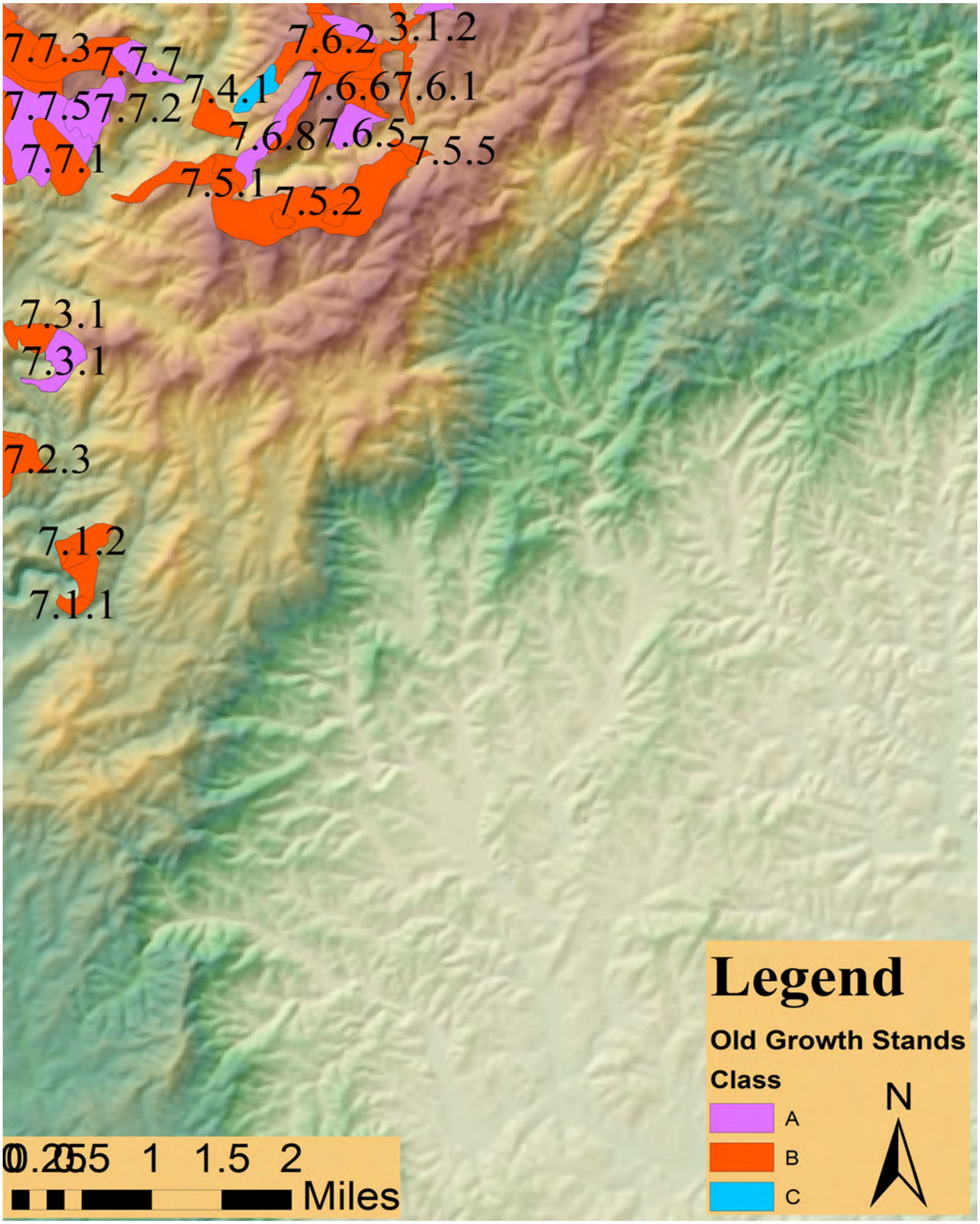

FIG 9. Old growth across the Tamassee quadrat.

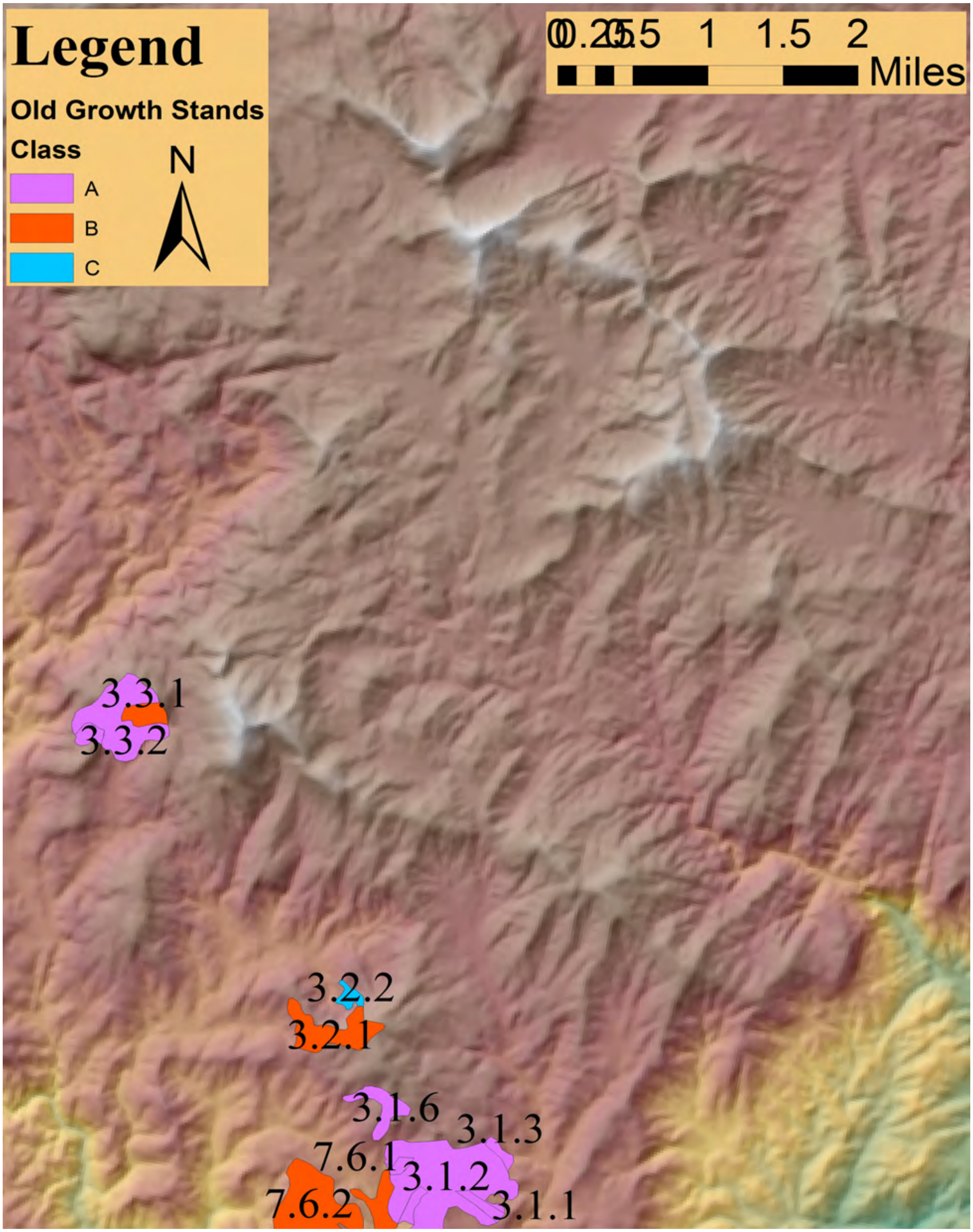

FIG 10. Old growth across the Scaly Mountain quadrat.

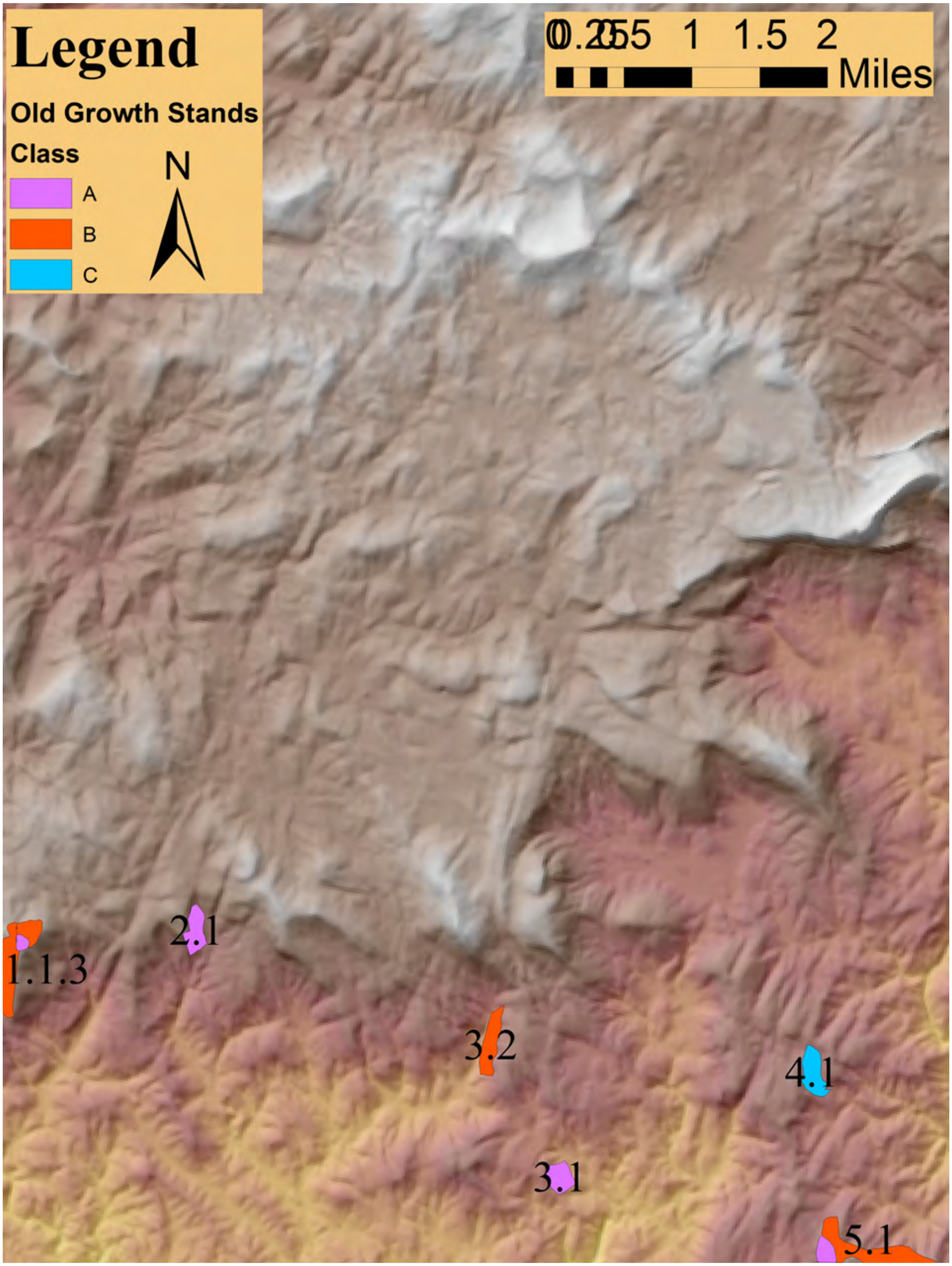

FIG 11. Old growth across the Highlands quadrat.

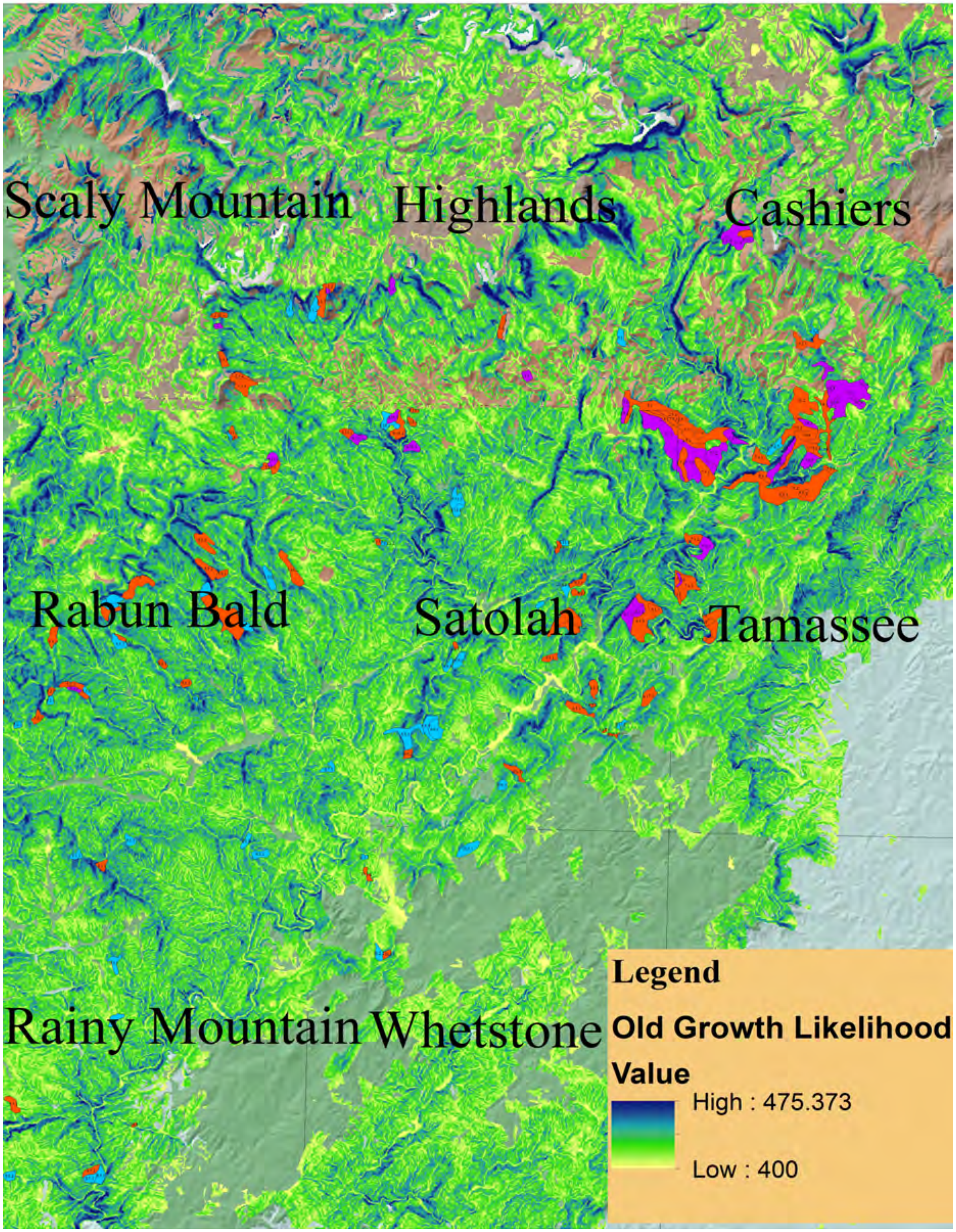

FIG 12. The yellow to blue range represents hospitable old growth terrain, based on the 1995 Carlson stand attributes. Blue areas are the most hospitable. The blue streaks run along higher elevations.

TABLE 1. Forest service data of forest typology in old growth areas in the Ellicott wilderness area, compared with Carlson's groundtruthing. Includes notes of interest from Carlson narrative and the old growth classification of each stand.

Stand (as correlated in the Carlson narrative)	Class	Carlson Forest Composition	USFS Forest Composition	Notes from Carlson narrative
1.3.3, 1.3.2 (B)	B	Red-white-chestnut oak	Red-white-chestnut-oak and white pine	Likely further old growth on the slope of Osage Mountain
1.3.1	B	White Oak-northern red oak	White Oak-northern red oak-hickory and white pine upland hardwood	
1.2.3	A	White oak-northern red oak-hickory	White oak-northern red oak-hickory	Chestnut blight disturbance nearby
2.1	A	Oak, hickory, maple	Chestnut-Oak, Brush species	
1.1, 1.2	A, B	Chestnut oak	Chestnut oak	1.2 is disturbed by chestnut blight
2.1	A	Chestnut-white-black oak, Yellow poplar, hemlock	Chestnut ok, Brush species	Evidence of logging around boundaries of stand. Evidence of past fire events.
3.2	B	Chestnut-white-red oak, black gum, Pitch and White pine, Hickory	White-red oak, Hickory	Disturbed by nearby road construction (Highway 28)
3.1	A	Chestnut-black-scarlet oak, Yellow poplar, sourwood,	White-red oak	"[the natural decline of the oldest trees] is contributing lots of standing dead and down wood… This is allowing for the development of a younger mixed oak and maple age class as well as the release of abundant white pine regeneration.
4.1	C	Cucumber magnolia, yellow poplar, elder hardwood	White-red oak, White pine	Most trees aged ranged from 157-162 years old; the 'C' designation comes from the disturbance presence of oak-chestnut removal. Surrounding areas were logged with abandon.
5.2, 5.1	A, B	White-black oak	Chestnut oak	Logging outside of the stands; chestnut blight prevalent in 5.1
3.3.1	B	Red oak, diverse shrub layer (mountain holly, striped maple), chestnut sprouts black birch, red maple, Carolina silverbell, yellow poplar	Chestnut oak	Highly afflicted by chestnut blight
3.3.2, 3.3.3	A	Chestnut-black oak dominant, with blackgum, pitch pine scarlet oak shrubby *Robinia* and *Tobinia viscosa*	Chestnut oak	Jack's Knob and its rocky gradient could have deterred logging in the area
3.2.2	C	Chestnut-white oak	White-Red oak	Surrounding areas host bountiful white pine. Selective logging and chestnut blight impacted the stand. The remaining trees, however, are old.

3.2.1	B	Chestnut-scarlet oak, white pine	White-red oak	Heavy cuts outside of the stand.
3.1.6	A	Chestnut-white oak, white pine	Chestnut-white-red oak, Hickory	White pine were younger than oak; outside of the stand ran logging roads, and forests disturbed by cuts.
7.6.2	B	Red-chestnut oak	White pine-Cove hardwood, Hemlock, White-red oak, Hickory	Two apparent age classes: I) 190-210 years old II) 125 years old. Evidence of logging surrounds the stand. Evidence of chestnut blight
7.6.3	A	White-black-Scarlett oak, red maple, white pine	Hemlock	Old trees are common yet relatively small (Old pitch to 205 years old, 25" dbh)

Stands described in table 1 were found across a range of elevation. Class A old growth was found from 600 to 1279 meters; class B from 354 to 1279; and class C from 295 to 1122 meters, elevation (table 2).

TABLE 2. Elevation range per stand class (meters)

Class	Minimum	Maximum
A	600	1279
B	354	1279
C	295	1122

TABLE 3. Slope range per stand class

Class	Minimum	Maximum
A	.337	47.26
B	.178	49.7
C	.328	39.7

DISCUSSION

The discrepancy between data used by the Forest Service to inform their management decisions and the observed stand assessment, based upon groundtruthing, is apparent in Table 1. It appears that the US Forest Service data do not match the old growth stands as identified in the 1995 field study. Thus, diversity is inherently overlooked, propagating a monolithic treatment of species. Diversity and nuance are lost in overly coarse-grained characterizations of forest stand. In the present analysis, it appears that the generalized categorization used by USFS land managers results in under-estimation of old growth and species richness. This in turn would have the effect of undermining management decisions that require accurate forest delineations.

Not all trees are equally profitable timber; the Forest Service tends to over-label stands as holding more lucrative species than actually exist, perhaps so it can claim an ecological basis for managing it as such, clearing all but a few mother trees to replenish the population. In this way, a species cleansing occurs, where native forest integrity is eschewed for long-term composition change; a diverse forest of oak-maple-pine turns into a white pine farm. In my analysis of Forest Service data on the Nantahala National Forest, I found that while 3,221 stands were evaluated as a type of white pine forest (white pine hardwood, white pine cove, white pine dominant), 4,450 stands were listed under white pine management. Thus, 1,229 non-white pine dominant stands will likely be cleared of their native composition.

Table 2 illustrates the idea of geography as an old growth insulator. The most pristine and least perturbed sites, class A, are found at the highest range (gradient) of elevation in the study. Often, the difference between class A and class B delineation is minor human disturbance. That class B can be found at 354 meters of elevation, whereas the lowest class A begins at 600 meters, also alludes this point of accessibility. Class C, meanwhile, had the lowest elevaional gradient range, as was suspected. Slope, the steepness of a gradient, is an additional way of gauging accessibility. From this perspective, class C appears across the landscape at less extreme slope gradients than classes A or B.

Fig. 12 visualizes the range of parameters for old growth areas based on the 1995 stands. Old growth forest stands appear at once sporadically- following arbitrarily lines of human interest, as in private and public land divides- and according to geospatial patterns (Stahle 1996). Ancient stands often ride escarpments and plateaus and hide in bayous (Stahle 1996). These environments all pose challenges to machinery and timber harvest incursions, making them great havens for old growth. That the least perturbed old growth in Carlson's sample also had the highest elevation range is consistent with this pattern. So is the representation in fig. 12, where the most likely old growth habitat runs along the higher elevation of greater slope.

The Ecologic Imperative

The southern Appalachians are among the most biodiverse regions in the world. And old growth stands, vestiges of primordial forest structure, not only harbor great species richness, but also are agents of inter-species preservation; a comparison of vernal herbaceous species richness between primary and secondary stands showed that older primary stands had higher species richness that secondary growth stands (Duffy and Meier 1992). Further, the hesitancy of vernal herbaceous plants to return to anthropogenically-disturbed stands- even ninety years later- suggests that species require natural gap-phase structure of an unperturbed forest. Gap-phase structure is established by the natural death of old trees, which is dualistically important for the ecosystem: when they fall, their decomposition sends resources back into the forest system. And in the canopy gaps they leave behind above them, secondary forest finds its opportunity, capitalizing on access to sunlight to grow (Meier et al. 1992). The temporal scale for this process for some tree species, when exposed to ample sunlight from the start, grow quickly and robustly. Yet, when trees raised in a natural gap-phase cycle, with shade in their youth, followed by eventual exposure to sunlight, they ultimately grow taller and hardier (Wohlleben 2016). Forest ecology plays out over a greater temporal scale than the logging industry finds efficient. Indeed, present logging management cycles of 40-150 years promises a continual degradation of biodiversity in the southern Appalachians, as the forest stands are deprived of the necessary time for maturation, cannot benefit from the decomposition of mature fallen trees, and do not develop stabilizing canopy-gap phasing structure. (Meier et al. 1992).

Herein lies an opportunity: to establish important connectivity between stands of old growth, to give the perturbed forest the necessary time to re-establish a natural structure, and to propagate human interaction and connection with nature. These goals could be simultaneously accomplished by the creation of a wilderness area. Designating land as a wilderness area protects it from resource extraction and development, while still allowing for camping, hunting, fishing and horseback riding, among other human activities. According to the Wilderness act of 1964, land must meet the following qualifications for designation: [that the land] (1) generally appears to have been affected primarily by the forces of nature, with the imprint of man's work substantially unnoticeable; (2) has outstanding opportunities for solitude or a primitive and unconfined type of recreation; (3) has at least five thousand acres of land or is of sufficient size as to make practicable its preservation and use in an unimpaired condition; and (4) may also contain ecological, geological, or other features of scientific, educational, scenic, or historical value. The main objection to these parameters may be to the first requirement, given the history of human disturbance. Yet, these acts of large-scale disturbance are largely only seen in structural vestiges, as single-age canopy covers and historically disproportionate numbers of white pine. The identified stands of old growth dot the landscape as an archipelago, yet with the benefit of connective wilderness area, they can become a peninsula.

The present study, by providing open access maps of known old growth stands in the Chattooga River watershed, becomes a vital tool for ecologic forest management- illuminating not only the known, existing stands, but also tracts of younger forest that act both as connective corridors between older forest, and as old potential old growth restoration sites themselves.

Given that the species richness per square meter is unaffected by the overall size of a given stand, every piece of old growth forest holds immense ecological value as localized diversity oases (Meier and Bratton 1992). However, due to the interconnected nature of ecosystem function, it remains crucial that species are not sequestered only into fragmented outposts of suitably primordial conditions. Such fragmentation of habitat propagates the likely reduction of biodiversity over larger spatial scales (Meier et al. 1992). Primary forest specialists, including southern Appalachia's renowned diversity of salamanders and vernal herbaceous plants, depend on unperturbed forest structures, with their natural cycles of canopy gap-phases producing uneven-aged canopy layers (Meier et al. 1992). The present study, by providing open access maps of known old growth stands in the Chattooga River watershed, becomes a vital tool for ecologic forest management- illuminating not only the known, existing stands, but also tracts of younger forest that act both as connective corridors between older forest, and as old potential old growth restoration sites themselves.

Avian communities also benefit from an old growth matrix: come winter, old growth makes exceptionally hospitable shelter and reliable feeding habitat for birds; snags, most prevalent in mature, uneven-aged forests, make important microhabitats for cavity nesting birds, including chickadees, nuthatches, woodpeckers, barred owls, chimney sifts and the common merganser; and ultimately, old growth forest structures provide the kind of age and physical variance that promotes diversity in bird communities (Haney et al. 1996).

Finally, the impact of old growth forests is felt from local-regional spatial scales to global prevalence: as powerful carbon sinks, old growth forests work as climatic stabilizers. Globally, forests store ~45% of terrestrial carbon- yet paradoxically, are a high source of annual carbon emissions due to hyper-deforestation (Ford and Keeton 2017). Qualities of old growth forests: structurally diverse and uneven-aged, also facilitate greater carbon capture than young, even aged forests (Keeton et al. 2011) The positive feedback loop of anthropogenic climate change transcends the temporal scale of old growth significance from a regional biodiversity pillar, to a player in global system stability. As Peter Wohlleben writes, in the *Hidden Life of Trees* (2016: 98) "If we want to use forests as a weapon in the fight against climate change, then we must allow them to grow old, which is exactly what large conservation groups are asking us to do."

The Human Imperative (Old Growth Redux)

The great forests of the past need not stay there. With a strategy of old growth preservation in tandem with old growth restoration- giving secondary forest time and space to grow old- we can repaint our landscape in shades of original ecosystem function. Yet we must do so within the framework of the Anthropocene, the mid-twentieth century divergence from the Halocene epoch into a new era, distinguished by the climatic, biologic and geochemic impact of human activity. Definitions of old growth as free of human disturbance (a prevalent differentiator between Carlson's class A and B stands) would be self-defeating, with human output detectable across virtually all space on earth. Rather, understanding old growth forests as those stands which supremely model natural forest structure, with uneven aged, canopy-gap phase occurrence, species diversity and the freedom for trees to fall and return to the ecosystem as ground habitat and a source of nutrition, will enable our forests to assume their most productive natural rhythms.

The mosaic of stands I present here as first identified by Carlson does not exist in a vacuum of perseveration; rather, they should be seen as foundations of a greater, forthcoming natural forest. For this to occur, we must consider connectivity of systems and forest corridors- and what happens when habitat is severed. An ecotone is a region of environmental transition. It's here that edge effects occur- changes in community and population structure at environmental boundaries. An ecotone can arise as a result of forest cuts, where an open terrain of former forest meets the canopy and growth of forestscape it was once was. A study to determine what kind of edge effect this type of ecotone would produce in a forest environment found that species retreated deeply into what was left of their original habitat, while only venturing shallowly into juxtaposing open space (Zurita et al. 2012). The study found that native habitats separated by 50 at least meters of open space, or 300 meters of disturbed forest essentially acted as geographic barriers, sequestering the creatures to their immediate range of native habitat (Zurita et al. 2012).

Great fragmentation arises in forestscapes through timber cuts; and this is the threat of the Southside timber sale. Not only do the few undisturbed stands of old growth in the watershed- unrecognized by Forest Service data- risk reduction into "high value timber," but potential old growth surrounding them, and the role they play as connective corridors for forest species, face potential devastation.

According to Paul Carlson, single-tree selection, a method of avoiding regional fragmentation through clear-cutting, is an ideal philosophy, yet, "I understand and empathize with large disturbance cuts," he said, "financially it's the only way you can do it" (Carlson, P., personal communication, 2017). A lack of workforce offering single-tree selection is a major issue for ecosystem-management-minded foresters, he said (Carlson, P., personal communication, 2017). He sees compounding issues with the commercial of timber cuts as being a viable economic player in southern Appalachia moving forward: "Stumpage values haven't increased in 30 years… logs aren't bringing much more value in terms of actual dollars, while the cost of forest management has gone up a lot, the equipment, the diesel, the labor" (Carlson personal communication, 2017).

Clear-cut normative forest management, the major agent of forest fragmentation in southern Appalachia, was largely driven by foreign interests, both northern and European (Davis 2000). Industrialists moving on from the resource-depleted northeastern United States envisioned Appalachia's mountains, still bountiful in forest and ore, as the grounds for a "South Pittsburgh."

Major players in the resource-extraction boom in Appalachia included Scottish-Canadian capitalist Alexander A. Arthur (investing in over 105218.3 hectares (260,000 acres) of land for the Scottish Carolina Land and Lumber Company), Englishman H.N. Saxton, who's business (Saxton Company) focused on hardwood exports to Europe and the Thomas Lumber Company, cutting timber for the Ohio Valley timber market (Davis 2000). While regionally born timber companies participated in the industrial logging epoch, a majority of the resources extracted from the region followed their financial backing to northern and foreign destinations (Davis 2000). And unlike the environmental impacts felt by the region, the financial gains from the extraction rush were fleeting for the people of Appalachia.

Approximately half a century after logging rates began to decline in the 1920's, a federally supported study on improving the economic viability of the Appalachian lumber sector called the region "an area of chronic depression encircled by poverty… (Lyons, 1969), and found a high amount of resources continued left the region: in 1962, 58 percent of all lumber produced in West Virginia was shipped out of state for further treatment. The study warned that any reduction of industrial labor-intensiveness, which happens over time through technological advancement, would lead to less jobs, mitigating any hopes of employment improvement (Lyons, 1969). These issues have proved pervasive. In modernity- according to the Appalachian Regional Commission's 2010-2014 report- the Appalachian poverty rate of 19.7% surpasses the national average, at 15.6%. On an intra-state level, stretches of Appalachia are disproportionately impoverished compared to the rest of a state; Kentucky, for example, has an average state poverty rate of 18.9%, while Appalachian Kentucky stands at 25.4% (Appalachian Poverty, 2017). Considering the ecologic implications of old growth in relation to the purportedly low economic ceiling for lumber and timber industries- from both future projections and historical precedent- for those who actually call Appalachia home, cutting old growth appears to be a myopic martyr. Conversely, there appears to be economic merit in preserving the region's unique resources.

A study on the Leuser National Park in Northern Sumatra provides a helpful template. Despite being a national park, it is continuously logged- the historic modus operandi. The study considered three prospects for the park's future: conservation, where logging ceases to operate and where eco-tourism is emphasized, deforestation, where present rates of logging continue, and selective use, where rates of logging of primary forest are reduced, and some attention to eco-tourism is given (Beukering et al. 2002). Their results are presented in terms of economic valuation. In a 30-year model, they concluded that a deforestation future produced $7.0 billion, a selective use future generated $9.1 billion and the conservation scenario created $9.5 billion (Beukering et al. 2002). Further, the conservation future propagated regional equality, whereas the money generated in the deforestation model disproportionately flowed up the corporate logging ladder, furthering the gap between rich and poor (Beukering et al. 2002).

By investing in their natural bounty of resources, Appalachia might flip the historic narrative. In keeping these unique resources in the ground, they can retain its economic potential locally, rather than watch it disappear with the harvested resources. By cultivating the landscape for long-term growth, for eco-tourism and ecosystem services, both landscape and people prosper, mutualistically.

This project aspires to provide a resource for old growth preservation and restoration; a compass for protecting forest corridors, for connecting relics of primordial forest stands to each other and all the diversity of life they harbor. Class B stands designated in 1995 likely have a case as class A old growth today, with over 20 years of additional growth. The same goes for class C, which have likely aged to class B. If we expand our definition of old growth outside of merely 'forests free from anthropogenic disturbance,' to consider all uneven aged, complex stands of forest that mirror native structure, including mature trees relative to their species-normative lifespan, to be a potential restoration site, then we can make further strides in empowering the southern Appalachian uplands to reach its greatest ecosystem function. However, I recommend not just preserving all identified stands of old growth in this watershed, but also fostering their connectivity. For example, with roadless areas in their surrounding environs, the stands of class B and C old growth on the gradients of Wilson and Double Knob can join stands running south all the way down to Rocky Mountain, in a restored pseudo-primary forest chain. The same opportunity exists for the myriad stands running south from the Ellicott Rock Wilderness, down to Whetstone Mountain.

There are presently no explicit federal protections for old growth forests on public land. The sanctity of the law old growth stands garner comes indirectly. Intimately linked to biodiversity and rare creatures who specialize in primary forest environ, old growth benefits from the National Forests Management Act of 1976, which requires the Forest Service to "provide for diversity of plant and animal communities," and to "maintain viable populations of existing native and desired non-native vertebrate species" (Sierra Club: 5). The National Environmental Policy Act of 1969 is a procedural mandate, requiring the Forest Service to act transparently, consider best available science- and involve the public- in their planning process (Sierra Club: 5).

We can define and understand old growth on a purer ecological basis- rather than simply as large trees- as an initial step in giving these important swathes of forest recognition as "diverse" and "native" under the National Forest Management Act of 1976. Old growth stands are not invariably large- they can be subtle and of average size. (Stahle 1996). While the Forest Service includes large size relative to species as a parameter, I recommend instead focusing on structural characteristics, as the Forest Service has: wide variation in tree sizes and spacing, accumulations of large-sized dead standing and fallen trees that are numerous relative to earlier stages (though not necessarily large-sized), decadence in the form of broken or deformed tops or boles and root decay, multiple canopy layers and canopy gaps and understory patchiness. I recommend deep, entrenched wrinkles in bark as a proxy large size for as an aesthetic indication of age. I also recommend that human disturbance not automatically deter old growth classification, considering its pervasiveness in the anthropocene; rather, old growth restoration and connectivity take on as much emphasis as preserving those most pristine stands.

In building off of Carlson's work by further exploring southern Appalachia for old growth, by groundtruthing Forest Service data, by observing chains of old growth running down mountain ridge spines, as seen in the maps digitized in this study; and by working to establish connective roadless areas, we might ensure a return to structural old growth character in the upperlands of southern Appalachia, and all of the ecosystem service that it provides. In these forests, we might know the land more intimately. It might earn greater purchase in our minds.

ACKNOWLEDGEMENTS

Thank you to the erudites: James Costa and Sarah Workman. First-rate naturalists, they are, moreover, first-rate people. Thank you to Nicole Hayler, an indispensable resource at the Chattooga Conservancy, and Buzz Williams, a transcendent mentor. They say that if you want something done, ask someone who is busy. This is Gary Wein, who helped make these maps possible. Thanks to Klio Stroubakis for joining me in the world of old growth study. And thank you to Paul Carlson, for jumping back in time to discuss his 1995 work with me.

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ASSEMBLY AND PRELIMINARY ANALYSIS OF A SMALL-SCALE ANAEROBIC DIGESTION SYSTEM

FOREST SCHWEITZER

Abstract. Anaerobic digestion is the process of breaking down organic material in the absence of oxygen. The result of the several reactions in this process is the evolution of biogas, a mixture of primarily Methane and Carbon Dioxide. Biogas is flammable and can be used to produce heat and/or electricity through its combustion, with a variety of applications, making anaerobic digestion an attractive alternative to natural gas. I designed and built, with the help of Timm Muth of the Jackson County Green Energy Park, a functioning anaerobic digester. A second unit, produced by HomeBiogas, was also purchased and assembled with the idea of comparing the two systems. Though the testing of these two systems is beyond the scope of this study, the ease and cost-effectiveness of assembly along with the portability of the unit make it an excellent tool for public education and outreach.

Key words: alternative energy, anaerobic digestion, biogas, community education, DIY assembly, Jackson County Green Energy Park, outreach, sustainability.

INTRODUCTION

Anaerobic digestion is the process of biogas production (of which Methane is the key element) by means of bacterial decomposition. A series of four processes — Hydrolysis, Acidogenesis, Acetogenesis, and Methanogenesis — break down the waste material into a number of different gases, primarily Methane (CH_4), Carbon Dioxide (CO_2), molecular Nitrogen (N_2), and molecular Hydrogen (H_2) (Beychok 1967). Systems can vary over several variables: temperature that the reactions occur at, amount of solids versus liquids in the input material, if production of biogas is continuous or in batches, number of phases present for the reactions to take place in, and whether or not the design of the system permits introduction of "inhibitors" in order to control reaction time.

It was first observed in the 17th century that flammable gases evolve from the decomposition of organic matter. In 1776, Alessandro Volta discovered that with the increase of input matter results in a correlated increase of evolved gas. Early in the 19th century, Sir Humphrey Davy concluded that Methane was present in gases resulting from anaerobic digestion. The first digester was built for a leper colony in Bombay (today Mumbai), India, in 1859 (Meynell 1976). Over the coming decades, the technology spread and diversified, first landing in England and moving on to Germany and other continental European nations. Though a bevy of research was conducted during these intervening years into the mechanisms behind anaerobic digestion and methods to increase gas yields, practical implementation of systems was not seen on a large scale until World War II. The second World War brought on severe energy shortages and necessitated the development of alternative sources for power. During this period, anaerobic digestion technology matured greatly and resulted in understanding, if not yet the materials, to produce systems capable of offsetting tradition means of power generation (McCabe 1957).

My project was based at the Jackson County Green Energy Park (JCGEP, Jackson Co., NC), a county initiative designed to capture methane from a former landfill for use in firing blacksmith forges, glass-blowing, other applications. One of the primary aims of the JCGEP is to educate the county as to the benefits of alternative energy. Accordingly, I helped design and assemble a home-made anaerobic digester that can be used for educational demonstrations.

The primary goal of building the digester is use for educational purposes. The system has therefore been built out of lightweight plastics and is mounted onto a wheeled pallet. This was done to enable the system, which, at full volume will weigh around one thousand pounds, to be pushed safely by two people with minimal effort. Portability will allow the digester to be taken to local schools, festivals, and other sites in order to provide hands-on learning experiences for community members. The more distant goal of the project is to inspire community leaders to invest in an industrial-scale digester which would reduce the carbon footprint of Jackson County while simultaneously bringing revenue and jobs to an area in need of economic stimulation. A secondary goal is to compare the home-made unit with a commercial pre-fabricated unit. This comparison would generate data regarding the efficiency of the system and what modifications may be made to augment production of biogas.

MATERIALS & METHODS

The goal of this study is the assembly and operation of an anaerobic digester as a test unit for educational demonstration. After a monitoring period, extending beyond the reach of this paper, the results will be used as part of a proposal to the Jackson County Commissioners to begin consideration of a community-scale system. In the near term, the digester will be used as an educational tool for local schools and community organizations. Specific design choices were made to ensure that the system is as robust and portable as possible, while not sacrificing efficiency or net productivity.

Materials

For the construction and assembly of the digester, the following materials were used:

- Three 208.2 liter (55 gallon) plastic drums
- 3.05 meters (10 feet) of 1.9 centimeter (¾ inch) PVC
- 0.6 meters (two feet) of 5 centimeter (2 inch) PVC
- Three 1.9 centimeter (¾ inch) bulkhead fittings
- Four 1.9 centimeter (¾ inch) valves
- 2.4 meters (eight feet) of garden hose
- 1.9 centimeter (¾ inch) brass hose clamps
- Spray-paint (black, yellow, orange, green, red, purple)
- Bung tool
- Two 0.6x3.7x2.4 meter (2x12x8 feet) wood beams
- Two 0.6x1.8x2.4 meter (2x6x8 feet) wood beams
- Eight 10 centimeter (four inch) screw eyes
- Eight ratchet straps
- Caster wheels (15 centimeter/6 inch diameter)

Design Phase

The digester consists of three 208.2 liter (55 gallons) barrels attached in tandem to one another by first a PVC pipe and then a common garden hose which has been fitted out with NPT standard male heads. The first barrel has a 1.21 meter (four feet) long, 5.08 centimeter (two inch) wide capped PVC pipe fitted in at the top to provide a means of inserting organic matter into the system. There is also a release valve approximately 12 centimeters (five inches) from the bottom of the barrel in order to discharge nitrogenated fertilizer once the methanogenic process is complete. This digester barrel, the one in which the reaction takes place, is connected to a second barrel by an 2.43 meter (eight feet) long, 1.9 centimeter (3/4 inch) wide PVC pipe attached at the top of both barrels. The methane generated in the first barrel travels through this pipe, passing a check valve to prevent its regression, through into the second barrel. Figure 1 depicts the course of waste material through the system, from input, through gas conversion and flow through to the output valve.

This second barrel possesses a 1.9 centimeter (3/4 inch) wide capped PVC pipe inserted into the top for venting of methane, and a 1.9 centimeter (3/4 inch) bulkhead fixture at its base that will sport the aforementioned garden hose. The second barrel is filled with water and connected to the third via the hose. The final barrel is elevated to such a level that the bottom of the third barrel will be approximately flush with the top of the second barrel. The final barrel also contains water. The second and third barrels act as a means of producing pressure for the methane. As the biogas proceeds into the second barrel, it increases in volume, pushing downwards on the water in that barrel. This positive pressure forces the water through the hose, upwards and into the third barrel. The positive, upward pressure is countered by the water in the third barrel being pulled downward by gravity and back into the second barrel, producing and sustaining pressure on the methane. In addition to producing biogas, the digester will also produce a slurry byproduct. The slurry, known as digestate or “compost tea”, is very rich in nitrogen and phosphorous that could not be otherwise broken down during methanogenesis, and is for this reason ideal for application as fertilizer. For this reason anaerobic digestion pairs well with farming as the material generated on the farm can be fed into the digester for power and fertilizer.

In addition to the model that we designed and assembled, a unit was purchased from HomeBiogas (HomeBiogas 2017), an Israeli company that produces pre-fabricated anaerobic digesters for home use. This second unit was assembled and will be monitored in the future to confirm figures of input vs. output and how much energy can be produced daily. Though, at this time, a true comparison study is not possible, basic calculations were made to compare the cost of the two units when correlated with their efficiency. Assumptions and predictions were made regarding the performance of the designed model, and confirmation of HomeBiogas reported gas evolution values has yet to be attained.

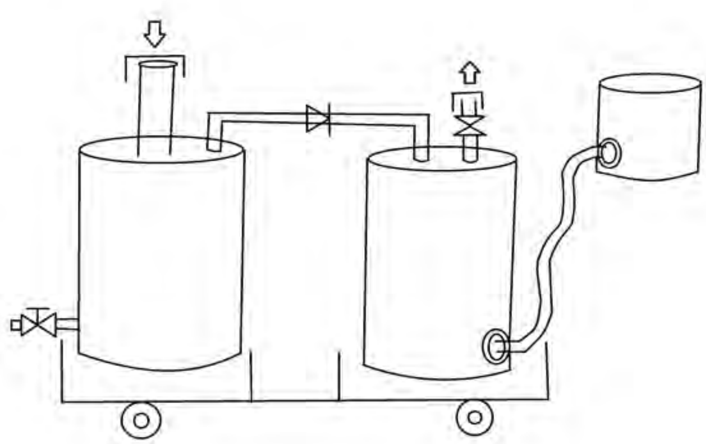

FIG. 1. A diagram of the self-assembled system. Organic waste is introduced to the barrel on the left. In this chamber the four requisite reactions occur to produce the biogas, which then travels through the connecting PVC pipe into the second barrel. In this barrel the gas collects under pressure of the water, which, present in both the second and third barrel, attempts to exert force against the collecting gas by traveling through the connecting hose to reach a level.

FIG. 2. Design detail of anaerobic digester built in the study. Top left: the input tube. Organic waste material enters the system by means of this tube. The cap prevents oxygen from entering the reaction vessel and noxious odors from escaping. Bottom left: view of the check valve which controls gas volume moving between the reaction and collection chambers. Until gas is needed, the valve is left closed, retaining the gas inside of the pressurized collection tank. When the valve is set to open, gas begins to leave through the nozzle (above the valve, not pictured) and into a secondary collection vessel or directly to a gas burner. Top right: hose and valve controlling the flow of water from the gas collection chamber into the water holding chamber, third in series, which pressurizes the biogas held in the second chamber. Middle right: bulkhead fitting connecting the PVC of the reaction chamber and gas collection chamber. This fitting enables the two to be separated during cleaning. Bottom right: entrance

point of water into water collection chamber. The visible metal clamp ensures that no water leaks from the hose during transfer.

Data to be collected in future evaluation of digester:

- Methane content and volume
- Monitoring gas pressure over time
- CO_2 and Oxygen content
- Monitoring CO_2 vs. methane (CH_4) over time
- Volume waste input vs. energy yield output
- Types of waste
- Digester temperature.

RESULTS

The primary result of the project was the production of a home-made anaerobic digester (Figure 2). The digester is equivalent in size and predicted production with the HomeBiogas unit, making comparisons of efficiency between the two apt and productive (Figure 3). Table 1 displays production values compared with initial investment for these systems, along with an estimate of how long each system would take in order to pay itself off through use. These values are not based on actual tests as yet, but are instead based on information provided by HomeBiogas and predictions given comparative similarity of the two systems.

For the purposes of this study, a small, single-stage digester was built out of three fifty-five gallon drums. Ease of use and portability were favored (Chen 2008). The first batch of gas produced consisted primarily of CO_2 This is due to the fact that, when first inoculating the primary barrel and sealing it with the cap, there will be air which remains in the barrel. The bacteria introduced utilize the oxygen present in an aerobic reaction to produce CO_2 instead of the intended methane (Meynell 1976). This is unavoidable in such a setup without evacuating the barrel. It is also impractical and unnecessary since, after this first batch of CO_2 is purged via the gas pipe, the system will be devoid of oxygen (the oxygen originally present having been used up in CO_2 production) and methanogenesis commences (Zhang 2014). An RKI GX-2012 gas analyzer (RKI Instruments, Inc.) will be utilized in the testing and recording of methane produced throughout the experiment. The data collected from the analyzer was used to determine exactly how much mass in organic matter was required to produce a target figure of 2 hours daily use of a stovetop burner.

Table 1. Projected output volume of gas vs. initial investment cost.

System	Cost	Input Waste Volume	Output Gas Volume	Hours on Burner	Time until system pays off
HomeBiogas	$1090	1 kilogram	200 liters	1	6,667 hours/277days
Homemade	$200	1 kilogram	200 liters	1	1,333 hours/55 days

FIG. 3. a) Pre-fabricated digester produced by HomeBiogas. b) Home-made digester assembled in this study

DISCUSSION

According to a national study conducted by the Environmental Protection Agency (EPA) in 2013, Americans produce two kilograms (4.40 pounds) of food waste per-person-per-day (Municipal Solid Waste 2013). According to HomeBiogas, their pre-fabricated system should produce, for a two- to four-person household, two hours of cooking time per-day. This is based on the fact that one kilogram of input organic matter yields, on average, 200 liters of gas, and one person produces two kilograms of each day. With that in mind, the HomeBiogas system should produce 400 liters of gas each day, which transfers to 2 hours of cooking on an average stovetop burner. It is true that in a four-person household, assuming that the average person produces two kilograms of waste per day, such a house should produce eight kilograms of waste per-day, yielding 1,600 liters of biogas, and seven to eight hours of cooking time, but the system is not built to process this volume of material.

The average stovetop burner uses approximately 1500 watts each hour, with the average utility charging $0.10 per kilowatt hour (Energy Use Calculator 2017). This means that each hour of cooking on a stovetop burner produced by biogas offsets the power bill by $0.15. Summing up these hourly values, one discovers that for the HomeBiogas system it would take 6,667 hours, or 277 days of use on the burner to pay off the initial investment on the digester. For the home-assembled unit, the figures are drastically lower at only 1,333 hours, or 55 days in order to pay the unit off. In addition, one must consider that after the unit has paid itself off in valued use, it begins essentially making a profit by offsetting annual power bills. The power generated by the anaerobic digester is not tied to a power utility and is generated on-site, meaning that, even given a natural disaster, the production of biogas should continue steadily. In this scenario dependence on utilities for energy production is decoupled and energy security is improved greatly. This is still without considering the environmental implications of recycling the energy in waste matter and lessening dependence on traditional, natural gas options.

The nature of anaerobic digestion is such that it requires an environment devoid of oxygen, such as that created by submersion in water. If the reaction is taking place underwater the requirement for container volume and load-bearing capacity is greatly increased. To keep the system small, light-weight and easy to assemble, HomeBiogas opted for a lower-capacity system, but one that is more likely to find its way into a larger number of homes, spreading awareness of the technology and benefits of anaerobic digestion.

The system that we designed and assembled has a similar reaction chamber volume (208.2 liters (55 gallons) in our system vs. 197.27 liters (50 gallons) in the HomeBiogas system), and is also targeted towards small, private applications such as household or small business use. For these reasons the two systems are very comparable, with the primary distinguishing difference being upfront cost. The HomeBiogas system costs a total of $1090 as of Fall 2017, whereas the homemade system cost a total of $200 to build. Though it has not yet been tested, the two systems should perform equivalently, making the case for the less expensive of the two. HomeBiogas is one of the few companies out there producing pre-fabricated anaerobic digestion systems for the consumer market. This means that not only are they able to charge more than if there were other, competing companies, but this lack of competition further stymies development of potentially cost-lessening technologies to facilitate the process.

Our system was put together using basic, available parts that can be found at any local Walmart, Lowes, or Home Depot store. The cheapness and ease of assembly illustrates just how simple a concept anaerobic digestion truly is. The point of our system is use as an educational implement. The system was designed to be as lightweight and portable as possible given our budget and available materials. In this way it is ideally situated for placement in the bed of a truck or tractor trailer for ease of transport to various establishments (schools, county offices, etc.). By familiarizing the public with the system, demonstrating how easy and cheap (less than $200 total) it is to assemble, how it might be modified for different uses (for example, as a home system, or in a school, or perhaps even more simply in a cabin or barn), we will be engaging the citizens of Jackson county in a discussion about their energy future.

Even outside of the United States the power of anaerobic digestion is being realized. The producer of the HomeBiogas unit is an Israeli company situated in Tel Aviv. An enormous number of people in the developing world still lack the power infrastructure that makes transportation, education, and equity attainable goals for all citizens (Barnett 1978). In South America, Africa, and the Middle East, personal-use digesters are being made with everything from plastic bags and barns filled with dung to highly advanced systems capable of powering entire towns and cities (Jingura 2009). Not only is there high demand for these systems in the developing world, but their lack of infrastructure actually puts them at a uniquely advantageous position when it comes to adoption of alternative energy technology. Without the burden of power stations, transformers and cables stretching across the county, communities in need of power need not jump through regulatory hurdles to approve an anaerobic digester installation, and the on-site generating nature of such systems makes them ideal for geographically disjunct communities which are otherwise left out of the development discussion altogether. In this way citizens are not forced to wait for potential future improvement in quality of life. The installation of an anaerobic digester provides the means of dealing with food waste, produces biogas for the generation of electricity, and a byproduct of this gas production is a nitrogen rich "compost tea" that may be used as a potent fertilizer.

CONCLUSION

This project was successful in designing and assembling a functioning anaerobic digester. While that in itself is a crucial first step, more research must be conducted into how this system fares over the long term. Questions such as what temperature range it will function over, ideal temperature range, and structural elements that may be added or modified in order to increase functionality all remain to be addressed. Furthermore, an in-depth analysis of how our system compares to the pre-fabricated assembly produced by HomeBiogas would serve to show the public the benefits of pursuing self-assembly.

With education and outreach comes the potential to garner widespread support for anaerobic digestion through Jackson County and beyond. A future recommendation would be to look into promoting to the county commissioners an industrial scale system. If such a system were invested in, not only could the carbon footprint of the county be offset by a reduction on natural gas, but revenue could be gained by soliciting neighboring counties to send their waste to Jackson County to be processed (Berglund 2006). In this way the county could tax the imported waste and sell excess biogas.

Furthermore, there is a great dearth of scientific literature documenting the proliferation of anaerobic digestion technology through the developing world. Rural communities in areas detached from a national grid are opting for community-scale digesters because of their stability of production, ease of assembly, and the independence which comes with producing one's own energy (Betterley 2014). Studies looking into what communities are taking these steps might help in the development of regionally-specific digestion advancements. Anaerobic digestion is currently only efficient between ambient temperatures of 20 and 45 degrees Celsius (68 and 113 degrees Fahrenheit) (Kim 2006). This limit of temperature means that certain parts of the world, particularly very arid Middle Eastern and North African climates, present challenges to the adoption of anaerobic digestion as a means of energy production. In producing systems capable of running efficiently over a larger temperature range, the benefits of the digester are made available to millions of people.

ACKNOWLEDGEMENTS

I would like to thank Timm Muth, Chelsea Miller, and Kevin McNiff, and all the artists at the Jackson County Green Energy Park for their support and encouragement. Without them my semester would not have been nearly as rich or productive. Additionally, I would like to give a hearty thanks to Dr. Jim Costa and Dr. Sarah Workman for taking the time not only to teach and inspire me but to read and edit all of my assignments and papers. Their attention to detail and passion for teaching was as impressive as the mountain landscape that surrounds us here in Highlands. Further thanks must be extended to Dr. Gary Wein, Brent Martin, Dr. Tom Martin, Dr. Mark Lord, and Patrick Brannon for enriching my semester with a new knowledge of Land Ethic, of plants and animals, and of the mountain ecosystem as a whole.

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CARBON STOCK IN ABOVEGROUND BIOMASS OF POTENTIAL OLD GROWTH STANDS, SOUTHSIDE TIMBER SALE, NANTAHALA NATIONAL FOREST

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Abstract. Old growth forests are invaluable due to the role they play in the environment, especially when considering the amounts of carbon that are sequestered within these rare ecosystems. Allometric equations were utilized to estimate carbon within the live and standing dead trees in this study for two stands in the Nantahala National Forest (NNF). Of the 121.4 ha proposed to be cut in the Southside Timber Sale in the Forest, this study focuses on two stands, Granite City (NNF Stand #31/18) and Brushy Mountain (35/41), totaling over 20 ha. Several parameters were measured to calculate the aboveground biomass within these stands and to evaluate whether these stands could be considered old growth, including: basal area, median number of snags, dbh of largest tree, and stand density. The total biomass in the Granite City site including both standing live, dead, and snag trees was 162,180 kg, and the total biomass in the Brushy Mountain site was 71,396 kg. The carbon stock for Granite City was 76,225 kg and in the Brushy Mountain site was 33,556 kg. Based on basal area, median number of snags, dbh of largest tree, and stand density, the Granite City stand qualified as old growth based on the USFS's old growth guidance from 1997. Brushy Mountain, currently classified as a dry-mesic oak forest, fits more closely to a western mesophytic forest type, and thus the stand would qualify as old growth based on stand density, dbh of largest tree, number of standing snags per acre, and minimum basal area.

Key words: aboveground biomass, allometric equations, carbon sequestration, old growth forests, timber harvest

INTRODUCTION

Old growth forests are important for their roles as biodiversity reserves, habitat for wildlife, human recreation, places of aesthetic beauty and cultural value, as well as their contributions to soil productivity, water quality, aquatic habitat, research, and carbon storage (USFS 1997, Bragg and Shelton 2014, USFS 2015, Ford and Keeton 2017). Old growth forests are especially rare in the southeastern United States where approximately 0.5 % of total forest acreage exists as old growth due to intensive logging during the early 20th century (USFS 1997, Sierra Club 2005). Even though millions of acres of old growth were lost to logging more than a century ago, there are existing forest stands approaching what is called secondary old growth, where forest stands are in the process of recovering and developing characteristics that can be classified as old growth (Scheff 2012, Noormets et al. 2015). Although the definition of what constitutes old growth is hotly contested, the US Forest Service (USFS) Chief in 1989, Dale Robertson, defined old growth forests as "ecosystems distinguished by old trees and related structural attributes. Old growth encompasses the later stages of stand development that typically differ from earlier stages in a variety of characteristics which may include tree size, accumulation of large wood material, number of canopy layers, species composition, and ecosystem function (USFS 1997)." Further, the criteria for what determines whether or not a stand is old growth is empirically delineated by the USFS, and once a stand is determined to be old growth, it is meant to be managed differently by the USFS (USFS 1997, USFS 2004, Sierra Club 2005). The USFS has established old growth guidance and strategy plans for the southeast, but since there are no federal laws protecting old growth forests on national lands, these are simply recommendations (Sierra Club 2005). Part of the old growth management strategy includes developing a network of old growth areas, considering the representation of all 16 possible old growth forest community types in the southeast, and managing the distribution and the linkage of old growth patches (USFS 1997).

Currently, deforestation is the third highest source of global carbon emissions (Keith et al. 2009, Ford and Keeton 2017), augmenting greenhouse gases in the atmosphere that trap heat and thus, contributing to climate change. In harvesting operations, 50 % to 80 % of tree carbon is removed (Gorte 2009), and by disturbing the soils through commercial timber harvesting, the carbon storage capacity of the soil is compromised (Gorte 2009, Noormets et al. 2015). A study conducted by Noormets et al. (2015) shows that forest soils are losing more carbon than they gain annually and this loss is most pronounced in managed forests. According to Harmon et al. (1990), during timber harvests, carbon storage for a site is reduced greatly, by approximately 30 +/- 6 % (Noormets et al. 2015), and does not "approach old growth storage capacity for at least 200 years," even when including the sequestration of carbon in timber products in their calculations; ultimately, timber harvesting culminates in a net flux of carbon into the atmosphere (Harmon et al. 1990). This is particularly important in US forests because approximately two-thirds of all the carbon in a forest is stored in the soil, and the USFS has defined 106.8 million hectares (246.4 million acres) of land as available for timber harvesting (Gorte 2009).

Although only 0.017 % of the earth's land surface is old growth, old growth forest conversion has accounted for 2 % of total carbon released globally due to land use changes over the past century (Harmon et al. 1990). A study by McGarvey et al. (2015) found evidence that carbon may continue to accumulate in old growth stands for over 200 years, which reiterates the importance of conserving forest stands that are functioning as old growth. Some studies suggest that old growth stands are not accumulating carbon, i.e. they are carbon neutral, and that instead, it would be better to harvest old growth forests and replace them with young, fast-growing forests in order to accumulate carbon (Odum 1969, Murkowski and Stevens 1989). Whereas other studies have found that the conversion of old growth forest to young forests will not decrease atmospheric carbon dioxide (Harmon et al. 1990, Luyassaert et al. 2008) and that old growth forests are actually functioning as carbon sinks (Keith et al. 2009, Hudiburg et al. 2009, McGarvey et al. 2015, USFS 2015, Ford and Keeton 2017). Following this thinking, the conversion of old growth to younger forests will add carbon to the atmosphere because it will take a long time for new plantings to sequester and store an equivalent amount of carbon as mature forests (Harmon et al. 1990; Keith et al. 2009).

Southside Timber Sale

The Southside Timber Sale is a proposal to harvest 142.4 hectares (352 acres) of forest; the majority of which will be a variation of an even-aged timber harvest, called a two-aged regeneration harvest (USFS 2017, Chattooga Conservancy 2017). The publicly available data, CISC (Continuous Inventory of Stand Conditions), was found to be inaccurate when classifying these stands (Carlson 1995). A number of the stands proposed for harvest were noted as old growth, and if that is the case, should not be cut according to USFS guidelines regarding old growth restoration (USFS 1997, Sierra Club 2005). The two sites I focused on were Granite City (31/18) and Brushy Mountain (35/41). The numbers in parentheses are the specific stand numbers that have been assigned by the Nantahala National Forest (NNF).

This research project explored the characteristics of the two stands at Southside in the NNF, and used data collected about dbh, number of snags, coarse woody debris, and tree species richness to evaluate whether these stands should be treated as old growth and how much carbon is stored currently in aboveground biomass. The importance of the carbon stock assessment is that a timber harvest would compromise the carbon storage potential of these forest stands for another 200 years (Harmon et al. 1990). A quantified value of the current carbon storage in these stands can provide weight as to why they should be left standing, and intact, as old growth forest.

METHODS

The plots in this study were chosen to represent the different characteristics within the sites selected for the Southside Timber Sale. After reconnaissance in the two sites and noting how the forest character varied (e.g. where aspect changed, number of canopy gaps, differences in slope within the site, etc.), we placed plots based on the differences within the site itself. The plots were circular with a 13 meter radius containing four transects extending from the center of the circle toward the cardinal directions, with the exception of one plot. The first plot we measured was a circle with a 30 meter radius. The 13 meter radius plots were 0.053 ha (0.13 acre), and the 30 meter plot was 0.28 ha (0.698 acre). The 0.053 ha plots proved sufficient to represent vegetation diversity because these forests are relatively dense. Within the plot, all live trees greater than 2.5 centimeters in diameter were measured at breast height (dbh) and the tree species was recorded. I excluded rhododendron and mountain laurel from the data collection because I considered them as more of a shrubby vegetation rather than tree vegetation.

Basal area was calculated with the following formula:

$$BA = 0.00007854 \times dbh^2 \quad \text{in meters}^2.$$

With this data, the live aboveground biomass was calculated using allometric equations from Martin et al. (1998) and Jenkins et al. (2003):

$$\text{Biomass} = 10^{(a+b\ log_{10}(dbh))} \text{ (Martin et al. 1998)}$$
$$\text{Biomass} = Exp\ (\beta_0 + \beta_1 \ln(dbh)) \text{ (Jenkins et al. 2003).}$$

The allometric equations in the Martin et al. (1998) paper are more specific to this region in comparison to Jenkins et al. (2003) that can be used for tree species nationwide. I used both equations and compared the differences of the biomass calculations for each tree species. For the seven species without a specific coefficient to be used in the biomass calculations, I used the general tree coefficients that can be applied to all tree species for the Martin et al. (1998) equation. For the Jenkins et al. (2003) equation, I put those species in the most fitting grouping established in that paper (i.e. American chestnut fit most closely with hard oaks and maples, whereas Carolina silverbell fit best under the mixed hardwoods group).

Within the plots, standing dead trees (snags) and snags leaning on other trees (i.e. trees where the trunk of the tree is not touching the forest floor) were accounted for as long as they were at least 2.5 centimeters dbh, and the species were recorded if it could be identified. In order to assess the carbon stock of standing dead trees, I used the equations from Martin et al. (1998) that estimate the total wood mass that allocates the mass by stem wood, branches, bark. Depending on the condition of the snag, different aspects of the Martin et al. (1998) equations were used to more accurately represent its biomass. For example, a snag with no branches or bark would have only the stem wood mass to account for, and thus a more moderate estimate of biomass. In order to calculate the dead aboveground biomass, USFS FIA protocol calls for coarse woody debris, defined at 7.5 cm width at the transect intersection. I collected data for the coarse woody debris on these plots (not presented in this paper), however due to time constraints I was unable to include this set of biomass calculations.

In order to convert the biomass calculated from these equations to carbon, a conversion factor for the live, standing and downed dead wood must be used, which is:

Carbon (kg) = biomass (kg) * 0.47 (Casarim and Grais 2013).

Within the plots, at least one tree was cored, usually an abundant species that seemed to dominate the stand, had a large dbh, or seemed less likely to have heart rot or other disease that would compromise the ability to read the core. Usually, two people estimated cores on-site separately and the reported age was the average of two estimates. The increment borer used at Granite City was about 30.48 cm (12 inches), so not all of the trees that were cored hit the pith. With that in mind, all of the ages for those trees can be assumed to be at least as old as the age that is reported, and more likely, older than the core. While at Brushy Mountain, we used a larger increment borer and were able to hit the tree pith on those cores and report those ages with more certainty. Three cores were brought back to be analyzed more thoroughly for age estimates, and those were sanded and then glossed with glycerin.

USFS Old Growth Criteria

Old growth is delineated by three different categorizations by the USFS: existing old growth, future old growth, and possible old growth. In order for forest stands to be classified as existing old growth, they must meet all four criteria (age, disturbance, basal area, tree size) for a specific forest community type (USFS 1997). Future old growth are forest stands that the USFS has decided to manage as old growth, but the stand did not meet one or more of the old growth criteria (USFS 1997). Finally, possible old growth is also a forest stand that does not meet one or more of the criteria, but has not been allocated as future old growth, which results in no land management decisions related to age for how to treat these stands (USFS 1997). The USFS criteria for disturbance in old growth stands states there should be no obvious evidence of past human disturbance that would conflict with old-growth characteristics in an area (USFS 1997).

The USFS has basic guidelines for stands to be considered old growth based on 16 different old growth forest community types. The two study stands have been classified by the NNF as either conifer-northern hardwoods or dry-mesic oak. Table 1 provides a comparison of old growth characteristics for these two forest community types.

TABLE 1. Basic guidelines with characteristics that determine what constitutes old-growth based on different forest types (USFS 1997).

Old-Growth Community Type	Minimum Stand Age (years)	dbh of largest trees	Minimum Basal Area (m^2/ha)	Median number of standing snags per acre
Conifer-northern hardwood	140	> 20 inches > 50.8 cm	9.18	6 to 73
Dry-mesic oak	130	> 20 inches > 50.8 cm	9.18	26 to 36

Mixed and Western mesophytic	140	> 30 inches > 76.2 cm	9.18	4 to 28

USFS Stand Characteristics of Granite City and Brushy Mountain

Granite City (31/18) is a stand of 8.69 ha (21.48 acres) and is classified as a conifer-northern hardwood forest, as subtype 2b: white pine- northern hardwood forest (Appendix A). Conifer-northern hardwood forests tend to have north or east facing slopes, where at least 25 % of overstory canopy is coniferous or deciduous trees (USFS 1997). Disturbance in this forest type is related to historical fire patterns, where fire plays a central role in maintaining eastern white pine in subtype 2b (USFS 1997). The most common species in this forest type are red maple (*Acer rubrum)* and red oak (*Quercus rubra)* on dry sites, whereas on moist sites, sugar maple (*A. saccharum*), beech (*Fagus spp.)*, white ash (*Fraxinus americana)*, and eastern hemlock (*Tsuga canadensis)* are more common (USFS 1997). The specific attributes of old-growth for this forest type are delineated in the old-growth guidance from 1997 (Table 2).

TABLE 2. Attributes for four tree species in an old-growth conifer-northern hardwood forest community: eastern hemlock (TSCA), white pine (PIST), red spruce (PIRU), and sugar maple (ACSA) (USFS 1997).

Old Growth Attribute	Old-Growth Parameters
Mean age of large trees	TSCA: 147-264 years PIST: 153 to 272 years PIRU: 97 to 129 years ACSA: 114 years
dbh of largest trees	TSCA: (15 to 51 inches) 38.1 cm - 129.54 cm PIST: (28 to 50 inches) 71.12 cm - 127 cm PIRU: (6 to 28 inches) 15.24 cm - 71.12 cm ACSA: (38 inches) 96.52 cm
Stand Density (trees per acre)	dbh ≥ (4 inches) 10.16 cm: 91 to 475 ≥ (20 inches) 50.8 cm: 3 to 33 ≥ (28 inches) 71.12 cm: 0 to 10

Brushy Mountain (35/41) is a stand of 10.43 ha (25.79 acres) and is classified as a dry-mesic oak forest (Appendix B). Dry-mesic oak forest types are typically found on dry, upland sites on southern and western aspects and ridgetops (USFS 1997). The major species found in this forest type are chestnut oak (*Q. montana*), red oak (*Q. rubus*), black oak (*Q. velutina*), white oak (*Q. alba*), and scarlet oak (*Q. coccinea*) (USFS 1997). Other species that can be found are southern red oak (*Q. falcata*), post oak (*Q. stellata*), blackjack oak (*Q. marilandica*), pignut hickory (*Carya glabra*), mockernut hickory (*C. tomentosa*), and red maple (*Acer rubrum*) (USFS 1997). Coniferous species like shortleaf pine, table mountain pine, and white pine comprise less than 25 % of the overstory coverage (USFS 1997). Dry sites in this community type are prone to recurring, low intensity surface fires. Another important form of disturbance in this forest type are blowdowns, where single or multiple trees fall and result in gap phase regeneration (USFS 1997). Other forms of disturbance include tornados, oak decline, gypsy moths, and ice storm damage (USFS 1997). The old growth attributes for this forest type are outlined in Table 3.

TABLE 3. Attributes for six tree species in the old-growth dry-mesic oak forest community type in the Southern Appalachians: white oak (QUAL), red oak (QURU), black oak (QUVE), chesnut oak (QUMO), mockernut hickory (CATO), and pignut hickory (CAGL) (USFS 1997).

Old Growth Attribute	Old Growth Parameters
Age of large trees	QUAL: 245 - 348 years QURU: 240 - 270 years QUVE: 180 - 211 years QUMO: 66- 362 years CATO: 335 years CAGL: 327 years
dbh of large trees	QUAL: (14 - 27 inches) 35.56 cm - 68.58 cm QURU: (22 - 26 inches) 55.88 cm - 66.04 cm QUVE: (18 - 26 inches) 45.72 cm - 66.04 cm QUMO: (14 - 22 inches) 35.56 cm - 55.88 cm *Carya* spp: (14-26 inches) 35.56 cm - 66.04 cm
Stand density (trees per acre)	dbh ≥ (4 inches) 10.16 cm: 251 - 401

RESULTS

Recording species present and the diameter breast height (DBH) of all live trees greater than 2.5 centimeters on the 0.053 ha plots provided an estimate of composition of each forest stand (Table 4). In total, Granite City had 17 tree species and Brushy Mountain had 15 tree species. The largest dbh in Granite City was a white pine (*P. strobus)* at 88.1 cm, and the largest dbh in Brushy Mountain was a tulip poplar (*Liriodendron tulipifera)* at 72.1 cm.

In the Granite City (31/18) plots, the basal area totaled 17.43 m^2 per 0.386 ha (0.954 acres), not including standing snags. In the Brushy Mountain (35/41) plots, the basal area totaled 7.805 m^2 per 0.159 ha (0.393 acres), also not including standing snags. Including the standing snags data increases the basal area for Granite City total to 17.92 m^2 per 0.386 ha, and the basal area for Brushy Mountain total to 8.735 m^2 per 0.159 ha.

TABLE 4. A list of the 22 tree species encountered at Granite City and Brushy Mountain, including the common name, by alphabetical order of scientific name, with the species code, and the average DBH by species.

Common Name	Scientific Name	Code	Average dbh
Unknown	*Unknown*	UNK1	3.1
Striped Maple	*Acer pennsylvaticum*	ACPE	2.8
Red Maple	*Acer rubrum*	ACRU	11.3
Yellow Birch	*Betula alleghensis*	BEAL	4.5
American Chestnut	*Castanea dentate*	CADE	5.2
Hickory	*Carya* spp.	CARYA	13.1
Flowering Dogwood	*Cornus florida*	CORNUS	2.8
Carolina Silverbell	*Halesia caroliana*	HACA	6.6
American Holly	*Ilex opaca*	ILOP	8.0

Tulip Poplar	*Liriodendron tulipifera*	LITU	22.9
Cucumber Magnolia	*Magnolia acuminate*	MAAC	3.4
Fraser Magnolia	*Magnolia fraseri*	MAFR	5.2
Black Gum	*Nyssa sylvatica*	NYSY	10.1
Sourwood	*Oxydendron arboretum*	OXAB	28.0
White Pine	*Pinus strobus*	PIST	17.7
Black Cherry	*Prunus serotine*	PRSE	65.8
White Oak	*Quercus alba*	QUAL	42.3
Chestnut Oak	*Quercus montana*	QUMO	31.9
Red Oak	*Quercus rubra*	QURU	39.7
Black Locust	*Robinia pseudoacacia*	ROPS	14.0
Sassafras	*Sassafras albidum*	SAAL	4.6
Eastern Hemlock	*Tsuga Canadensis*	TSCA	8.4

The trees cored at Granite City included: a northern red oak (dbh: 50.9) that was at least 110 years old, another (dbh: 37.9 cm) that was at least 115 years old, and a third (dbh: 69.5 cm) that was at least 145 years old; along with a white oak (dbh: 41 cm) that was 127 years old (Table 5). The trees that were cored at Brushy Mountain included: a black cherry (*P. serotina)* (dbh: 65.8 cm) that was 179 years old; a tulip poplar (dbh: 72.1 cm) that was 90 years old; a sourwood (*O. arboreum)* that was also 90 years old; and a white oak (37.8 cm) that was 211 years old (Table 6). Comparing the mean ages of the large trees, white oak at Brushy Mountain seems to miss the old growth mark by a handful of decades; however, when considering the dbh of the largest trees, all of the trees fall inbetween the ranges provided—and the northern red oak (*Q. rubrum)* actually exceeds the diameter range at 66.6 cm dbh (Table 6). Under the current classification of dry-mesic oak old growth forest community (type 21), Brushy Mountain falls short on qualifying for old growth under the stand density parameter.

Of the species present, Granite City had white pine (*P. strobus)* in common with the ones listed by the USFS old growth guidance of 1997 (Table 5). The stand density for each dbh class size fell within old growth range. This shows a trend typical of old growth forests, where the forest is uneven aged, and there are fewer trees of a larger dbh in comparison to many trees of a smaller dbh (Table 5).

TABLE 5: Comparison of the old-growth parameters for the conifer-northern hardwoods community (type 2) with the results from Granite City (USFS 1997). There were no PIRU (*P. rubens)* or ACSA (*A. saccharum)* in the plots surveyed.

Old Growth Attribute	Old-Growth Parameters	Granite City Results
Mean age of large trees	TSCA: 147-264 years PIST: 153 to 272 years PIRU: 97 to 129 years ACSA: 114 years	QURU: 145 years QUAL: 127 years
dbh of largest trees	TSCA: 15 to 51 inches, 38.1 cm - 129.54 cm	

	PIST: 28 to 50 inches, 71.12 cm - 127 cm PIRU: 6 to 28 inches, 15.24 cm - 71.12 cm ACSA: 38 inches, 96.52 cm	PIST: 88.1 cm
Stand Density (trees per acre)	dbh $\geq$ 4 inches, 10.16 cm: 91 to 475 $\geq$ 20 inches, 50.8 cm: 3 to 33 $\geq$ 28 inches, 71.12 cm: 0 to 10	$\geq$ 4 inches, 10.16 cm: 153 $\geq$ 20 inches, 50.8 cm: 14 $\geq$ 28 inches, 71.12 cm: 6

TABLE 6. Comparison of old growth parameters for the dry-mesic oak community (type 21) with the results from Brushy Mountain (USFS 1997).

Old Growth Attribute	Old Growth Parameters	Brushy Mountain Results
Age of large trees	QUAL: 245 - 348 years QURU: 240 - 270 years QUVE: 180 - 211 years QUMO: 66- 362 years CATO: 335 years CAGL: 327 years	QUAL: 211 years PRSE: 179 years LITU: 90 years
dbh of large trees	QUAL: 14 - 27 inches, 35.56 cm - 68.58 cm QURU: 22 - 26 inches, 55.88 cm - 66.04 cm QUVE: 18 - 26 inches, 45.72 cm - 66.04 cm QUMO: 14 - 22 inches, 35.56 cm - 55.88 cm *Carya spp*: 14-26 inches, 35.56 cm - 66.04 cm	QUAL: 45 cm QURU: 66.6 cm QUMO: 50.5 cm *Carya* spp: 46.2 cm LITU: 72.1 cm
Stand density (trees per acre)	dbh $\geq$ 4 inches, 10.16 cm: 251 – 401	$\geq$ 4 inches, 10.16 cm: 85

There totaled 51 standing snags in the three Granite City plots in an area of 0.386 ha, and 35 standing snags in the three Brushy Mountain plots in an area of 0.386 ha. The median number of standing snags in Granite City was 13 and in Brushy Mountain, it was 11.

The most abundant live species at the Granite City site were red maple (*A. rubrum)* with a count of 82 individuals, hickory (*Carya* spp.*)* with a count of 52 individuals, white pine with a count of 52 individuals, and chestnut oak (*Q. montana)* with a count of 48 individuals (Figure 1). The most abundant live species at the Brushy Mountain site were Carolina silverbell (*H. carolina)* with a count of 49 individuals, red maple with a count of 42 individuals, white pine with a count of 40 individuals, hickory with a count of 18 individuals, and northern red oak (*Q. rubra)* with a count of 15 individuals (Figure 1). The species that occupied the most basal area at Granite City were in the following order: white pine, chestnut oak, red oak, and then white oak; the species that occupied the most basal area at Brushy Mountain were white pine, red maple, hickory, and chestnut oak (Figure 2).

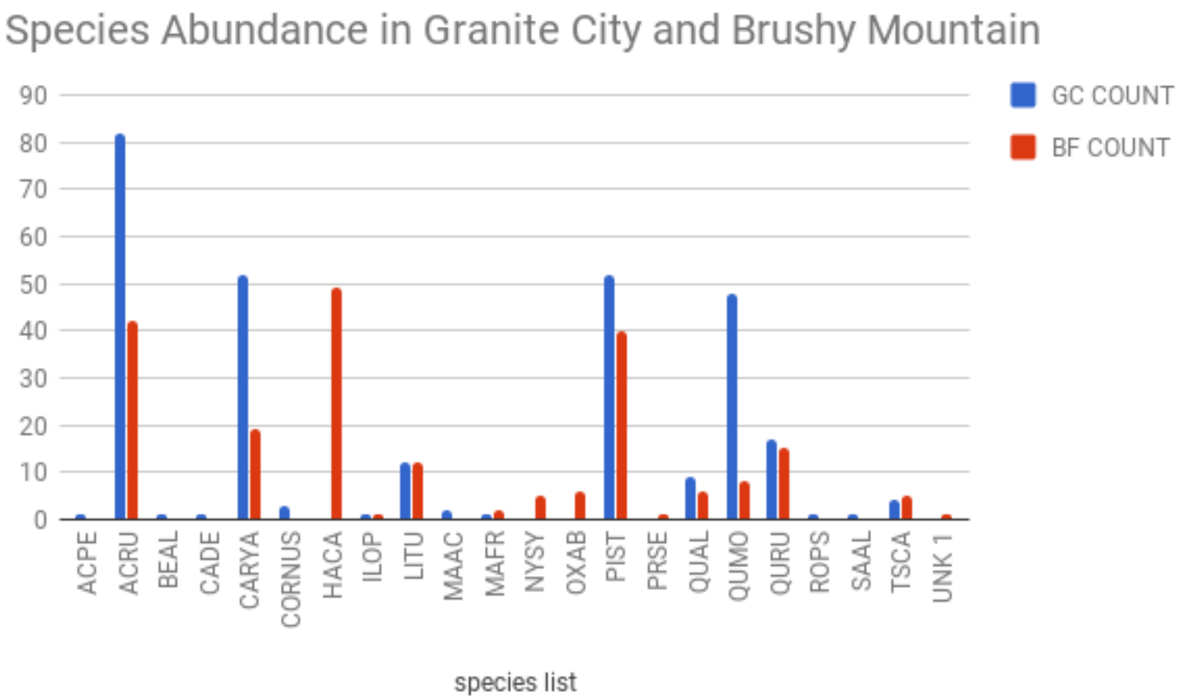

FIG. 1. Species abundance at each site.

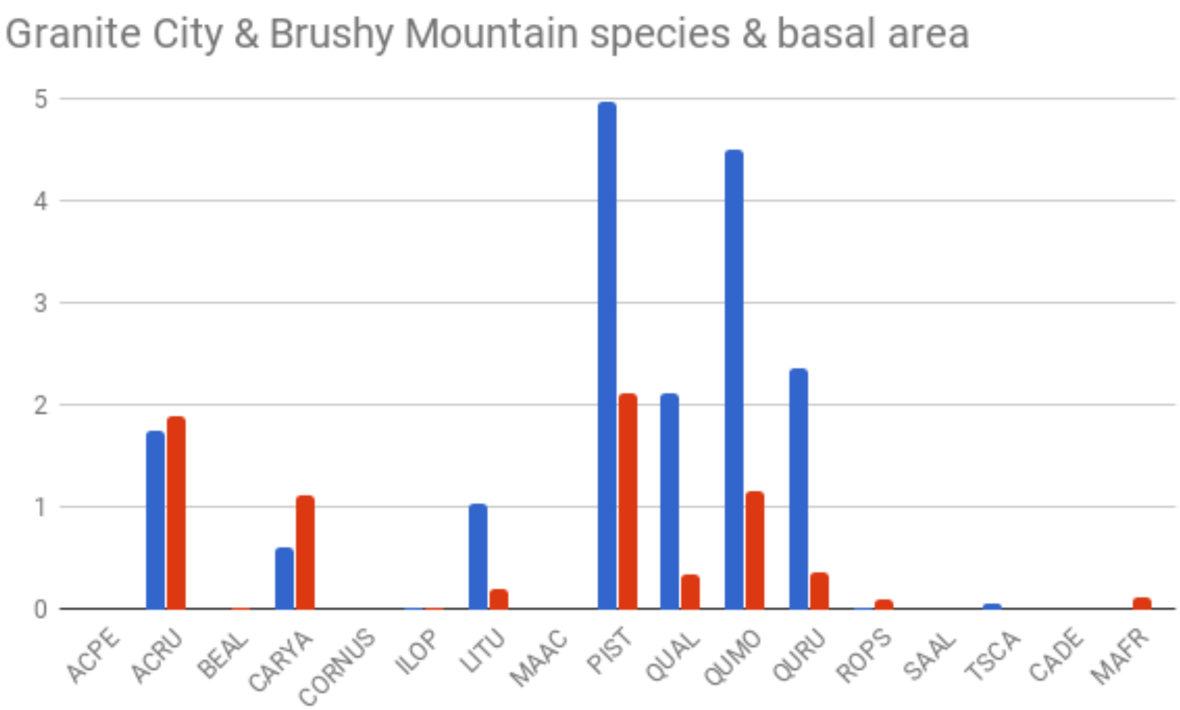

FIG. 2. A graph demonstrating which species occupy the most basal area (m^2) between the sites. Blue bars are Granite City, the red bars are Brushy Mountain. The x-axis shows codes for the species that were comparable between sites.

Due to the species found at the Brushy Mountain site, a western mesophytic forest type could better describe the stand (Figure 1). Western mesophytic forests (type 5) are found on the western side of the southeast, primarily in the southern Appalachians (USFS 1997). The western communities occur on drier sites than mixed mesophytic forests, on a wide range of topographic positions, typically dominated by oaks but including many of the species found in mixed mesophytic forests, the most biologically diverse ecosystems in the US (USFS 1997). The most common species in the mixed forest type are sugar maple (*A. saccharum)*, beech (*F. grandifolia),* eastern hemlock, Carolina silverbell, tulip poplar, red maple, white oak, northern red oak, yellow birch, yellow buckeye (*A. flava),* and basswood (*T. americana)*. However, yellow buckeye is absent from western mesophytic forests, and of the plots sampled, there were no sugar maples, beech, yellow birch, or basswood. The disturbance regime for these forests is often severe but low frequency windstorms, which we saw evidence for this type of disturbance at the second plot within the Brushy Mountain site.

Brushy Mountain had a stand density of 85 trees that were greater than 10.16 cm dbh (4 inches) among all three plots, totaling an area of 0.1593 ha (0.394 acres) (Table 7). The parameters provided by the USFS for this forest community is more difficult to use because there is a maximum value only rather than a range of age/dbh for certain trees, most of which we did not find at the site, those being: basswood, sugar maple, yellow buckeye, and beech (USFS 1997).

TABLE 7. Old growth attributes for mixed mesophytic and western mesophytic old growth forest community (type 5).

Old Growth Attribute	Old Growth Parameters	Brushy Mountain Results
Maximum age of large trees	LITU: 226 years TSCA: 607 years	QUAL: 211 years PRSE: 179 years LITU: 90 years
Maximum of dbh of large trees	LITU: 65 inches, 165.1 cm TSCA: 45 inches, 114.3 cm	LITU: 72.1 cm QURU: 66.6 cm QUAL: 45 cm QUMO: 50.5 cm TSCA: 29 cm *
Stand density (trees per acre)	dbh $\geq$ 4 inches, 10.16 cm: 68-184	$\geq$ 4 inches, 10.16 cm: 85

* largest dbh of a hemlock, but it was a standing dead tree. The largest dbh of a live hemlock was 10.7 cm.

Carbon Assessment

Using the Jenkins et al. (2003) equations, the total biomass of live trees above 2.5 cm dbh surveyed in Granite City was 135,033.73 kg per 0.386 ha, and the total biomass in Brushy Mountain was 56,023.3 kg per 0.159 ha. In contrast, using the more regionally specific Martin et al. (1998) equations, the total biomass of live trees above 2.5 cm dbh surveyed in Granite City was 160,078.62 kg, and the total biomass in Brushy Mountain was 62,039.56 kg per 0.159 ha. The total carbon sequestered in the live trees at Granite City, using the biomass derived from the Martin et al. (1998) equations, was 75,236.95 kg per 0.386 ha, and the total carbon sequestered in the live trees at Brushy Mountain was 29,158.59 kg per 0.159 ha (Table 8).

TABLE 8. The total biomass of live trees above 2.5 cm dbh and standing snags surveyed in Granite City and Brushy Mountain using Jenkins et al. (2003) and Martin et al. (1998) equations, with the total carbon sequestered calculated from the biomass (BM) estimates.

	Granite City		Brushy Mountain	
	Jenkins BM	Martin BM	Jenkins BM	Martin BM
Live trees	135,033.7 kg	160,078.6 kg	56,023.3 kg	62,039.6 kg
Snags and standing dead	n/a	2,102.1 kg	n/a	9,356.6 kg
Both	n/a	162,180.7 kg	n/a	71,396.2 kg
Carbon total	76224.9 kg		33556.2 kg	

The total biomass of standing snags above 2.5 cm dbh surveyed in Granite City was 2,102.1 kg per 0.386 ha and the total biomass in Brushy Mountain was 9,356.6 kg per 0.159 ha. The total carbon sequestered in the standing dead trees and snags in Granite City was 987.99 kg per 0.386 ha and the total carbon sequestered in standing dead trees in Brushy Mountain was 4397.6 kg per 0.159 ha. The total biomass when considering live, standing dead, and snag trees in Granite City was 162,180.7 kg per 0.386 ha, and the total in Brushy Mountain was 71,396.2 kg per 0.159 ha. The total carbon in all pools (live, standing dead, and snag trees) in Granite City was 76,224.94 kg per 0.386 ha, and the total carbon in all pools in Brushy Mountain was 33,556.19 kg per 0.159 ha.

Regardless of a specific site, the trend in the live tree biomass is correlated positively with increasing dbh in a reverse-J shape (Figure 3). This graph shape of biomass versus dbh is characteristic of old growth forests (Scheff 2012).

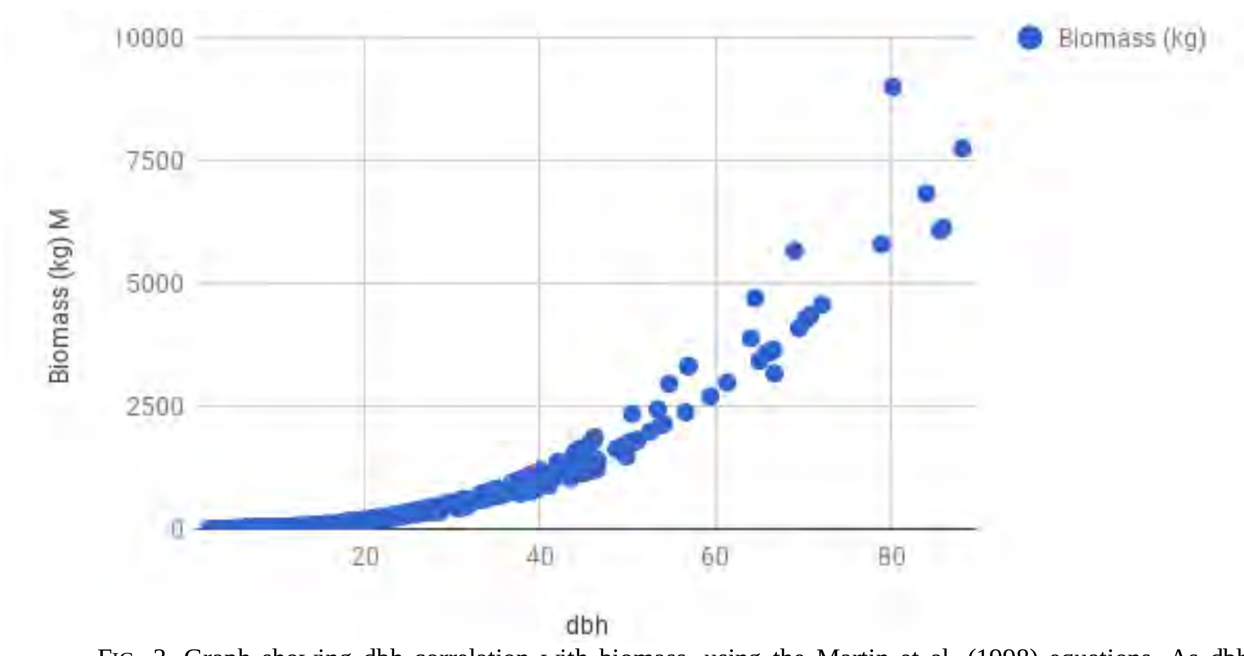

FIG. 3. Graph showing dbh correlation with biomass, using the Martin et al. (1998) equations. As dbh increases, so does biomass.

DISCUSSION

The Granite City plots qualified as old growth based on the USFS basic guidelines for old growth forest type conifer-northern hardwoods following these parameters: basal area, median number of snags, dbh of largest tree, and stand density (Table 1 and Table 5). The Granite City site has not been designated as old growth by the USFS, and that being said, the other old growth parameters should be measured at Granite City so this forest stand can be officially classified as old growth.

The minimum stand age in conifer-northern hardwoods is at least 140 years old (Table 1). Most of the trees cored in Granite City were at least 110 years old, if not older. The majority of trees cored at Granite City did not sample the full tree to center (reach the pith) because the increment borer was not long enough. A study focusing on aging these stands is necessary before continuing the timber sale to ensure that potential old growth habitat can be maintained and conserved. The minimum stand age in dry-mesic oak forests is at least 130 years old (Table 1). At Brushy Mountain, one of the white oak trees that were cored was 211 years old and the black cherry tree was 179 years old (Table 6).

According to the guidelines for old-growth of the dry-mesic oak forest type, Brushy Mountain would qualify as old growth based on dbh of largest trees (Table 6) that included QUAL (45 cm), QURU (66.6 cm), QUMO (50.5 cm) and *Carya* spp (46.2 cm). The plots surveyed at Brushy Mountain do not qualify it as old growth based on basal area for this forest type. Because only two-fifths of an acre were surveyed however, a survey of an entire acre (0.4 ha) on the Brushy Mountain stand should qualify assuming our plots are representative of the stand. The median number of snags surveyed in the Brushy Mountain stand was 11, so extrapolated to an entire acre (27.5), Brushy Mountain would also qualify as old growth based on 26 to 36 median number of snags (Table 1).

Brushy Mountain does not qualify for old growth on the stand density parameter under the dry-mesic oak old-growth guidance community type (USFS 1997). However, due to the species composition of the stand (Table 7), and most specifically, the sheer abundance of *Halesia carolina* (Figure 1), the forest type would be better classified as a Western Mesophytic old growth community (USFS 1997). If classified as such, this stand would qualify under all of the other old growth parameters, including stand density. White and Lloyd (1998) describe how the southern Appalachian region is composed of oak-chestnut on the slopes, mixed-mesophytic in the coves, and pine or oak-pine on the dry slopes and ridges. With that in mind, it is difficult to absolutely and accurately define a forest stand, which can comprise all of these separate topographies. However, the difference between the old growth classification of a mixed-mesophytic and western mesophytic forest versus a dry-mesic oak forest is the amount of protection that forest type receives, which historically has been dry-mesic oak forests are the least protected forest type in this region (Sierra Club 2005).

The total biomass in the Granite City site was greater than the Brushy Mountain site and included a greater acreage than Brushy Mountain. Cumulatively between the two, there was 233,576.9 kg in an area that is just over half a hectare. Assuming these plots are representative of the entire stand, within just these two sites there were approximately 18 hectares-- reflecting how much biomass is stored in the rest of the stand. The total amount of carbon sequestered cumulatively between the two sites, including the live, standing dead and snag trees was 109,781.14 kg in an area greater than half a hectare.

Additional studies would have to be done to calculate the carbon threshold limit that these forests can store to know how much more potential these forests have to store carbon (Hudiburg et al. 2009). If these forests are logged during the Southside Timber Sale, then the carbon storage capacity of these forests will not reach this current of carbon storage for another 200 years (Harmon et al. 1990, Noormets et al. 2015). These calculations do not represent the shrubby or dead aboveground biomass, nor any of the belowground biomass and carbon that would be disturbed or lost due to logging activity. In addition, logging these areas would also remove standing dead trees and snags which are important habitat for a variety of species (DNR South Carolina 2014).

Due to time constraint, I was only able to survey six plots total, with three plots at each site. Additional plots would have made this study more representative. Unfortunately, a request for FIA data for comparable elevation and forest types in Western North Carolina was not fulfilled within the two month period before this paper was written. We identified trees as accurately in the field as possible, but due to the start date on this project, species on the last three plots were more difficult to identify due to seasonal leaf drop; therefore, at least one tree was not identified. A more random plot selection would have been ideal to eliminate the possibility of prioritization of specific areas and comparison to a larger data set would make the study more robust. I was not able to measure all of the USFS's old growth parameters, and the ones I omitted were the volume of downed woody debris, number of canopy layers, and percentage of canopy gaps. One should not officially classify these stands as old growth until these last three parameters have been measured.

Estimating the carbon and the biomass has its own error built in since these equations are general and every tree, especially as it reaches the upper limits of its species' dbh, is unique and it becomes more difficult to accurately estimate biomass with general allometric equations.

In conclusion, the most important findings of this research were that Granite City does qualify as an old-growth stand under the parameters measured, Brushy Mountain may be better described as a western mesophytic forest type which would then also qualify it as old growth under the parameters measured, and that the total carbon sequestered in the standing live and dead trees that were surveyed over six plots is a total of 109,781.14 kg. Considering that both of these stands, according to the 1997 old growth guidance for region 8, fall under the old-growth classification under the parameters measured, I propose that the USFS do a standwide survey for at least these two stands that are posed to be cut in the Southside Timber Sale.

Climate change threatens southeastern forests especially due to species migration, exotic species invasion, and the transformation of natural disturbance regimes (Bragg and Shelton 2014). Not only is there value in conserving old growth for ecosystem services such as carbon sequestration, but there is also value in conserving this rare habitat for research since there are so few old growth stands remaining in the southeast. My recommendation is that these sites be conserved as intact old-growth habitat and for the purpose of sequestering as much carbon as possible, especially in the face of climate change and increasing greenhouse gas emissions.

Acknowledgements

I would like to first thank Dr. Costa and Dr. Workman for their tremendous support in this research project. I am also grateful for the guidance, assistance, and impartment of knowledge by none other than Buzz Williams. Thank you to Eric Schwartz for accompanying and helping me with gathering data in the field. I also must thank Dr. Chris Oishi for all of his help with developing the research methods and focus, as well as help with analyzing data. Thank you to Nicole Hayler for sharing various documents with me which were essential to this research. I would also like to thank Michelle Ruigrok for helping me with the formatting of this paper. Finally, I would like to thank all of the people who helped me in any way, whether that simply be reading and editing my paper or helping me get an increment borer unstuck from a massive chestnut oak: Winter Gary, Jackson Denton, Aidan Buie, Ishika Kumbhakar, and Anna Kelly.

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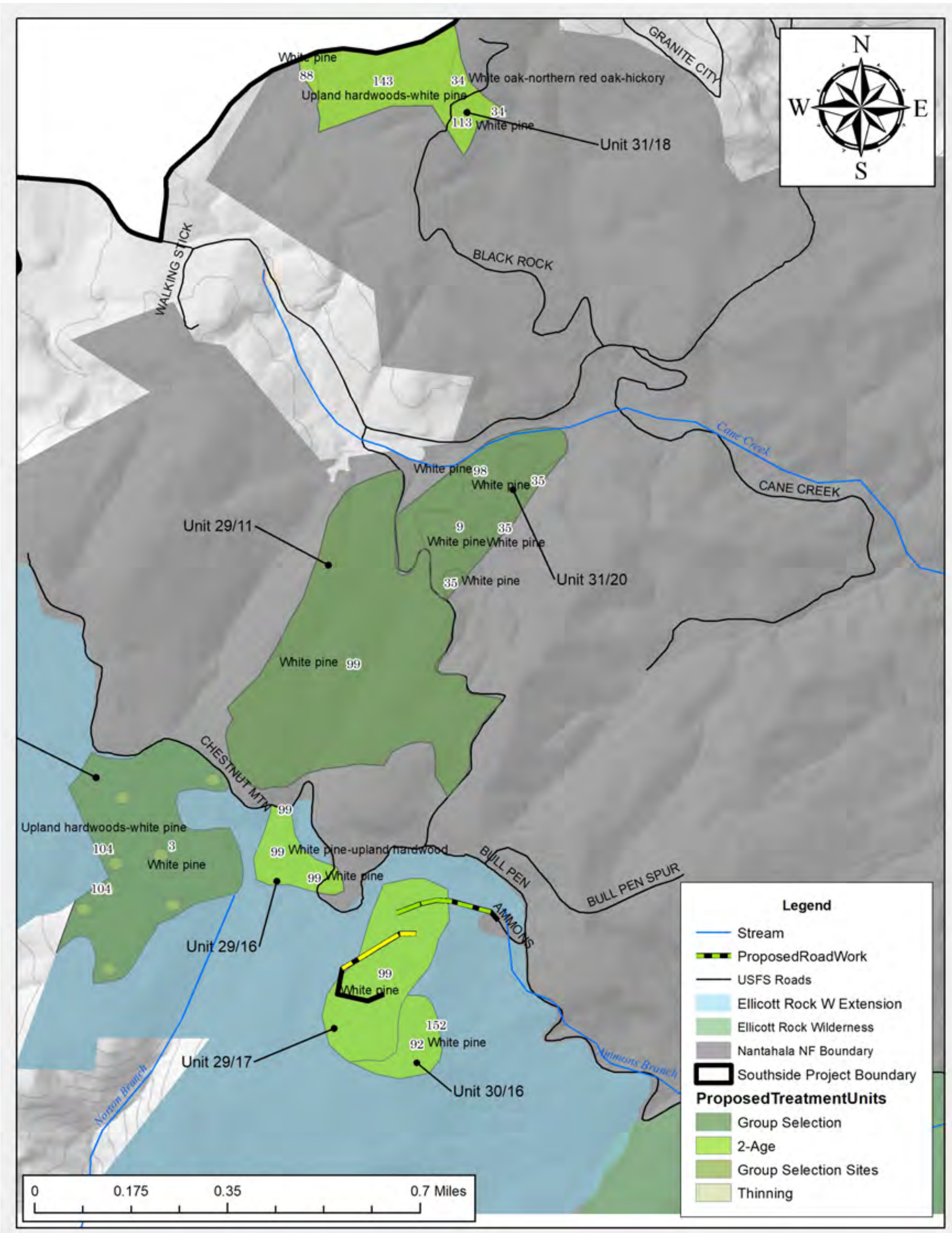
APPENDIX A.



Appendix B.

Evaluation of Southside 35-41 old growth characteristics
Reference: p.60 R8 Old Growth Guide
(Dry-Mesic Oak Forest - Old Growth (OG) Community Type 21
Sothern Appalachians Region)

SouthSide 35-41 Stand Characteristics:
Forest type (FIA EV code): chestnut-black-scarlet oak (515)
WP SI: 98 (measured) WO SI: 80 (Doolittle chart comparison)
110 years old in 2017 (age year 1907)
Stand is primarily on a ridge, with large rhodo. thickets on the downslope and scattered

Live Tree Size Class	TPA	BA/ac	% BA
0-3.99" DBH	0	0	0%
4-11.99" DBH	97	35	31%
≥ 12" DBH	46	78	69%
Total	143	113	

Species Composition	TPA	BA/ac	% BA
CO > WO > HIC > BO > SO > NRO	50	56	49%
BG > SW > RM > BL	47	30	27%
WP	41	20	18%
Y-P	5	8	7%
Total	143	113	

OG Parameter	OG Type 21	Southside 35-41	Meets OG Criteria?
Age of large trees (years)			
WO	245-348	increment borer issues	
NRO	240-270	"	
BO	180-211	"	
CO	66-362	"	
HI	327-335	"	
Overall average	259	stand is 110 year old	no
DBH of large trees (inches)			
WO	14-27	22.3	yes
NRO	22-26	21.4	no, but close
BO	18-26	18.4	yes
CO	14-22	20.3	yes
HIC	14-26	no HIC > 11.99"	no
Overall for trees 12.0" and larger	21	17.5	no, but close
Stand Density - TPA			
Trees dbh ≥ 4 inches	251-401	143	no
Stand Basal Area			
Trees dbh ≥ 4 inches	73-115	112.5	yes
# Snags per Acre			
Trees dbh ≥ 4 inches	avg of 26-36	30	yes
Volume (ft3) Downed logs per Acre			
Logs ≥ 8 inches	403-1438	256*	no
Percent canopy in gaps			
%	1-13	90% canopy closure**	unknown

*Volume was not measure for downed wood but it was measured for standing snags and is used as a surrogate.
One plot has one piece of downed wood ≥ 8 inches
** Average canopy cover across plots, "gappiness" was not estimated

RESTORING SHEETFLOW TO KANUGA BOG, A SOUTHERN APPALACHIAN FEN

WINTER GARY

Abstract. Southern Appalachian Mountain wetlands are unique habitats that require management to counter negative anthropogenic impacts such as urban development, wetland drainage and conversion, and hydrological alteration. This study followed one aspect of the management plan for Kanuga Bog in Hendersonville, NC. We attempted to decrease the channelization of the main stream in the bog, which was altered by a culvert and enhanced by beaver activity, and to increase the overall sheet flow in the historically wet northeastern portion of the bog. This was done to benefit two endangered species that inhabit the bog: mountain sweet pitcher plants, *Sarracenia rubra spp. jonesii*, and bog turtles, *Glyptemys muhlenbergii*. We installed three experimental water/ sediment control models throughout the stream and placed four wells in the eastern section of the bog to record groundwater level fluctuations each week. Biotic indices were calculated and water quality measurements were taken prior to and after the water/ sediment control models were installed. These measures were used to determine if any water quality changes occurred, and no major concerns were detected. After the models were placed, sheet flow visibly increased in the eastern side of the bog, and the well data showed that the groundwater levels also rose. Rain gauge data supported that the increase was due to our models and not precipitation.

Key words: American beavers; bog turtles; channelization; Glyptemys muhlenbergii; *groundwater; fens; Kanuga Bog; mountain sweet pitcher plants; sheet flow;* Sarracenia rubra spp. jonesii; *Southern Appalachian Mountain bogs; wetlands*

INTRODUCTION

Southern Appalachian Mountain wetlands, such as bogs and fens, are increasingly rare habitats found scattered within this geographic region. The valley floors of the region are believed to have once contained an estimated 2,023 hectares of boggy wetlands, but today approximately one-tenth of the scarce and threatened habitat remains (Chapman 2017). These wetlands are considered to be biological hotspots and are home to several rare and endangered species, such as mountain sweet pitcher plants, *Sarracenia rubra spp. jonesii*, and bog turtles, *Glyptemys muhlenbergii* (Mountain Bogs National Wildlife Refuge 2016). Anthropogenic impacts are the most extensive threats to these wetlands through practices such as drainage, conversion of the habitats to agriculture fields and pastures, development, and hydrological modification (Schafale and Weakley 1990).

The U.S. Fish and Wildlife Service (FWS) established the Mountain Bogs National Wildlife Refuge in 2015 in an effort to protect and conserve these unique wetland habitats and the endangered wildlife within them. Unlike most refuges, which are focused in one location, this refuge consists of 30 small, isolated wetlands spread across western North Carolina and eastern Tennessee. It has been estimated that it could take decades, and tens of millions of dollars, for FWS to acquire all the land, therefore, they rely greatly on private landowners to help them conserve wetland habitats which is their ultimate goal (Chapman 2017).

Many of the organizations that manage the Mountain Bogs National Wildlife Refuge, such as FWS, MoutainTrue, and North Carolina National Heritage Program (NCNHP), collaborate with Kanuga Conferences Center to manage Kanuga Bog, a 0.81 ha southern Appalachian Mountain fen located in Hendersonville, North Carolina. The classification of mountain wetlands is somewhat tentative due to their variability in vegetation and gaps in knowledge about their hydrology and nutrient dynamics (Schafale and Weakley 1990). Fens are defined as wetlands fed by groundwater with both surface and subsurface outflow, while bogs are wetlands fed by precipitation with primarily groundwater outflow (Wilcox 2017).

In the southern Appalachian region, bogs and fens are often collectively known as bogs. This is because the fens have bog-like chemistry which causes the ambiguity in the use of the term. A typical fen has a pH range between four and eight, but in this region, fens have a lower pH like bogs. The conductivity of southern Appalachian fens is also lower than the usual measure of greater than of 100 μScm^{-1}, and they have different ions associated with them than the typical fen. Bogs have lower pH values than fens because they are fed by precipitation, which makes the water more dilute. In this region, the location where precipitation meets a point in the watershed and percolates into the groundwater is so close to the location where it eventually flows into the fen that the water is relatively unchanged. In other regions, it can take months or years for the water to reach a fen through groundwater, so it changes substantially. This unique characteristic of this mountainous region gives the fens bog-like chemistry and, therefore, bog-like vegetation (Wilcox 2017). Fens in this region can also be characterized by their mucky, relatively acidic soils due to the acidity of the prevailing southern Appalachian substrates, and environments dominated by shrub and herbaceous vegetation underlain by sphagnum moss mats (Schafale and Weakley 1990).

Kanuga Bog is fed primarily by groundwater but is also fed by precipitation, runoff, and a moderate-gradient first-order stream, known as High Rock Creek, originating in a forested area about 762 m northwest of the bog (Casebeer and Caldwell 2012). This creek flows south underneath Kanuga Chapel Drive, which is the main road through Kanuga Conferences Center property, and into the center of the bog through a culvert with a 0.91 m (three ft) diameter. The northwest corner of the bog is fed by a natural spring that slowly flows south 35.36 m and joins the main stream (fig. 1). The northeast corner does not have a natural spring feeding it. In the past, it had water flowing through it from the main stream, however, the culvert concentrated the flow of the stream to the center of the bog, causing major channelization which eliminated sheet flow on the eastern side. The resulting enlargement and channelization of the stream lowered the overall water table in the bog (Casebeer and Caldwell 2012). The row of large pine trees along the outer edge of northeast section of the bog also contributed to the decrease in the water table in the eastern portion of the bog. A lowered water table makes the area suitable for encroachment by other woody vegetation, which leads to an increase evapotranspiration and further lowering of the water table (Casebeer and Caldwell 2012). The channelization has caused the bog to become significantly less bog-like and more like a closed woodland area with a stream. This has ultimately caused stress on the wetland flora and fauna.

These conditions within the bog threaten two endangered species that inhabit it: mountain sweet pitcher plants and bog turtles. Mountain sweet pitcher plants, only found in a few counties in southwestern North Carolina and upstate South Carolina, are known for their carnivorous diet and prefer wet conditions, ample light, and nutrient poor habitats (Gupton 2015). When in dry, shady conditions, they are inhibited by the high photosynthetic costs of maintaining complex insect trapping mechanisms and cannot outcompete habitat generalists. Dry, closed canopy conditions cause wilted leaves, less flowering, and even mortality (Casebeer and Caldwell 2012). Bog turtles, the smallest turtles in North America and very rare to find, were encountered in Kanuga Bog only in 1983 and again in 2012, so there is little information about their populations here. The turtles are known to prefer spring-fed wetlands with saturated, nutrient poor soils, and open canopy, and prefer moderate amounts of running water rather than deep standing water habitats (Casebeer and Caldwell 2012).

The bog, and the wildlife it contains, requires regular management to control the pressure from several threats it currently faces. Beaver activity, sedimentation, encroaching woody vegetation, and invasive species are some of the most substantial threats. The American beaver, *Castor canadensis,* is a key species for wetlands. Their dams facilitate the establishment of riparian vegetation by increasing the extent and duration of soil moisture, providing sediment for seedling establishment, and reducing woody vegetation by raising the water table to levels they cannot survive (Boyles and Owens 2007). Relatively recent beaver activity in this bog has had damaging impacts on the pitcher plant and bog turtle habitats. In 2007, a FWS professional reported a beaver dam measuring 0.91 m high in the southwest corner of the bog, causing water levels to rise a few centimeters over the soil in the pitcher plant area. This threatened the existing pitcher plant populations because they cannot tolerate prolonged inundation (Casebeer and Caldwell 2012). This was also unfavorable for bog turtles, because they dislike deep standing water.

In 2014, Kanuga had a beaver population of 26 individuals. Clemson beaver pond levelers were placed in the bog in 2010 to control the water height impounded by their dams. These are devices made from PVC pipe that can be placed through a beaver dam to allow water to flow through them and help control flooding (MDNR 2001). Beavers used the main stream as a game and social path causing it to become increasingly deep. They also cut the trees along the stream causing the water in peripheral streams to move freely toward the deeper main stream with fewer root mats in its path, therefore, contributing to the overall channelization (DeWitt 2017). In 2015, Kanuga had many of the beavers captured and removed and then began breaking up the beaver dams on a daily basis causing the remaining population to relocate off property. The portion of the bog that was once causing the upper pitcher plant area to flood due to the beaver impoundment, is now drying out due to the channelization and lack of surface flow reaching this portion of the bog.

This study focused on the section of the bog known as Management Zone 3. It was less than 0.40 ha (one-acre) and located north of the upper boardwalk that bisects the bog east to west (fig. 1). The boardwalk contained a gazebo area that extended south and possessed a view of the pitcher plants. The goal of this study was to decrease the channelization of the main inflow stream in Kanuga Bog, which was artificially altered by a culvert and amplified by beaver activity, and to increase the sheet flow overall. We did this in order to enhance the environment for the rare mountain sweet pitcher plants and bog turtles.

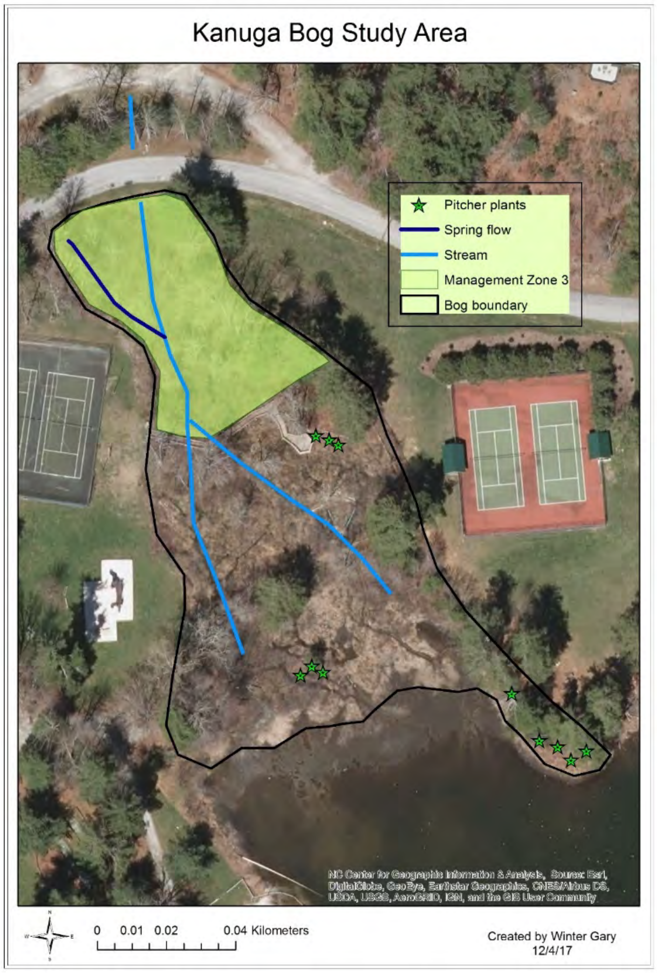


Fig. 1. Kanuga Bog with Management Zone 3 depicted. Created in ArcMap 10.5 (NC OneMap 2016).

METHODS

Baseline Health Measurements

In order to obtain a baseline health of the stream in Kanuga Bog, we calculated a biotic index (BI) for the primary inflow stream and two main outflow streams. We collected macroinvertebrates using a one-minute kick-net sample at three sections along the inflow stream and two sections along each of the two outflow streams. We identified macroinvertebrates to the order level using a key (UWEX n.d.) and calculated scores for each of the streams from zero, indicating more pristine conditions, to ten, indicating stress. This allowed us to evaluate the water quality and determine the amount of organic pollution or other factors likely within each stream (NCDWQ 2016). The scores were calculated by multiplying the abundance value (n) of each order (i) by the tolerance value (T) for that order (i), summing the products, and dividing by the total arthropods in the sample (N).

$$BI = \frac{\Sigma(n_i)(T_i)}{N}$$

We conducted baseline health measurements of the water in the bog prior to and after implementing our water and sediment control models. The measurements included chemical analysis for wells 1, 2s, 2d, 3s, 3d, 4s, 4d, and 5 (fig. 5) and water samples from the stream inflow, both outflows, and above the boardwalk in the main stream. We measured the water temperature, pH, and conductivity at each of these locations, measured the water depths within each well, and measured turbidity at the four sites that were not wells. Turbidity was also measured in the stream right beside well 1.

We also performed a habitat assessment on the stream prior to any water model installations using the Southern Appalachian Stream Visual Assessment Protocol (saSVAP) customized for the southern Appalachian region by the Land Trust for the Little Tennessee (LTLT) with the University of Georgia as adapted from the USDA's stream visual assessment protocol (2009). The saSVAP assigns a numerical score to stream health based on the bank condition, riparian buffer quantity and quality, canopy cover, riffle embeddedness, trash and garbage, non-trash/nutrient pollution, livestock, pools, available habitat/cover, and barriers to fish movement. The scores range from one, indicating very poor stream health, to four, indicating excellent stream health.

Water/ Sediment Control Models

The first major water/ sediment control model was a drop structure and two modified step-pools located 0.61 m from the culvert where the main stream flows into the bog. The step-pools expand 3.51 m to the east and 4.27 m south into the section of the bog that was once well saturated but currently lacks sheet flow due to channelization of the main stream (fig. 2). Drop structures are low-elevation formations that span the entire width of the channel, creating an abrupt drop in channel bed and water surface elevation in a downstream direction and help redistribute and dissipate energy (Saldi-Caromile et al. 2004). Step-pools are rock grade control structures in streams that recreate natural step-pool morphology. They are constructed of large rocks with the pools filled with smaller rock material (DCRDSWC 2004). We implemented this model to redirect the stream to the eastern portion of the bog so that sheet flow was increased and channelization of the main stream was decreased. The step-pool design was meant to slow water velocity and allow sediment to drop and settle while allowing water to continue to flow over the bog surface (Monteith and Janes 2013).

We constructed the drop structure and step-pools using stones recycled from a building on Kanuga's property. Each stone was about 0.30 m (one ft) in length or less. We designed the drop structure with a 0.76 m length, while the two step-pools were each 1.83 m wide and 0.91 m long with a 0.76 m left arm that reaches up to the previous structure (fig. 2). We built the structures with a 0.30 m overlap. The pools were sealed with impermeable black plastic to ensure no leakage or flow underneath the structures (DCRDSWC 2004). We used smaller stones to fill the step-pools in so that the black plastic was covered to prevent it from accumulating heat from the sun and causing the water to increase in temperature. We also predicted sediment accumulation would help avoid this.

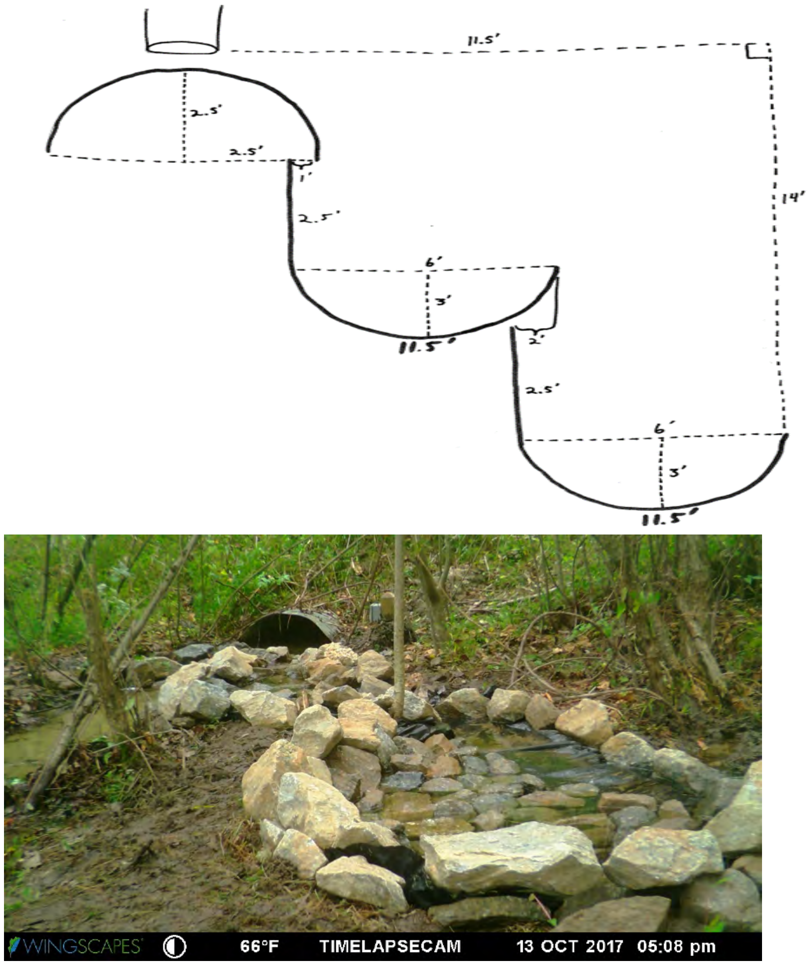


FIG. 2. (a) Above. The original drop structure and modified step-pools design from an aerial view. The culvert and drop structure are shown at the top, and the step-pools expand 3.51 m to the east and 4.27 m south into the bog. (1 ft= 0.3 m) (b) Below. Drop structure and step-pools after installation was completed on October 13, 2017.

Further down the stream below the installed drop structure and modified step-pools, we implemented three other water/ sediment control models to increase sheet flow in the bog and decrease channelization of the main stream. We surveyed the bog stream to determine the slope between the check points where we planned to implement our models. We placed the more obtrusive and artificial models upstream and the more natural models downstream by the boardwalk to avoid negative aesthetic consequences along the boardwalk. The models were intended to act much like beaver dams in regards to redirecting water and capturing sediment since the beavers were no longer abundant in the bog.

The more natural appearing water/ sediment control model was placed along the main stream closest to the boardwalk. We created a debris check dam, constructed from natural debris in the bog, such as fallen trees, placed it in the stream just above the boardwalk, and secured it in place using rebar (fig. 3). Woody debris alters surface flow by allowing some water to flow straight, but encourages water to flow to the right of the stream toward the pitcher plants (Bridle et al. 2000). We used leaf debris that was already in the stream to fill gaps between the trees in the natural debris dam. The purpose of this model was to allow water to continue to flow in the stream while simultaneously causing the water upstream to pool and flow over the stream bank towards the eastern section of the bog.

FIG. 3. The natural debris check dam installed in the stream and secured with rebar on October 13, 2017.

The last check dam we placed was a burlap check dam. We constructed it using 13 rolls of 100% natural all-purpose burlap. We unrolled each one and rolled them together loosely into one large roll. The rolls were 0.91 m x 7.62 m individually, but after combing them, were 0.91 m long with a 0.36 m diameter. We placed the burlap roll across the stream and secured it in place using rebar (fig. 4). We installed it in the stream below the drop structure and modified step-pools and upstream of the natural debris dam. It was oriented so water still flowed through and under it a bit in the main stream bed, but was also intended to cause the water upstream to pool and flow over the stream bank towards the old, dried stream bed in the eastern section of the bog. The burlap is biodegradable, so we hope it will encourage sheet flow permanently such that the water flow continues to flow away from the main stream and towards the dried stream bed even after it biodegrades.

FIG. 4. (a) Left. The burlap check dam with a 0.36 m diameter prior to installation. (b) Right. The burlap check dam after installation on October 27, 2017.

Vegetative filter strips are meant to help filter any pollutants in the stream water, support sheet flow, decrease sedimentation, and prevent erosion along stream banks by decreasing flow velocity and spreading the flow across a wide area (NRCS 2014). Our last model was a filter strip of wetland species planted on the eastern portion of the bog. In mid-September, we collected seeds from three species: *Carex lurida*, *Sparganium americanum*, and *Impatiens capensis.* These plants were chosen because they were wetland species already abundant in the bog. We collected them in numerous sections of the bog to increase genetic diversity. We dried the seeds and then planted them in trays to germinate. We originally planned to plant strips along each side of the main stream, however, we decided it would be better to plant around the newly inundated areas caused by our water/ sediment control models. During the building of our previous models, much of the soil in the eastern section of the bog became aerated as we continued to walk through it, and we had to remove some woody vegetation to make the area more accommodating for bog turtles. We decided it would be best to plant the wetland species around these inundated areas to introduce root systems that will help continue to encourage increased sheet flow and to filter pollutants such as sediment from the water coming from the culvert and runoff from the road. They have not yet been planted, because we thought it would be best to wait for more ideal planting conditions after winter when the temperature begins to rise and increase their chances of survival.

Water Table Measurements

We inserted three temporary wells along the eastern portion of the bog on October 5th to facilitate weekly groundwater measurements before and after model installations to record changes in the water table level (fig. 5). Permanent wells 2s and 2d were also measured weekly since they were located in the eastern section. After the models were installed and we knew the direction the water was flowing, we placed a fourth temporary well in its path. The temporary wells had to be surveyed in relation to each other and to existing wells, because the well pipes are different lengths and are at varying depths. We used survey equipment to create an arbitrary line that was called Station A. We assigned the line an arbitrary value of 3.048 m (10 ft) in height to calculate the heights of the wells relative to it. All the wells were measured from the top of their casing to this line. Their measurements were then each subtracted from 3.048 m and these values were used as the well heights. Each week when the groundwater levels were measured in each well, the water depth measurements were subtracted from the well height values to see how the water depths changed over time. We collected data from the rain gauge in Kanuga Bog to determine if groundwater changes were caused by our models or by precipitation inputs.

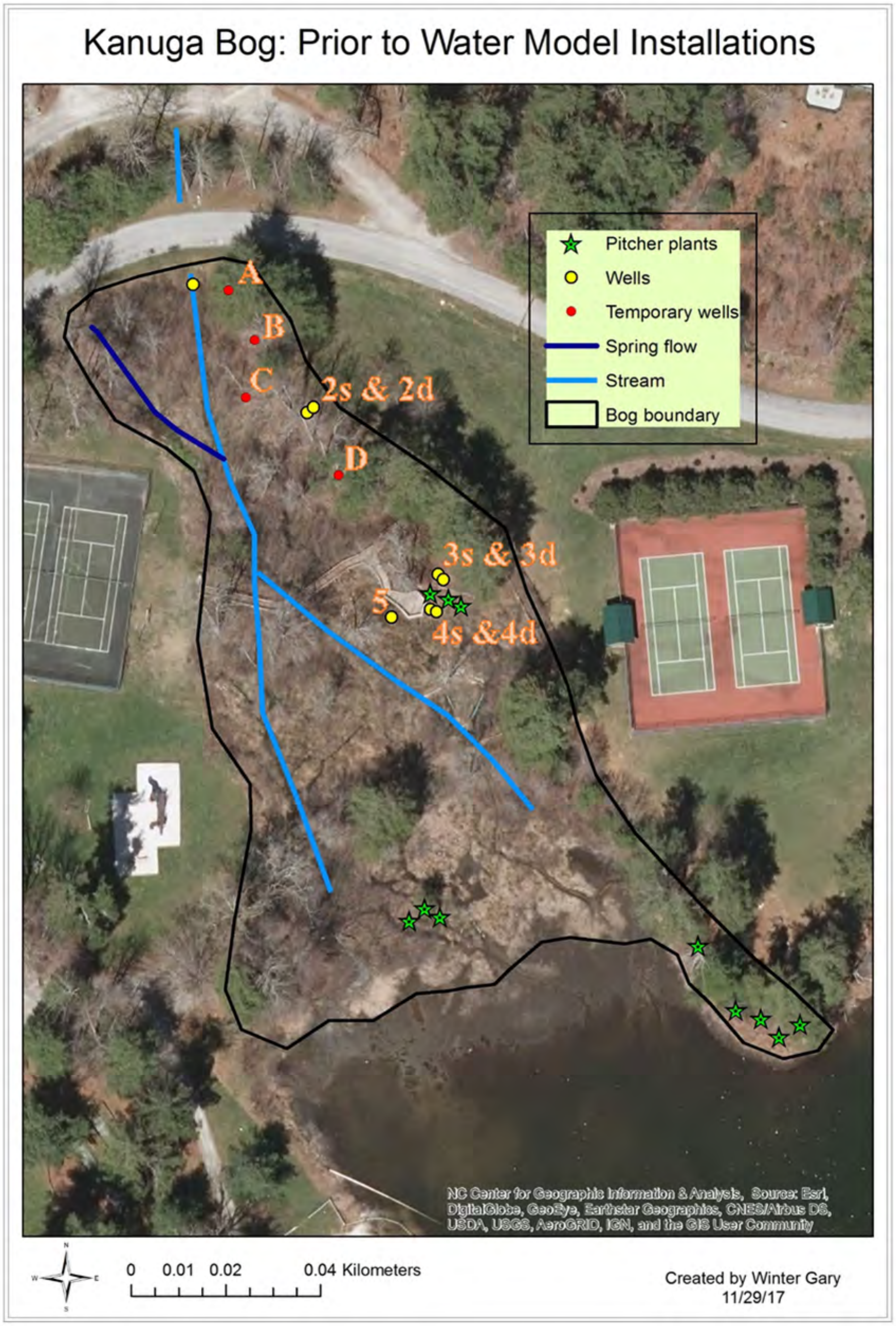


FIG. 5. Kanuga Bog stream flow prior to water/ sediment model installations in relation to pitcher plant populations and installed wells. Temporary wells A, B, C, and D are labeled. Created in ArcMap 10.5 (NC OneMap 2016).

RESULTS

Baseline Health Measurements

Our biotic index scores had little variation between the pre- and post-model installations. The inflow score increased from 3.58, indicating excellent water quality, to 3.97, indicating good water quality. The outflow scores both decreased slightly showing improved water quality.

Outflow A, the southwestern stream, had an initial score of 7.42 which lowered to 6.60, both qualifying the stream as poor water quality. Outflow B, the southeastern stream, had an initial score of 4.98 that decreased to 3.89. The saVAP score for the stream in Kanuga Bog was 3.1 classifying the stream as fair quality.

The chemical analysis we performed at the inflow, both outflows, above the boardwalk, and in the wells 1, 2s, 2d, 3s, 3d, 4s, 4d, and 5 varied between sites prior to and after our model installations (table 1). The temperatures changed naturally due to the weather cooling. Prior to model installations, the water temperature ranged from 13.4 ºC to 17.7 ºC and after the installations, it ranged from 11.2 ºC to 15.1 ºC. The pH values ranged from 5.4 to 7.1 and changed very little to a range of 5.4 to 7.4 after the models were installed. The more basic pH values were from collection sites within the main stream, while the well pH levels in the wells were more acidic. The conductivity originally ranged from 18.3 μScm^{-1} to 49.3 μScm^{-1} and decreased to 0.13 μScm^{-1} to 46.3 μScm^{-1} once the models were installed. The average conductivity prior to our changes was 27.6 μScm^{-1}. It decreased to 21.6 μScm^{-1} afterwards. Outflow A was an outlier each time and had the highest conductivity with its measure being about 20 μScm^{-1} higher than the average. Conductivity at well 4d, the deep well located by the pitcher plants, dropped 20.47 μScm^{-1} from its value prior to the model installations.

Turbidity measurements ranged from 1.28 NTU to 15.0 NTU originally and changed to 1.10 NTU to 25.5 NTU after model installations (table 1). Outflow A was an outlier once again and had the highest turbidity measurements each time. Its turbidity rose 10.5 NTU by the time we recorded our post-model installation measurements. Each permanent well saw an increase in groundwater levels after the water/sediment models were installed. The groundwater increased by an average of 0.076 m. Well 3d, a deep groundwater well located northeast of the pitcher plants, saw the greatest increase, 0.122 m while well 5, located west of the gazebo area on the boardwalk, saw the lowest increase, 0.030 m.

TABLE 1. Pre- and post-model installation chemical analysis results for our wells and stream sample sites. Measures included groundwater levels for permanent wells from the same days.

Well/ Sample Area	pH		Turbidity (NTU)		Conductivity (μScm^{-1})		Temperature (C)		Groundwater Levels (m)	
	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
1	7.1*	6.8	1.28*	1.46	20.6	18.5	13.4*	11.3	1.582	1.625
2s	6.0	6.0	~	~	36.4	34.8	16.8	12.5	1.067	1.116
2d	5.9	6.0	~	~	22.5	17.3	16.9	12.8	1.070	1.131
3s	5.7	6.2	~	~	39.1	21.4	17.7*	14.4	0.280	0.390
3d	5.4*	5.6	~	~	18.3*	19.0	17.7*	15.1*	0.265	0.387
4s	5.5	6.1	~	~	26.6	21.3	16.6	11.4	0.198	0.308
4d	5.8	5.4*	~	~	20.6	0.13*	17.0	13.1	0.189	0.274
5s	5.7	6.0	~	~	36.3	24.2	17.1	12.1	0.299	0.329
inflow	6.7	7.4*	2.04	1.10*	19.8	17.9	14.6	11.4	~	~
boardwalk	6.6	7.3	2.51	2.09	20.6	19.1	15.1	11.6	~	~
outflow A	5.8	6.4	15.00*	25.50*	49.3*	46.3*	16.0	11.2*	~	~
outflow B	6.5	7.0	3.86	2.55	20.8	19.5	15.2	11.4	~	~

*Highest and lowest values for each set of parameters measured.
~ Measure not applicable to sample site.

Increased sheet flow occurred in the eastern portion of the bog as soon as the drop structure and modified step-pools were installed, because it physically separated the main channel into three flow directions: towards the pools and into the eastern section, straight into the main channel, and to the western section where the water usually flowed when there were storm events. The water was slowed and split by the drop structure, and the portion that went east slowed further into the first and second modified step-pools. Water then flowed out causing a visible increase in overall sheet flow (fig. 6). Within the first week, sediment filled in around the stones in the step-pools, showing that it was allowing the water to pool enough for suspended solids to settle out. The water flowed slowly as it made its way south through the bog along the eastern side (fig. 9).

FIG. 6. New sheet flow in northeastern corner, by well A.

The burlap check dam also caused immediate physical changes in sheet flow. The placement of the dam across the stream caused the water to slow and build up behind the dam while some still continued underneath and through it. The water pooled high enough, that with the help of us walking along the bank to compact the soil in paths directed east, it overflowed the bank and flowed east and joined the water flowing from the modified step-pools (fig. 7).

FIG. 7. New sheet flow from burlap check dam.

The natural debris dam had similar results. Water was able to pass through it, but it slowed the flow enough to allow the water to pool behind it and break over the bank and towards the east (fig. 8). The flow spread out and some flowed south eventually joining the main stream while some flowed east and met the flows from the previous two models (fig. 9). None of the new flows were channelizing extensively as they spread throughout the bog and the water continued to take the paths of least resistance, visibly increasing the sheet flow overall. Each micro-dam was effective in rerouting new flows to the east, allowing seepage in dry but historically wet areas.

FIG. 8. Water in stream pooling and breaking over the bank creating new sheet flow patterns.

The filter strip was designed to encourage this water spreading and prevent channelization of new sheet flow as well as help filter sediment, but results for this will not be clear until the wetland species are planted when the weather warms.

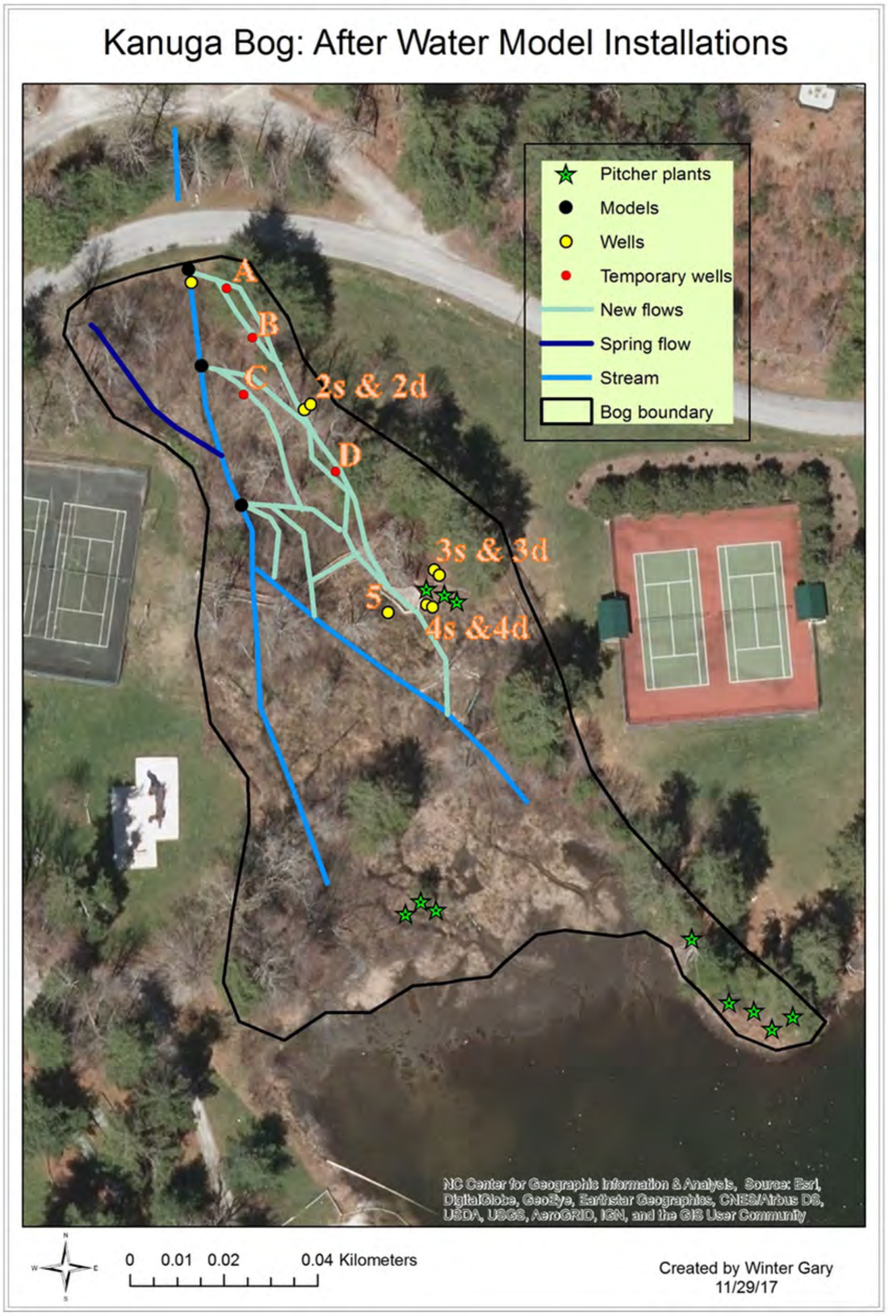


FIG. 9. Kanuga Bog stream flow after water/ sediment control model installations including estimations of the new sheet flow patterns in the eastern section. Created in ArcMap 10.5 (NC OneMap 2016)

Water Table Measurements

Temporary wells A, B, and C and permanent wells 2s and 2d all showed an overall increase in water levels since the date they were installed on October 6th. Between this date and the first measure a week later, after the drop structure and modified step-pools were installed, the groundwater level decreased at well A, but increased in all except one of the others. Figure 10 demonstrates the abrupt increase in water levels between October 12th, when the drop structures and modified step-pools began redirecting the water to the east portion, and the next day. Water levels in well A increased by 0.083 m in less than 24 hours. Well B increased by 0.058 m, but the rest of wells did not vary much from their previous measurements. Figure 11 demonstrates that there was no precipitation between October 12th and October 13th to account for the groundwater increase at these two wells. The week of October 26th shows that wells A and B decrease in groundwater levels, while the other wells increased slightly (table 2). The next day, wells A and B had groundwater levels higher than past measurements. This can be seen visually in figure 9. Wells C, 2s, and 2d only changed 0.003 m to 0.006 m from the previous day. Again, figure 11 shows that there was no precipitation on October 26th or 27th to account for this abrupt increase in groundwater level at these wells. Over the next few weeks, wells A and B continued to increase while wells C, 2s, and 2d started to see more drastic changes in water levels after the burlap check dam was installed on October 27th. They each increase significantly from their pre-model installation measurements. Wells A, B, C, 2s, and 2d increased an average of 0.099 m. Well D, which was also a temporary well, was installed three weeks later, south of the others. It was the only well that had an overall decrease in water levels. It decreased 0.054 m in the three weeks it was installed. Figure 10 displays this decrease in well D.

TABLE 2. Groundwater level measurements (meters) for wells A, B, C, 2s, and 2d prior to and after water/ sediment model installations. Groundwater level measurements collected for Well D after water/ sediment model installations only.

Date	Well A	Well B	Well C	Well D	Well 2s	Well 2d
10/6/2017*	1.561	1.317	1.207	~	1.064	1.058
10/12/2017	1.545	1.335	1.268	~	1.085	1.055
10/13/2017	1.628	1.393	1.277	~	1.088	1.055
10/20/2017	1.615	1.387	1.289	~	1.094	1.079
10/26/2017	1.591	1.375	1.305	~	1.097	1.122
10/27/2017	1.670	1.423	1.308	~	1.103	1.119
11/3/2017	1.664	1.469	1.366	0.981	1.125	1.146
11/9/2017	1.676	1.481	1.338	0.924	1.125	1.146
11/17/2017	1.670	1.466	1.295	0.927	1.116	1.131
11/24/2017	1.689	1.466	1.295	0.927	1.119	1.134

*Measurements taken prior to water/ sediment model installations
~No data collected

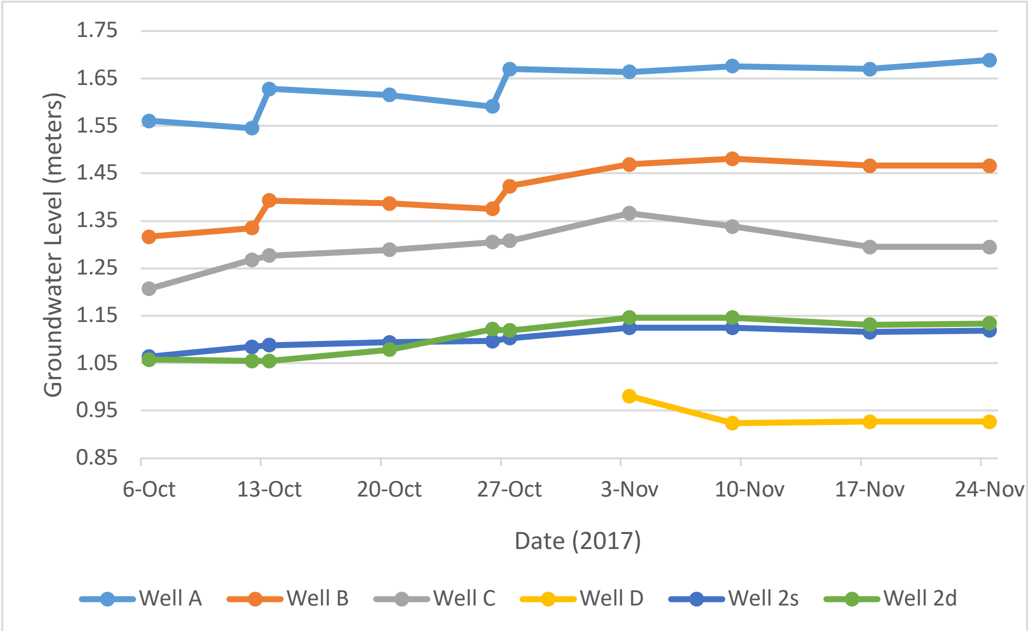


FIG. 10. Groundwater level changes for wells in the eastern section of bog.

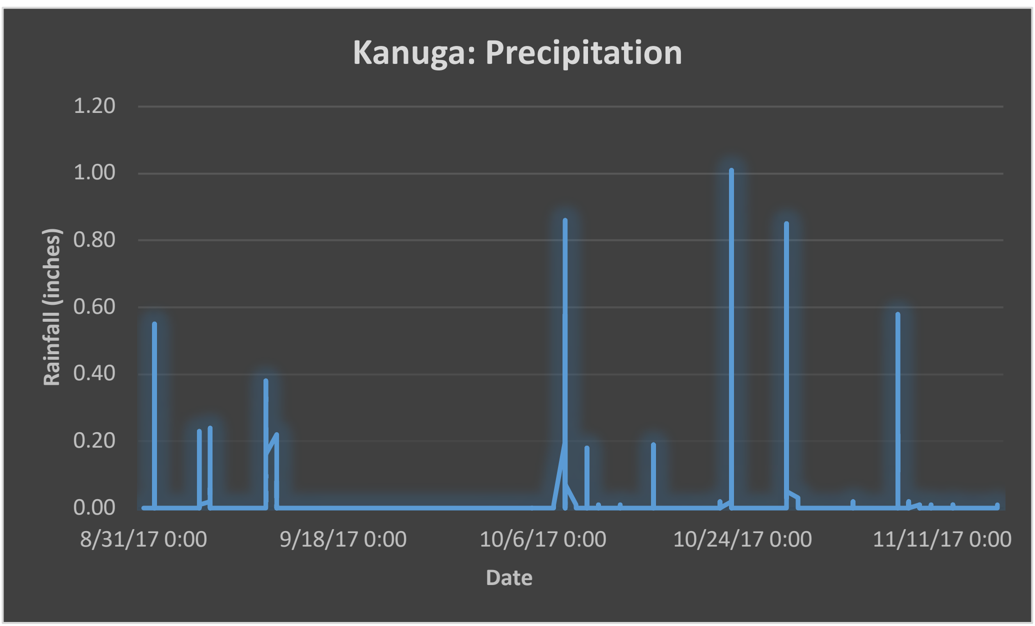


FIG. 11. Precipitation data collected from Kanuga Bog's rain gauge from September through November 2017. (1 in= 2.54 cm).

DISCUSSION

The abrupt groundwater level increase for wells A and B between October 12th and 13th can be explained by the increase in sheet flow caused by the drop structure and modified step-pools. We concluded that it was not due to precipitation since there was no precipitation in the bog during these two days (fig. 11). The decrease in groundwater levels in these two wells between October 20th and 26th can be explained by the very large storm that passed through Hendersonville on October 23rd (fig. 11). The storm waters that rushed through the culvert that day caused the drop structure to partially collapse and the modified step-pools to become filled with sediment that prevented water from flowing through them and into the eastern section of the bog (fig. 12). Sediment also filled the eastern section of the bog around well A, making it visibly drier. We had to clean the sediment from the step-pools and reconstruct the drop structure to get the water flowing towards the east again. The next day, October 27th, wells A and B were at their highest groundwater levels to date. The decrease in groundwater levels after the storm and the drastic increase in water levels after we fixed the drop structure and step-pools, despite no rain on the 26th or 27th, endorse our models as being the major influence of the groundwater changes in the eastern section of the bog.

These changes occurred before the burlap dam was placed, and the natural debris dam was too far south of wells A and B to make a major impact on them. For these reasons, the drop structure and modified step-pools had the most extensive impact on increasing the water table. The two check dams amplified the effects of our main water/ sediment control model. The significant increase in groundwater levels for wells B, C, 2s, and 2d between October 27th and November 3rd were most likely due to the installation of the burlap dam. It is unlikely that the increase was from groundwater discharge finding its way into the bog from rainfall on October 28th (fig.11), since we would did not see the same increase between November 3rd and 9th due to the high levels of precipitation on the 7th. The water levels in well C decreased during that time, however, while levels in 2s and 2d remained constant. For the following weeks, the groundwater levels in wells A, B, C, 2s, and 2d fluctuated, but there were no drastic changes, and they were all much higher than their pre-model measurements.

FIG. 12. (a) Left. The modified step-pools filled with sediment causing them to be ineffective at redirecting water to the east. (b) Right. The area to the east of the step-pools filled with sediment and visibly drier than when the step-pools are workings.

The decrease in groundwater levels for well D over the month it was installed may have been caused by us trampling the area while planning our filter strip. We may have compacted the soil making it difficult for the water to percolate down into the water table at this location, but the exact cause of the decrease is unclear. Sheet flow physically covered the area around the well as shown in figure 13, so the decrease in the groundwater levels at this spot does not mean that our models were unsuccessful.

FIG. 13. Sheet flow around well D.

The groundwater levels recorded from permanent wells 2s, 2d, 3s, 3d, 4s, 4d, and 5 all increased after model installations. This indicates sheet flow has not only impacted the groundwater levels in the northeast section, but also further south where the pitcher plants are located, the main goal of the study. While we did not measure the depth of the new sheet flow, there were a couple portions in the northeastern section of the bog that became slightly inundated which is less favorable for bog turtles. Sediment accumulation should build up on these areas over time making the habitat less inundated while the new flow helps keep the soil saturated to attract the turtles. There are many areas further south that did not become inundated such as near wells 3s, 3d, 4s, 4d, and 5 and in areas between the new flows and main stream. Groundwater levels did rise and the soil became more saturated in these areas making them more favorable for the turtles, therefore, they should be monitored for bog turtle populations in the future.

The saVAP score of 3.1 classified the overall stream as fair quality. This procedure can be replicated in the future to see if the stream habitat quality changes. The shift in inflow water quality from excellent to good, shown through the biotic index scores, cannot be attributed to our water/ sediment control models since it is upstream of the bog and models. We also cannot attribute the slight improvement in water quality of each of the outflows to our models, but it does reassure us that the models are not negatively impacting the water quality of the stream at this point in time.

No negative water quality impact was confirmed by the chemical analysis conducted prior to and after the model installations at the inflow, both outflows, above the boardwalk, and at wells 1, 2s, 2d, 3s, 3d, 4s, 4d, and 5. Each time, permanent wells 3s and 3d had the highest temperatures, because they are directly in the sun and the water within them does not flow quickly like in a stream. The range of temperatures overall was caused by the weather cooling during our study period and the location of the sites relative to the sun and shade. The pH ranges prior to and after the model installations, 5.4 to 7.1 and 5.4 to 7.4 respectively, did not vary much from the pH values of 5.7 to 6.8 Nguyen found during her hydrogeochemistry research in Kanuga Bog (2015). It is logical that the water in the stream is more basic than the groundwater located within the acidic bog soils.

The conductivity within wetlands tends to range from 50 to 50,000 μScm^{-1}, however, outflow A is the only site that came close that range (Sanders 1998). This is because Kanuga bog has a channelized stream flowing through it instead of the relatively deep, stagnant water typical of bogs. Outflow A was stagnant compared to the other flows in the bog which means it would have accumulated more dissolved solids. This explains why it was an outlier with conductivity values of 46.3 μScm^{-1} and 49.3 μScm^{-1}, however, all the conductivity values except for one fall within the stream conductivity range of 2 to 100 μScm^{-1} (Sanders 1998). The only value that did not fit in this range was well 4d which measured 0.13 μScm^{-1}. It is unclear why this value dropped 20.47 μScm^{-1} from its original value, and it should be monitored to see if it remains low or if there was an issue with our conductivity meter.

The decrease in turbidity by well 1 may have been due to the drop structure slowing the water enough for particles to settle out before reaching the well located downstream. Outflow A was also an outlier for turbidity. It increased by 10.5 NTU after the models were placed. The turbidity averaged 3.19 overall, which means that outflow A was 11 to 21 NTU higher. The stagnant nature of this section of the stream, being less oxygenized, had algae growing on the surface, and was mucky overall.

This study would have benefited from collecting more groundwater level measurements in the temporary wells prior to the models installations, however, time did not permit. Measuring sediment accumulation prior to and after the models were installed would have been ideal in order to see how much the models were helping decrease this pollutant. More quantitative data for whether the new flows would attract bog turtles would have been useful. We could have measured water table distance to the soil and the water depth above ground to ensure the soils were saturated and that there were not many areas inundated more than 5cm which becomes unfavorable for bog turtles (Feaga 2010).

In the future, the study can be expanded by implementing water/ sediment control models in the water pooling and flowing underneath the boardwalk. Channelization should be discouraged in this area before it amplifies. Extending the water quality measurements such as the chemical analyses would benefit the bog through recording any changes that may suggest a problem due to sedimentation, runoff from the parking lot, or materials used in this study.

CONCLUSION

This study was able to guide water into historically wet areas in Kanuga Bog which improved the habitat for bog turtles and the mountain sweet pitcher plant populations. The experimental water/ sediment control models including the drop structure, modified step-pools, burlap check dam, and natural debris dam were successful in increasing sheet flow in the bog. Proper management of southern Appalachian Mountain wetlands and correcting artificial alterations to their hydrology, as demonstrated, are essential steps in conserving and recovering federally endangered species. Management is also vital for wetlands to continue to regulate water flow by storing storm waters during floods and slowly releasing them during droughts and allowing them to continue to act as water filters for species downstream.

ACKNOWLEDGEMENTS

I would like to sincerely thank Clint DeWitt of Kanuga Conferences Center for directing this study and offering great mentorship and support throughout the semester. Great thanks to Jeffrey Wilcox who offered his advice, equipment, and time to this study. I would also like to thank the staff at Highlands Biological Station, particularly James Costa and Sarah Workman, for matching me to this internship and being supportive guides along the way. Finally, thanks to all the IE students who contributed their time and effort into helping with this project through peer reviews.

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BROOK TROUT DISTRIBUTION IN THE UPPER TUCKASEGEE WATERSHED

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Abstract. This study was conducted to learn more about brook trout, *Salvelinus fontinalis*, distribution throughout the Upper Tuckasegee watershed and determine predictors of brook trout presence in a given stream in the watershed. For each stream sampled, we gathered water quality data, physical stream habitat characteristics, collected benthic macroinvertebrates, and electroshocked for fish. The presence of brook trout can serve as an indicator of the health of headwater streams and tributaries. We collected brook trout at 10 of the 17 sites that we sampled. Using an information theoretic approach, we found that the type of bottom substrate was the best predictor for brook trout detection, and the presence of collector-gatherer macroinvertebrates, predator macroinvertebrates, and the abundance of low pollution-tolerant macroinvertebrates were the strongest predictors of brook trout occupancy. Our findings can be applied to sustainable land management practices by both land owners and land managers.

Key words: electroshocker; estimation; habitat assessment; macroinvertebrates; occupancy; Salvelinus fontinalis; *southern Appalachians; Tuckasegee River watershed*

INTRODUCTION

Brook trout (*Salvelinus fontinalis*) are members of the Salmonidae family, belonging to the char subgroup. The fish are characterized by a dark-colored ventral side speckled with light spots, a red dorsal side, and small scales. Colloquially known as “speckled trout” and “brook char,” this game fish requires cool water to survive. Brook trout are found in headwater streams, generally within pool habitats and in slow-flowing sections of streams (Ecret and Mihuc 2013). The trout prey on a wide array of macroinvertebrates and are important population regulators in a functional river ecosystem. In terms of chemical composition of habitat, brook trout tolerate a water pH range of about 4.1 - 9.5 without any apparent preference for a particular pH within this range (Creaser 1930). Brook trout are sensitive to water temperature and generally are not present in streams with water temperatures above 19 degrees Celsius (Creaser 1930). As a bioindicator species that thrives in cold, highly-oxygenated water with low levels of pollution, brook trout presence or absence provides valuable insight into the ecology of the area and health of stream habitats (Trout Unlimited For the Eastern Brook Trout Joint Venture 2006).

Within the *S. fontinalis* species are two genetically distinct populations: the northern brook trout and the southern brook trout (Hayes et al. 1996). The southern strain, the focus of this study, is indigenous to the southern Appalachian Mountains, ranging from northern Georgia and South Carolina to Virginia (National Park Service 2015). This range is shrinking, however, as brook trout are heavily influenced by anthropogenic disturbances including the introduction of invasive species, logging, acid deposition, competition, and increased fishing pressure (Marschall and Crowder 1996). Brook trout populations are also threatened due to competition with non-native invasive species such as rainbow and brown trout (Hayes et al. 1996). Because of these pressures, especially pollution (nutrient runoff from agriculture, sediment from construction projects, logging, roads, etc.), brook trout habitats have become largely limited to the uppermost headwaters of rivers (Marschall and Crowder 1996, Trout Unlimited For the Eastern Brook Trout Joint Venture 2006). Reintroduction efforts in several streams throughout the southern Appalachians have had largely negative effects, as these programs stocked the non-native northern brook trout. This resulted in displacement and hybridization, ultimately reducing the genetic diversity of the southern brook trout populations (Hayes et al. 1996). By categorizing and monitoring the native brook trout in their natural range, we can gain a better understanding of how to improve conservation and rehabilitation efforts.

We sampled brook trout at 17 stream sites in the Upper Tuckasegee watershed and collected site-specific data on three habitat factors: water chemistry, macroinvertebrate diversity, and habitat quality assessment. Each site was evaluated for the presence of brook trout, indicating brook trout habitat viability. This allowed for an analysis of trends in brook trout distribution and habitat preferences.

At each site, several water chemistry tests were conducted. Water chemistry parameters provide quantitative, numerical data that is useful for statistical analysis and modeling. Taking chemistry data can also prove to be critical due to the brook trout's known sensitivity to pH and temperature (Creaser 1930).

A relationship between brook trout presence and benthic macroinvertebrate diversity was explored in first and second-order streams because macroinvertebrates can serve as indicator species for presence of fish and pollution levels. Compared to an instantaneous measurement provided by chemical testing, population structure of macroinvertebrates can reflect continuous or long-term stream health. In addition to providing a more temporally complete assessment of stream health, identifying diversity and abundance of macroinvertebrate taxa is faster and more affordable than chemical testing (Hocutt 1975).

The visual habitat assessments provided a quantitative metric representing observational data of the stream. This presented an opportunity to look for trends in brook trout presence and more nuanced factors such as bottom substrate type, canopy cover, and bank vegetation.

The goal of this project is to identify and document brook trout populations in the Upper Tuckasegee watershed and develop a model that accurately predicts the presence of brook trout based on environmental variables. Such a model will be valuable in assisting managers and ecologists in identifying prospective habitats suitable for brook trout and managing current habitats for population preservation. As a species both ecologically and culturally significant to the southern Appalachians, brook trout are essential in preserving the natural heritage of this region.

METHODS

Site Selection

We limited this study to sampling only "blue-line" streams above 3,000 feet (914.4 m) that were on private land located in the Upper Tuckasegee watershed (USGS cataloging unit 06010203). Sampling was done between September 11th and November 6th 2017.

Brook Trout Sampling

To accurately estimate the presence of brook trout, fish samples were collected at various sites. Sampling was conducted in three 30-meter sections of the stream sites, each separated by 10-meter breaks where fish were not sampled (to provide a buffer to reduce disturbance created by sampling). Moving upstream, a Halltech Aquatics® HT-2000 backpack electroshocker was used to stun all fish within a small radius in the stream. A group with hand-dip nets followed the electroshocker, collecting and depositing the fish into a bucket partially filled with water. We identified and tallied the fishes, measured head-to-tail length of brook trout to the nearest centimeter, and checked for disease, ectoparasites, and physical anomalies. After the measurements were completed, all fish were released back into the stream. The average width of the stream was also included in the measurements. All possible fish habitats within the sample reach were sampled, including pools, undercut banks, and other areas possibly occupied by fish.

Macroinvertebrate Collection

Benthic macroinvertebrates at stream sites were collected using an adapted version the Qual 4 Method from the Standard Operating Procedures for Collection and Analysis of Benthic Macroinvertebrates by the NC Department of Environmental Quality (2016). The collection procedure consisted of one 1-minute riffle-kick sample in approximately two meters reach of substrate, one 5-minute sweep-net sample, and one 5-minute leaf-pack sample. Four 5-minute visual collections were performed to target habitats that may have been missed or under-sampled during the first three sample methods. We put the organic materials from each method into shallow white tubs in order to sort for specimens, which we then preserved in glass vials containing 95% ethanol for each respective method of collection.

We identified the collected macroinvertebrates to the family level using dichotomous keys from Merritt et al. (2008) and Project Search (McCollum 2009), recorded the families found at each stream site along with each families' tolerance value and functional feeding group, and calculated the family-level biotic index (FBI) for each (Hilsenhoff 1988). Using the index, we calculated scores for each site from 0.0, indicating more pristine conditions, to 10.0, indicating ecological stress and pollution, in order to evaluate the water quality and determine the amount of organic pollution likely within each stream site (NC Department of Environmental Quality 2016). The score was calculated by multiplying the abundance value (n) of each family (i) by the tolerance value (T) for that family (i), summing the products, and dividing by the total arthropods in the sample (N).

$$FBI = \frac{\sum(n_i)(T_i)}{N}$$

Habitat Assessment

The North Carolina Habitat Assessment Field Data Sheet included in the Standard Operating Procedures for Collection and Analysis of Benthic Macroinvertebrates by the NC Department of Environmental Quality (2016) was completed to give each site a quantifiable score representing the average condition of the stream. It was conducted over a 100-meter reach and incorporates a standardized qualitative approach. Observations included scores for stream bank condition, canopy cover, channel modification, bottom substrate, extent of light penetration, and riparian vegetative zone width. These values sum to yield a concrete appraisal of the biological, chemical, and physical health of the stream.

Over each sampling area, the Southern Appalachian Stream Visual Assessment Protocol (saSVAP) was also implemented to make a preliminary evaluation of stream health. The saSVAP is an iteration of the USDA published Stream Visual Assessment Protocol (2009) customized for the Southern Appalachian Region by the Land Trust for the Little Tennessee (LTLT) and

University of Georgia (Sullivan et al. in press). Similar to the NC Habitat Assessment, the saSVAP assigns a numerical score to stream health based on a qualitative assessment of eleven elements: bank condition, riparian buffer quantity and quality, canopy cover, riffle embeddedness, trash and garbage, non-trash/nutrient pollution, livestock, pools, available habitat/cover, and barriers to fish movement (saSVAP). Elements of this score, in conjunction with the North Carolina Habitat Assessment Field Data Sheet, were used as an indication of stream health to later correlate with *Salvelinus fontinalis* presence.

Water Chemistry

Measurements for chemical analysis were collected with a YSI Model 85 Handheld Oxygen, Conductivity, Salinity, and Temperature System and an Oakton Model 35615-80 PH110 Meter Kit. To supplement the pH, dissolved oxygen, temperature, and conductivity data, a qualitative assessment of water turbidity was also recorded. In the interest of avoiding any influence of disturbance from other sampling activities, water chemistry data was measured above the uppermost 30-meter sampling section.

Statistical Analysis

Brook trout catch data per each unit were categorized into two states: not detected, 0; or present, 1. We evaluated the relative plausibility of over 60 occupancy and detection models for brook trout (Table 3); each model represented *a priori* hypotheses regarding the effects of physical, chemical and spatial site-specific covariate effects on detection and occupancy. Estimates were given for brook trout presence, defined above, as the probability of capturing a brook trout when they are present (p) and the proportion of area occupied by brook trout (Ψ). Detection was modeled as constant among all sites, varying among sites due to channel modification, bottom substrate, in-stream habitat, pool variety, riffle habitats, bank stability and vegetation, light penetration, and riparian vegetative zone width. Candidate models also were constructed based on site-specific covariate effects on occupancy (Table 3). These models were based on previous studies and included occupancy as: constant; a function of the previous listed covariates as well as macroinvertebrate related covariates: FBI scores, frequency of low/high tolerance macroinvertebrates, and frequency of feeding group types. All of the above covariates were modeled on a continuous scale.

Candidate occupancy models were fit for each species using the occupancy estimator in Program MARK (White and Burnham 1999). We used an information theoretic approach (Burnham and Anderson 2002), Akaike's Information Criteria (AIC; Akaike 1973), to evaluate the relative fit of candidate models and calculated Akaike weights (w_i) that range from 0 to 1, with the most plausible model having the greatest weight (Burnham and Anderson 2002). Models with weights that were within 1/8th of the value of the best-fitting model were considered in the confidence set of candidate models (Royall 1997). We based all inferences on parameter estimates from the best-fitting model and assessed precision by calculating 95% confidence intervals.

RESULTS

We sampled a total of 17 stream sites (Table 1), all on private land and above 3,000 feet elevation (914.4 m) in the Upper Tuckasegee watershed (Fig. 1). At 14 of the sites, we sampled for macroinvertebrates which were categorized based on feeding type including predators (Fig. 2) and collector-gatherers (Fig. 3).

The FBI scores represent the species richness of the macroinvertebrates present and their tolerance to water pollution (Table 4).

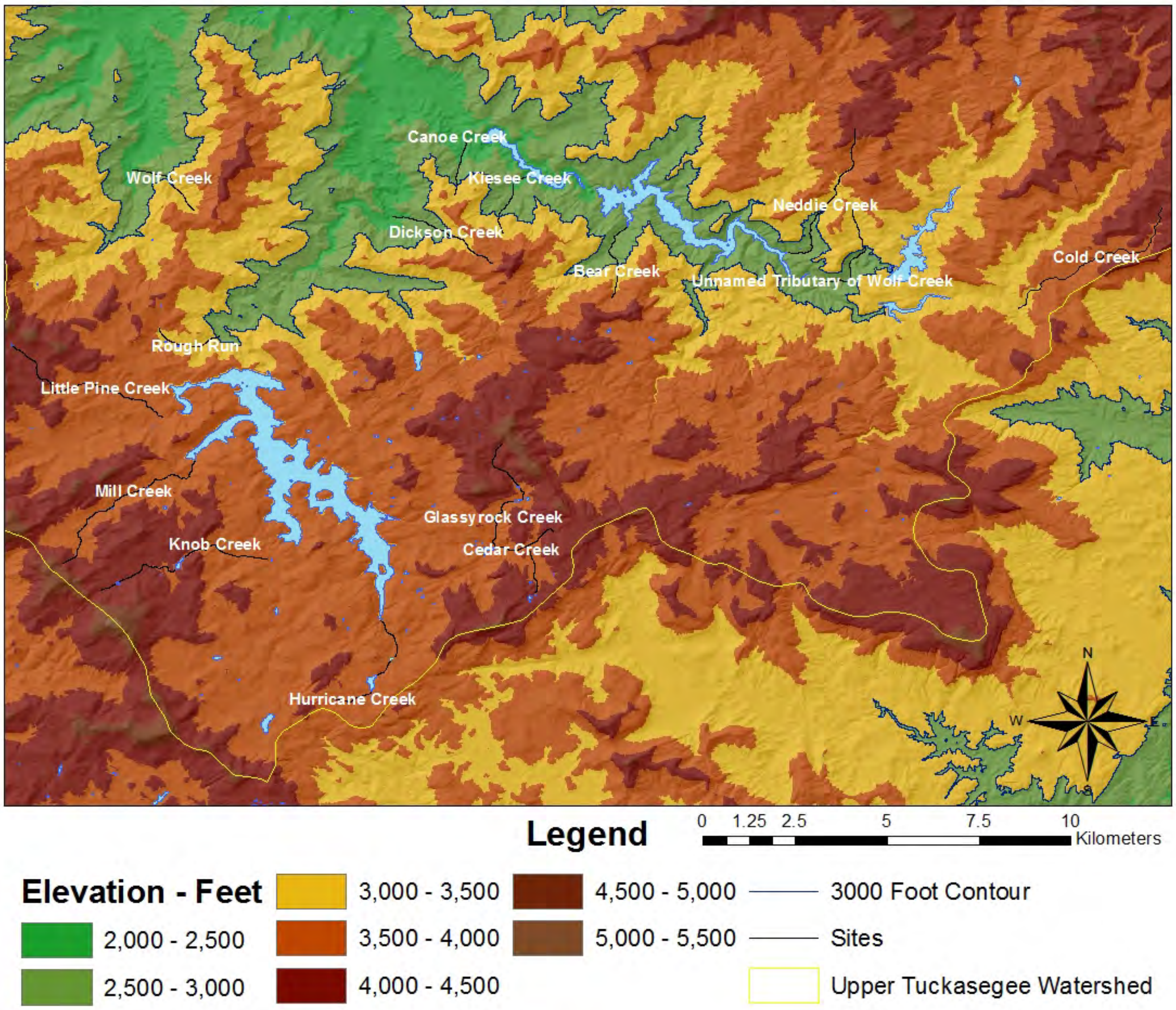

FIG. 1. Map of the locations of the 17 streams sampled in the Upper Tuckasegee watershed. Areas that are shaded yellow, orange, red, and brown are elevations above 3,000 ft. The yellow line delineates the Tuckasegee watershed, with the purple line showing 3,000 ft. elevation boundary. (1 Foot = 0.305 Meters).

TABLE 1. Sampling sites by stream name and notation used.

Creek Name	Abbreviation
Hurricane Creek	HC
Little Pine Creek Section 1	LPC1
Little Pine Creek Section 2	LPC2
Little Pine Creek Section 3	LPC3
Wolf Creek Unnamed Tributary	UTW
Neddie Creek	NC
Bear Creek	BC
Dickson Creek	DC
Knob Creek	KC
Mill Creek	MC
Rough Run	RR
Wolf Creek	WC
Cedar Creek	CC
Glassyrock Creek	GC
Canoe Creek*	CaC
Kiesee Creek*	KC
Cold Creek*	CoC

*Not sampled for macroinvertebrates.

Habitat Assessments and Water Quality

Brook trout were absent from several streams: Hurricane Creek, Dickson Creek, Rough Run, Canoe Creek, Kiesee Creek, and Wolf Creek off Cullowhee Mt. Road and an unnamed tributary of Wolf Creek. In these streams, the pH ranged from 6.49 to 7.41, and the conductivity ranged from 4.3 μS/cm to 33 μS/cm. Brook trout were found at all other sites: Little Pine Creek, Neddie Creek, Bear Creek, Knob Creek, Mill Creek, Cold Creek, Cedar Creek, and Glassyrock Creek. In these streams, the pH ranged from 5.56 to 8.2, and the conductivity ranged from 1 μS/cm to 126 μS/cm. (It should be noted that early in the study the Oakton pH meter kit stopped functioning; pH test strips were used for most stream sites, but because the measurements were within a biologically meaningful range, we believe this substitution did not incorporate bias.) In the brook trout occupied streams, the turbidity varied greatly between the streams, and the temperature ranged from 9.17 degrees Celsius to 17.2 degrees Celsius. Brook trout were present in streams with a habitat assessment score that ranged from 36.5 to 86, and the saVAP scores ranged from 30 to 51.5 (Table 2). At streams where brook trout were present, bottom substrate ranged from 1 (substrate nearly all silt or clay) to 15, where the substrate had a good mix of gravel, cobble, or boulders and was embedded less than 20%. Streams where brook trout were not present had a bottom substrate type of 3, where substrate had a good mix of gravel, cobble, or boulders and was embedded less than 80%, or type 8, where the substrate was either gravel that was less than 50% embedded or where substrate had a good mix of gravel, cobble, or boulders was embedded less than 40% to 80%, or type 11 where the substrate was only gravel and cobble and was embedded 20% to 40%.

TABLE 2. saVAP and Habitat Assessment Scores.

Creek Name	saSVAP	Habitat Assessment
Hurricane Creek	35	39
Little Pine Creek Section 1	49	86
Little Pine Creek Section 2	46	75
Little Pine Creek Section 3	45	75
Wolf Creek Unnamed Tributary	43.5	53
Neddie Creek	51.5	78
Bear Creek	50	80
Dickson Creek	49.5	83
Knob Creek	30	47.5
Mill Creek	42.5	63
Rough Run	38	52
Wolf Creek	47	75.5
Cedar Creek	32	36.5
Glassyrock Creek	43	40
Canoe Creek*	41	70
Kiesee Creek*	40	64
Cold Creek*	42.5	85

*Not sampled for macroinvertebrates.

Biotic Indices

We identified 41 macroinvertebrate families from the sampled stream sites. We collected the most diverse sample of macroinvertebrates from Knob Creek, with 19 different families identified and the least diverse sample from Cedar Creek with only six different families identified (see table in Appendix A). The FBI scores we calculated for each stream indicate that all the sites we sampled had excellent or very good water quality suggesting that organic pollution was minimal. The average FBI score was 2.8 and the scores ranged from 1.1 to 4.0 (Fig. 4). The macroinvertebrates we collected were categorized into five feeding groups: scraper, predator, collector-filterer, collector-gatherer, and shredder. On average, predators and shredders were the most abundant (Fig. 2).

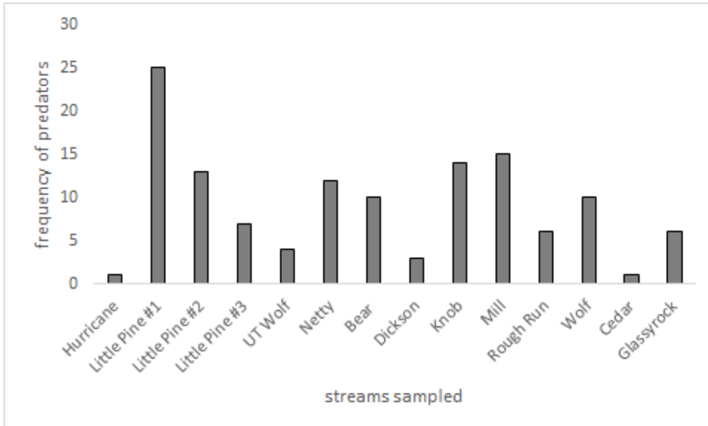

FIG. 2. Macroinvertebrates: predator feeding group frequencies (individuals per site).

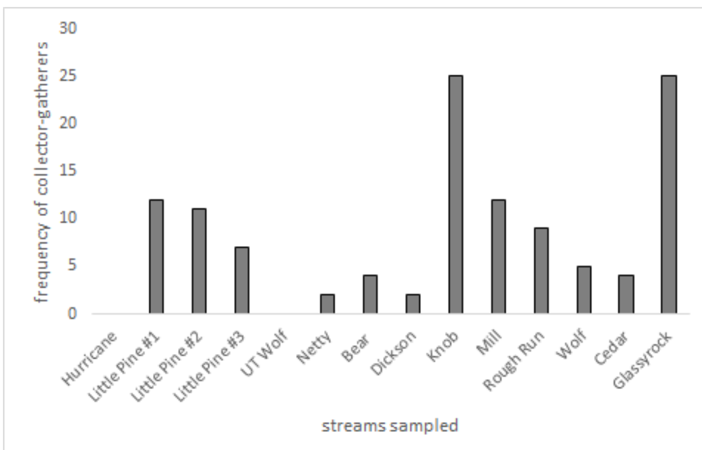

FIG. 3. Macroinvertebrates: collector-gatherer feeding group frequencies (individuals per site).

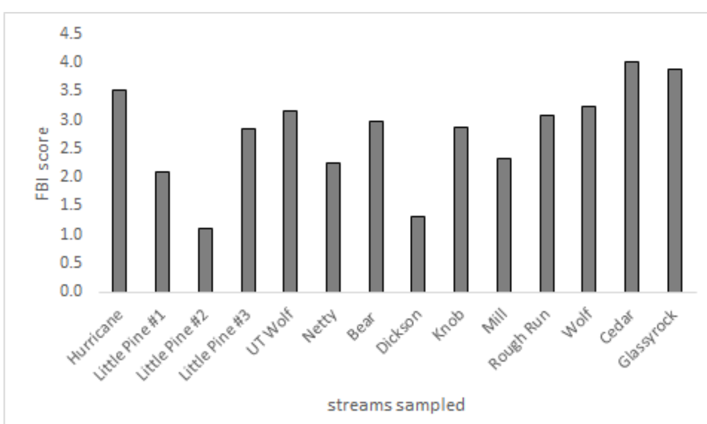

FIG. 4. Family-level biotic index scores with presence or absence of brook trout. Based on the table by Hilsenhoff (1988), the water quality of all the streams are excellent or very good, indicating a low presence of pollution (Appendix B).

TABLE 3. Covariates of occupancy(Ψ) and detection(*p*) with the resulting AIC and weighted AIC scores. Dashed line represents models within the Confidence Set, having AIC weights within 1/8 of the best fitting model.

P	Ψ	AIC	AIC_w
bottom substrate	feeding type, collector-gatherer	55.5471	0.28048
bottom substrate	feeding type, predator	55.5881	0.27479
bottom substrate	tolerance value, low	56.8716	0.14464
bottom substrate	tolerance value, high	57.9610	0.08389
bottom substrate	feeding type, shredder	58.8737	0.05315
bottom substrate	FBI score	59.3032	0.04288
bottom substrate	feeding type, scraper	59.3117	0.04270
bottom substrate	feeding type, collector-filterer	59.3219	0.04248
bank stability	tolerance value, high	61.4351	0.01477
constant	Constant	62.2682	0.00974
bank stability	feeding type, predator	64.1357	0.00383
bank stability	feeding type, collector-gatherer	64.5878	0.00305
bank stability	tolerance value, low	66.2100	0.00136
bank stability	feeding type, shredder	68.2048	0.00050
bank stability	bottom substrate	68.3016	0.00048
bank stability	feeding type, collector-filterer	68.4869	0.00043
bank stability	feeding type, scraper	68.5805	0.00041
bank stability	FBI score	68.6207	0.00041

Statistical Analysis

In Program MARK, we ran over 60 models to determine which physical, chemical, and spatial site-specific factors are plausible predictors of detection and occupancy of brook trout. We looked at all of our stream parameters as predictors of occupancy and detection including channel modification, bank stability, bottom substrate, pool variety, and light penetration. We ran models with multiple combinations of parameters to determine the significance of combined factors. Our confidence set of models included bottom substrate as a predictor of detection, and occupancy being a function of macroinvertebrates. Frequency of collector-gatherers were the best predictor for occupancy, with frequency of predatory macroinvertebrates and species with a low tolerance to pollution being the other two models included in our confidence set.

Detection:

Bottom substrate was the strongest predictor for detection. All models that included bottom substrate as a function of detection came out as a better predictor, regardless of what predictor was used for occupancy. Bottom substrate had 19.0 (0.28048/0.01477) times more support than the second best approximating model for detection, bank stability (Table 3).

Bottom substrate values were based on physical habitat observations using the NC Habitat Assessment. We categorized bottom substrate into three components, based largely on embeddedness. Poor substrates were those with silty soils or areas with approximately 80% embedded materials. Moderate substrates had hard substrate material present with roughly 50% embeddedness. Good substrates had a well-represented mix of material, such as gravel, cobble, and boulders, with low embeddedness.

Average detection estimates for each category were 42, 67, and 90 percent for poor, moderate, and good bottom substrates, respectively (Fig. 5). Keeping in mind that we sampled each site three times, we were $1 - (1 - 0.42)^3 = 80.5\%$ certain that we would have captured at least one brook trout in that area, if it were present. Compare this to $1 - (1 - 0.9)^3 = 99.9\%$ certainty of capturing a brook trout, given it is present, at a stream with good bottom substrate.

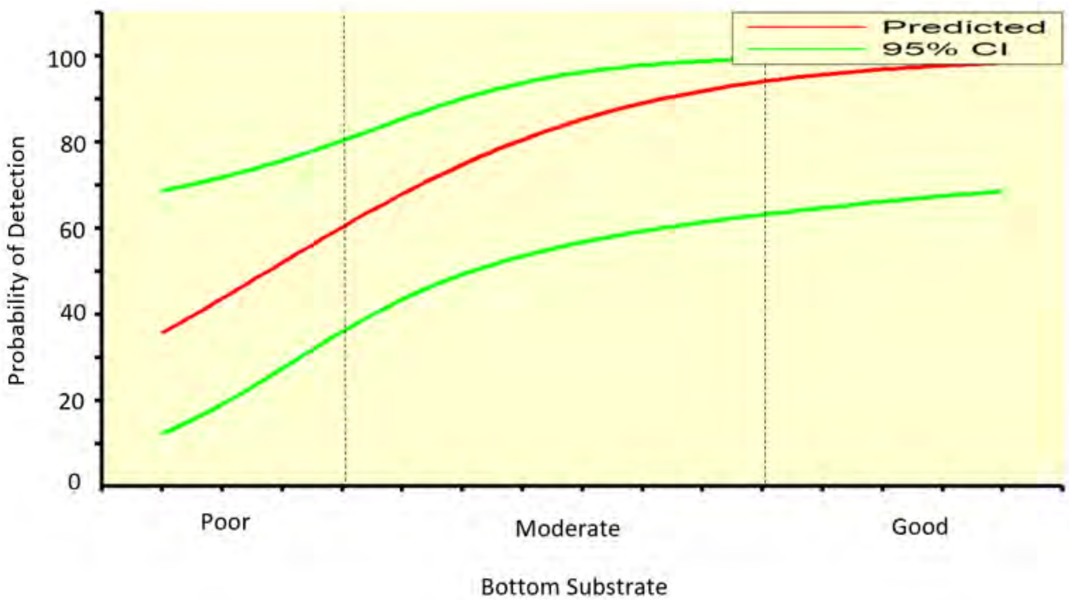

FIG. 5. Bottom substrate confidence model based off criteria within the NC Habitat Assessment representing the detection estimates of brook trout. Poor scores represent greater than 50% embeddedness of substrate, including sand and bedrock. Moderate scores represent around 50% embeddedness of substrate such as gravel, cobble, and boulders. Good scores represent less than 50% embeddedness of substrate such as gravel, cobble, and boulders, ideally around 20-40%.

Occupancy:

Naïve occupancy estimates are estimates that do not account for other factors that affect occupancy values. It is calculated by determining the sites that where observed brook trout/ the total number of sites; (10/17) = 58.8%. However, by accounting for incomplete detection, the general occupancy estimate, which is the probability a brook trout is present in any given stream within our area, without considering any covariate influences, was 67.1%. This emphasizes the need to account for incomplete detection.

While frequency of collector-gatherers had (0.28048/0.27479) = 1.02 times more evidence at predicting occupancy of brook trout over the frequency of predators, we cannot omit the plausibility of predators as a factor for occupancy (Table 3). Collector-gatherers were (0.28048/0.14464)=1.92 times better predictor over low pollution tolerant macroinvertebrates also included in our confidence set of models.

Occupancy estimates, based on the frequency of collector-gatherers, ranged from 25% to 90% (Fig. 6). Sites with little to no collector-gatherers, were not likely to hold brook trout, whereas sites with greater frequency (20+), had occupancy estimates around 90%. Presence of collector-gatherer macroinvertebrates, whether as a food source, or as an indicator of water quality, are an important predictor of brook trout.

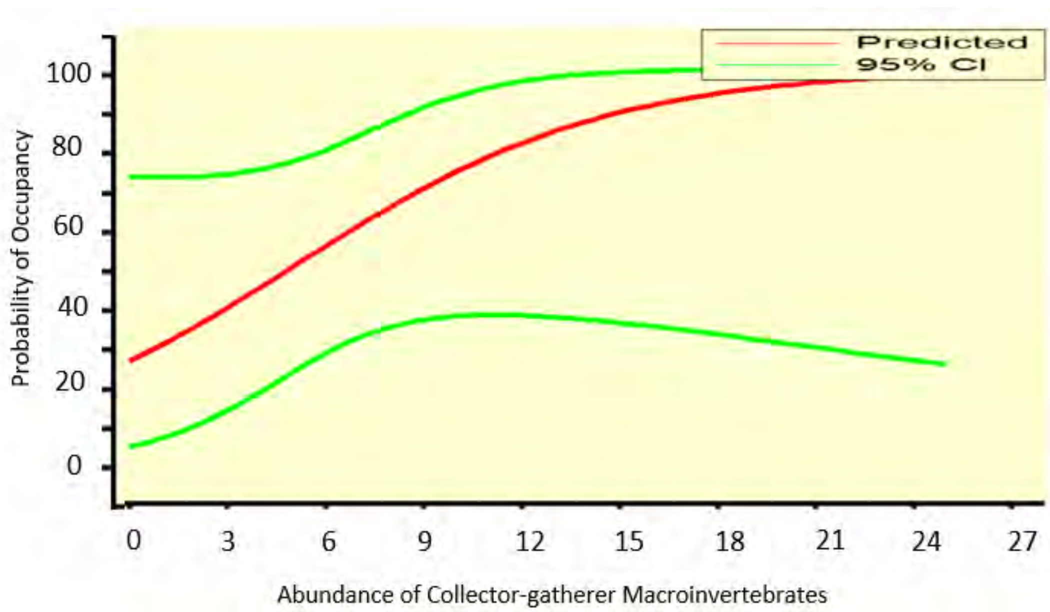

FIG. 6. Best model representing collector-gatherer macroinvertebrates affecting the occupancy estimates of brook trout.

DISCUSSION

The abundance of collector-gatherer macroinvertebrates in streams was the most plausible factor in determining the presence of brook trout. We postulate two contributing explanations to this result. Macroinvertebrates are a primary component of brook trout diet (Montana Natural Heritage Program and Montana Fish, Wildlife and Parks). And as indicator species, macroinvertebrates, through the richness of their populations, can be proxies of long-term stream health (Hocutt 1975, Wallace and Webster 1996). There were multiple streams without brook trout that also had strong FBI scores. This demonstrates how macroinvertebrates population structure can persist in streams where brook trout might not. Conversely, no brook trout were found at sites without macroinvertebrates. The candidate set of models suggest that for environments to be suitable for brook trout, they are likely to be occupied by macroinvertebrates.

Bottom substrate was the greatest predictor of fish detection in the sampled reaches. Brook trout were more likely to be detected in streams where a boulder substrate was predominant over finer sand or silt. These findings are consistent with Coggins, Bacheler, and Gwinn (2014) who found that fish species are correlated with a preference for a given bottom substrate. In their analysis, not only the type of substrate, but the presence of biota such as aquatic plants and macroinvertebrates predicted detection of Red Snapper on rock ledges and outcroppings. Similarly, in our study it was found that detection values were higher when large, rocky substrate was present (Fig. 5).

We hypothesized that shredder macroinvertebrates, which process leaves and other coarse particulate organic matter (CPOM) in the water, would be a predictor within our models because the majority of the streams had substantial overhanging canopy (Merritt et al. 2017). Shredder macroinvertebrates break down CPOM into fine particulate organic matter (FPOM), which collector-gatherers primarily collect from substrate along stream bottoms (Merritt et al. 2017). Collector-gatherers tended to be abundant in areas with shredders and greater canopy cover like in LPC1 and LPC2 (Table 1). In the areas with less canopy, collector-gatherers were the dominant feeding type of macroinvertebrates like in GC, perhaps because they can survive off the FPOM that has traveled downstream from areas with higher canopy cover and shredder abundance (Stumpf et al. 2009). The presence and abundance of collector-gatherer macroinvertebrates was a strong indicator of brook trout occupancy. In sites where brook trout were present and collector-gatherers were abundant, all but two of these sites (LPC1 and LPC2) possessed gaps in overhanging vegetation. While areas heavily shaded by vegetation are typically considered to be beneficial habitat features for supporting brook trout, areas with canopy gaps have higher levels of light penetration, which increases water visibility and temperature. Another contributing factor could be vegetation type; many of the stream sites' riparian zones were dominated by rhododendron, the leaves of which are thick and waxy and more difficult to break down than the leaves of a deciduous tree. Therefore, streams heavily shaded with rhododendron may lack the ideal CPOM for collector-gatherer macroinvertebrates. The factors explained above may improve the brook trout's ability to locate prey and potentially increase metabolic efficiency (Nislow and Lowe 2006).

The FBI scores indicated that all the stream sites have excellent or very good water quality, therefore, they cannot be used as indicators for brook trout presence or absence. Further evaluation should be conducted to determine if the water quality of the Upper Tuckasegee headwaters has an effect on brook trout presence.

Based on the results of our study, it would appear that sedimentation and the introduction of fine substrate into streams corresponded to fewer fish detected. This could be due to the decrease in visibility, thus making it harder to physically see and capture the fish, or this could mean the environment cannot physically sustain a larger number of fish, leading to less being detected. Finer sediments and sedimentation tend to correlate with lower numbers of macroinvertebrates, which would not be able to sustain a larger brook trout population. Many of these streams were within a few meters of a gravel road, and given the annual volume of rainfall in Western North Carolina, it is inevitable that the dust and small stones that compose these roads will wash into the stream during periods of precipitation. This change in substrate appears to negatively impact populations of brook trout in the southern Appalachian headwaters. The detection of the fish is directly related to its occupancy and abundance; an increase in sedimentation will impede detection and, indirectly, abundance. Therefore, we recommend that greater attention be paid to the proximity of streams to transportation pathways, be they paved, gravel, or dirt pathways.

There is no panacea to maintaining fluvial system health. Rivers, more than the immediately visible ribbon of blue, are dynamic aggregates of their surrounding environment (Ward and Stanford 1995). Our covariates in this study, while quantified individually, were intimately linked. For example, habitat vegetation influences bank stability, and varying degrees of bank stability determine erosion rates, which influence turbidity and water chemistry levels. Due to this complex interplay of factors, our first recommendation towards land proprietors interested in maintaining suitable aquatic habitat for brook trout is to observe the present health of the river system. Brook trout thrive in cold, clean water. Vegetative buffer zones around the river can mitigate contamination from sediment runoff, the leading source of pollution in western North Carolina. It is important to take note of impermeable infrastructure nearby, including roads and parking lots, as well as logged mountainscapes with loose, easily erodible soil, which further perpetuates sediment runoff into riverways. The root systems in vegetative buffers can also act as bank stabilizers, especially pertinent when river turbidity is excessively high.

Management considerations extend beyond the physical landscape to manipulating the biotic community itself. Non-native fish species, including rainbow trout and brown trout, can outcompete the native brook trout. Their introduction into the water can lead to brook trout being pushed out of further habitat. Also, livestock should be kept from grazing up to the water's edge, as this practice perpetuates erosion and, through these animals' interaction with the water, adds to contamination levels. Instead, we recommend livestock owners fence their animals off from the banks, and provide them drinking troughs.

Ultimately, a private landowner can procure an ideal habitat for brook trout, only for a lack of proper land management upstream to destabilize the system. Thus, we recommend communication and networking between landowners to continue the conversation of land management, and further the discussion of preserving the iconic native brook trout.

Limitations of this study include the number of samples taken. Seventeen sites were sampled for fish, with fourteen sampled for macroinvertebrates. More samples would be necessary in order to refine the estimates. A low sample size results in uncertainty and high variability of results due to decreased statistical power and an increased error margin. Increasing the sample size would also serve to reduce random error.

Other limitations include bias from sampling methods. The macroinvertebrate sampling is designed so that we look for the most likely sites to find macroinvertebrates (seeking out leaf packs, undercut banks, ideal riffle habitat, etc.). Sampling techniques also vary by individual, affecting how samples are taken and how thoroughly the samples are sorted for macroinvertebrates. These could alter the abundances of macroinvertebrate functional feeding groups collected, which we found was a significant factor in brook trout presence. However, given that we had a standardized sampling protocol, the samples were fairly representative and consistent between sites. Similarly, the nature of quantifying qualitative aspects of the habitat forms provides the potential for error due to the variability of personal observations.

Other areas that may account for error include weather/sampling conditions. For example, water depth and turbidity may be dependent on the intensity of precipitation in the near past. Additionally, sampling conditions also include environmental conditions, such as parts of a stream we could not access (due to steep grade and/or thick vegetation). This means that although our sampling methods were systematic in approach, the stream segments that were sampled also relied on convenience. We must also take into account the limitations and failures of equipment. The water chemistry meters tended to be problematic in their calibration, such that some sites required pH strips to be used instead.

Conclusion

We conclude that an abundance of collector-gatherer macroinvertebrates was the greatest indicator for the presence of brook trout in the Upper Tuckasegee watershed. All sites sampled were found to have good or excellent water quality, as indicated by the family-level biotic index. As such, factors that influenced our ability to detect and capture the fish were the substrate composition of the stream bed, the stability of the bank, and the quality of the riparian vegetation. This study can assist in the design of land management practices to preserve the headwater streams for the iconic native brook trout.

Acknowledgements

There are many people whom we would like to thank for their contribution to this project. We thank Jason Meador and Michelle Ruigrok, whose enthusiasm and expertise in aquatic ecology contributed greatly to our success. They gave much of their time helping us collect and process our data. Michelle's incomparable reverse driving astounded us all! We would like to thank the Highlands Biological Station staff for a wonderful semester and for making us feel at home, especially Dr. Costa and Dr. Workman. Their constant support throughout the semester did not go unnoticed nor unappreciated. Finally, we thank the landowners for their permission to sample on their property. This project would not have been possible without their help.

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APPENDIX A. Benthic macroinvertebrate families identified at each site. Creek names are abbreviated (see Table 1).

	HC	LPC1	LPC2	LPC3	UTW	NC	BC	DC	KC	MC	RR	WC	CC	GC
Aeshnidae		X												
Amphipoda									X					
Arachnida		X								X				
Athericidae										X				
Baetidae		X										X		
Baetiscidae	X											X		
Caenidae												X		
Calopterygidae									X		X		X	X
Ceratopogonidae									X					
Chloroperlidae		X		X			X		X			X		X
Chironimidae		X	X			X	X		X		X		X	X
Cordulergastridae			X	X	X		X	X		X	X			
Dryopidae	X													
Dytiscidae				X										
Elmidae		X							X					X
Entomobryidae		X		X								X		
Ephemerellidae		X		X					X	X	X			
Ephemeridae			X							X	X			X
Gastropoda												X		X
Gomphidae		X	X	X		X			X		X			
Heptageniidae	X	X	X	X	X	X	X	X	X	X		X		X
Hydropsychidae	X	X	X	X	X	X	X	X	X	X	X	X		X
Isonychiidae		X		X		X								
Lepidostomatidae														X
Leptophlebiidae			X				X		X	X			X	
Leuctridae		X						X	X					
Limnephilidae	X								X					
Nemouridae												X		
Odontoceridae	X				X		X		X	X				
Peltoperlidae		X	X	X		X	X	X	X		X	X		
Perlidae		X	X	X	X	X	X	X	X	X	X	X		X
Perlodidae			X			X				X	X			
Philopotamidae	X		X		X	X			X	X		X	X	
Phryganiidae													X	
Potamanthidae												X		
Psephenidae										X		X	X	
Pteronarcyidae			X			X				X				
Simuliidae			X							X	X	X		
Siphloneuridae			X	X					X	X	X			
Tabanidae										X				
Tipulidae	X					X	X		X	X				
Totals	8	15	14	12	6	10	10	6	19	18	12	15	6	10

APPENDIX B. Family Biotic Index used to determine water quality and pollution (Hilsenhoff 1988). Refer to Fig. 4.

Family Biotic Index	Water Quality	Degree of Organic Pollution
0.00-3.75	Excellent	Organic pollution unlikely
3.76-4.25	Very good	Possible slight organic pollution
4.26-5.00	Good	Some organic pollution probable
5.01-5.75	Fair	Fairly substantial pollution likely
5.76-6.50	Fairly poor	Substantial pollution likely
6.51-7.25	Poor	Very substantial pollution likely
7.26-10.00	Very poor	Severe organic pollution likely