INSTITUTE FOR THE ENVIRONMENT

HIGHLANDS FIELD SITE

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HIGHLANDS BIOLOGICAL STATION

HIGHLANDS, NORTH CAROLINA

INTRODUCTION

Every class has its own personality, and that of the IE-HFS Class of 2014 is upbeat and "can-do," whether in classroom and lab or river and mountain. They have thrown themselves into their coursework, their research projects, and the community, and with unfailing good cheer have endured — even enjoyed, we suspect — downpours on the trail, hours immersed in icy rivers, hiking in the snow, bushwhacking the suspect terrain of serpentine barrens, even poetry night. There is a reason that the IE Field Sites are administered through Study Abroad at UNC-Chapel Hill: landing in an outpost like Highlands is not so different from landing in any remote and far-flung spot somewhere in the world, except maybe for the trendy boutiques and five-star hotels — but it's no less exotic to the typical college student for all that. At times our students may have felt like strangers in a strange land, and we have great admiration for them — they are brave, after all, to elect to take an immersion program like this, living, working, and playing at close quarters with same small cohort for an entire semester. It might be a bit like Mars One training, or reality TV — and fortunately, to our knowledge, no one has been voted "off the plateau" (though some have come close). Diversity is a wonderful thing in all of its manifestations, and once again we were delighted to have a multi-institutional cohort of students, this year representing UNC-Chapel Hill, Western Carolina University, and UNC-Pembroke. HBS is a founding Field Site of the IE program — we have been the students' "home away from home away from home," where they've spent untold hours in classroom, lab and field — *especially* the field, as we see the field as an extension of the classroom and lab Our students are thus part explorer, part academic — which is what working and studying at a field station is all about. This puts us in a mind about field stations and the legacy that we are heir to.

Agassiz in Highlands

If one thinks of a field station as a base of operations for logistical support of fieldwork in far-flung places, the concept seems as old as the Age of Exploration — base camps, bivouacs, ship's cabins like that of Darwin on the *Beagle* are all bases of operation supporting fieldwork in far-flung places, after all. But the idea of a research lab purpose-built and more or less permanent in such places — *that* idea has only been around for a century and a half. The visionary German zoologist Anton Dohrn is credited with coming up with this idea: Dohrn established the world's first field station in the Bay of Naples in 1872 — the *Stazione Zoologia di Napoli* — and it's still going strong. (The

reason, incidentally, that Dohrn called it a *'Stazione'* [Station] is that he envisioned a string of such institutions like depots along a rail line, established in biologically interesting areas.) That same year, 1872, saw the birth of what may have been the first summer field course in the US — Asa Gray's field botany summer course, offered at Harvard. Curiously enough the success of that course may have led to the establishment of the first US field station: Louis Agassiz, Gray's Harvard rival, was not to be outdone and the following year (1873) he established the Anderson School of Natural History on Penikese Island, in Buzzard's Bay, Massachusetts. Named for its benefactor, Boston businessman John Anderson, Agassiz's Anderson School was intended to do Gray one better: rather than a mere field course, there would be several, and taught in a dedicated field teaching and lab outpost.

In one sense it didn't last: Agassiz died later in the very year the Anderson School was founded, and despite the efforts of Agassiz's son Alexander, who succeeded his father's chair at Harvard, the Anderson School closed after its second season. Yet paradoxically, the school's eventual influence in the US was in inverse proportion to its duration, inspiring a generation of naturalist-educators who experienced the Anderson School to become not only zealous about field-station-based research and education, but in many cases to establish such institutions of their own. David Starr Jordan was one of those: he became a leading ichthyologist and educator, president of Indiana University, founding president of Stanford University, and not incidentally, founding director of the Hopkins Marine Station. (The salamander *Plethondon jordani*, well known to herp students and researchers at HBS, is named for D. S. Jordan to boot.) In his memoirs Jordan wrote about that magical summer in Buzzard's Bay:

> "...the school with the most extended influence on scientific teaching in America
> was held in an old barn on a little offshore island. It lasted only a few months, and
> it had virtually but one teacher..."

Note Jordan's language: this summer field school had the *most extended influence on scientific teaching* in America. How was its influence felt? Quite a few "Anderson School Alums" like Jordan launched a fleet of field stations and marine labs, transforming field biology in US science: Castle Hill Biological Laboratory in Newport, RI; Chesapeake Zoological Laboratory in Maryland and Beaufort, NC; the Marine Biological Laboratory at Woods Hole; the Brooklyn

Institute Biological Laboratory in NY; the Salem Summer School of Biology in Salem, MA; all these and more were founded by students who spent the summer of 1873 with Agassiz at Penikese, applying what became known as the "Agassiz Method" of field-based education. In no small measure, these field stations were reactions against a perceived reductionism of lab-based biology. In the words of William Keith Brooks, an Anderson School alum who eventually became professor at Johns Hopkins University and founder of the Chesapeake Zoological Laboratory:

> "Is not the biological laboratory which leaves out the ocean and the mountains
> and meadows a monstrous absurdity? Was not the greatest scientific
> generalization of [our] times reached independently by two men [Darwin and
> Wallace] who were eminent in their familiarity with living things in their homes?"
> (1899)

The "immersion education" philosophy of HBS and the IE is a direct legacy of Agassiz's Anderson School — we embody the "Agassiz Method" where laboratory and field experience are viewed as the most essential element of a student's education. Curiously, there is a double Coker connection in this regard with respect to HBS and another IE field site.

A Tale of Two Cokers (And Their Field Stations)

One Robert Ervin Coker, cousin of the UNC botanist and second HBS director William Chambers Coker, was a zoologist inspired by the "Agassiz Method" — he studied with Brooks at Johns Hopkins where he earned his PhD, and attended Brooks' Chesapeake Zoological Laboratory outings. In the 1890s R. E. Coker taught summer field courses at Beaufort and by 1899 had established a semi-permanent facility there. Coker was recruited to UNC in 1922, and was instrumental in establishing what grew into the UNC Institute of Marine Sciences at Morehead City, home to the Morehead City Field Site of the IE — housed in Coker Hall, and, like HBS, a founding IE field site.

On our end of the state, in 1928 our founder, Atlantan and conservationist Clark Foreman, heeded the advice of herpetologist Clifford Pope of the American Museum of Natural History in New York that Highlands should establish a dynamic research station in addition to a natural

history museum for public education. (Pope had just returned from working at the Tropical Research Station at Kartabo, Guyana, run by renowned zoologist William Beebe, who founded no fewer than four tropical field stations between 1916 and 1949.) In response to Pope's suggestion Foreman recruited William Chambers Coker of UNC and E. E. Reinke of Vanderbilt to help establish a research lab at Highlands; Coker did not need much persuading, excited about the idea of establishing a field station — as it happens, like his cousin, he too got his PhD at Johns Hopkins and knew Agassiz's protégé Brooks. Reinke became the first director of our forerunner institution, the Highlands Biological Laboratory and Museum, Inc., and W. C. Coker the second. Coker, in fact, donated the very land that the Coker Laboratory at HBS now occupies.

Thus between the Museum and Laboratory at Highlands was born an institution of *research and education*, *discovery and interpretation* here in the southern mountains. Soon graduate and undergraduate students and investigators senior and junior were coming and going, conducting research and taking courses. And here are our IE program students, some generations later... working and studying in another building named for a Coker, at another founding IE field site, studying the natural world up close and personal just the way the Cokers, and Agassiz, would have it. In the tradition of Agassiz field stations like ours advance a certain philosophy of education and research: we're all about place-based teaching, learning, and investigation, which means we're all about getting *out* there, jumping in. From creeks and coves to barrens to balds, in the lab and on the trail, these students have been immersed all right — they've had an ecological and educational baptism of sorts. The internship research projects form the centerpiece of this Highlands baptism, and in this volume we are delighted to present the fruits of their scientific labors.

> ~ *Jim Costa and Karen Kandl*
> *Highlands Field Site Directors*

TABLE OF CONTENTS

Research Papers

THE EFFECTS OF HUMAN ACTIVITY ON ACIDIC COVE FORESTS AND GRANITIC ROCK OUTCROP COMMUNITIES IN THE HIGHLANDS-CASHIERS PLATEAU

WHITLEE ANGEL AND LINDSEY EBAUGH

Abstract. In this study, we compared two granitic dome rock outcrops and two acidic cove forests to determine whether or not human impact plays a role in the plant diversity at these sites. These four sites are owned by the Highlands-Cashiers Land Trust in Highlands, North Carolina. At the two acidic cove forest sites, we surveyed the flora and salamanders, and at the two rock outcrop community sites we surveyed only the flora. We compared the two granitic rock outcrop sites, one with little to no human contact and one that is highly visited each year. Our results showed no distinct trend in plant abundance or diversity between human impacted and non-human impacted sites. We found that the salamander abundance was greater and more species-rich in the undisturbed acidic cove community. These data will be used to create a database of the flora present at these locations and will be monitored for years to come. Further studies should be conducted in order to determine if human presence effects these plant and animal habitats.

Key Words: Acidic cove forest; granitic dome rock outcrop; Highlands Cashiers Land Trust; Highlands-Cashiers Plateau; human impact; salamanders.

INTRODUCTION

The Highlands-Cashiers Land Trust (HCLT) in western North Carolina has been saving land since 1909. They work locally to protect valuable land resources such as rare species, wildlife habitat, scenic view sheds, and public recreation opportunities. While these goals tend to overlap, occasionally they may clash. On some HCLT properties, public access may be impacting the rare species and communities. Our study focuses on four properties owned and protected by the HCLT. Two of these properties – Satulah Mountain and Sunset Rock – are high elevation granitic domes (Schafale 2012). The other two sites – Henry Wright Preserve and Kelsey Trail – are acidic cove forest communities (Schafale 2012). Sunset Rock and Kelsey Trail served as our highly public-impacted locations, compared to Satulah Mountain and the Henry Wright Preserve which are more remote, less public-impacted areas. Understanding the impacts of public access on these properties can help the HCLT better manage its properties.

Sunset Rock was attained by the HCLT and opened to the public in 1914. For the past 100 years, the property has seen increasingly heavy use by the public, with upwards of 16,000 annual visitors to the property in the past decade to enjoy the scenery and beautiful sunsets. In contrast, parts of Satulah Mountain, which has been protected by the HCLT since 1909, are more secluded and less disturbed. Our site at Satulah Mountain is not open to the public, and therefore only receives a handful of visitors each year from people wandering off nearby trails. Both sites are considered to be high elevation granitic dome communities, an unusual natural community type that is home to a number of rare and endemic species for the area (Wiser et al. 1996). The HCLT attained its first section of the Kelsey Trial in 1998, however part of the property was historically used as a cart road and hiking trail beginning in the 1890's, and part of the property was used as a small pasture and pond in the early-mid 1900's. The Henry Wright Preserve has been protected since 1964, and contains one of the last intact pieces of primeval forest that has remained largely undisturbed from human development.

The Highlands-Cashiers Land Trust has no specific data on the flora or fauna at these four sites. In this study, we compared the two granitic dome rock outcrops and the two acidic

cove forests to determine whether or not human impact plays a role in the plant diversity at these sites. In addition to a survey of plant diversity, we also studied salamander species richness and abundance within the two acidic cove forest sites. We looked primarily at aquatic salamander species and noted the incidental individuals found outside of our plots. This study will be used to create a database of plant and animal species to monitor trends in abundance for future years.

<p style="text-align:center">METHODS</p>

<p style="text-align:center">Study Sites</p>

We surveyed each of these four Highlands-Cashiers Land Trust properties in the fall of 2014. Our study area at the Henry Wright Preserve started with plot one at the top of a slope near the parking lot of an undisclosed site (kept secret to protect the site locality). Plot two was located halfway down the slope, and plot three was at the bottom of the slope near the stream. The Henry Wright Preserve is 22 acres total. At Kelsey Trail we did a similar study where our three plots followed the trail from the parking lot to the stream. The main difference is that Kelsey Trail is relatively flat compared to the Henry Wright Preserve. The GPS coordinates for those plots are as follows: plot one (N35.08075, W083.17827), plot two (N35.05964, W083.19145), and plot three (N35.06057, W083.18968). The Kelsey Trail property is 12.96 acres total.

Our study area at Satulah Mountain (N35.03532, W083.19211) is off of the main pathway and inaccessible to visitors. It contains a small open outcrop area surrounded by dense vegetation. The Satulah Mountain property is 66.8 acres total. Sunset Rock (N35.31224, W083.11582.) is widely visited, resulting in a much more open area. For our survey, we chose the sparsely vegetated area near the center of the outcrop located halfway from the top with the coordinates The Sunset Rock property is 10 acres total.

<p style="text-align:center">Acidic Cove Forests</p>

At each site, we collected and recorded different species of flora. At both the Henry Wright Preserve and Kelsey Trail, we created three plots, 10 x 10 m each. One plot was at the beginning of the trail, one plot was toward the middle of the trail, and one plot was at the end of the trail. In each of the three plots, we surveyed the plant species by looking at ground cover, midstory cover, and canopy cover. Because the properties are so large, we decided to make three plots in order to create a representation of the flora present in the entire site. This gave us six total plots to survey for plants stem abundance and species abundance between the two acidic cove sites.

At the end of the trail in both the Henry Wright Preserve and Kelsey Trail, there are creeks where we surveyed for salamanders. At each of the two creeks, we created three 5 x 5 m plots adjacent to each other, for a total of six plots between the two acidic cove sites. We spent 15 minutes per plot looking for salamanders under leaf detritus, moss, logs, and rocks. We identified and recorded the number and species of each individual found. We included any salamander species found outside of our plot and counted them as incidentals to have a better representation of the salamander diversity present at both sites.

Granitic Dome Outcrop

In the granitic dome outcrop sites, Sunset Rock and Satulah Mountain, we made a 74.5 ft transect across the rock and counted everything we saw from the top of the outcrop to as far down the rock face as we could see. We recorded every plant species and the number of stems surrounding our transect and their relative abundances.

At all four sites, we included incidentals of animals we heard, found scat of, found tracks of, or saw outside of our designated plots. We included these as incidentals because although they were not in a plot, they were present at the site and should be included in our overall results.

RESULTS

Acidic Cove Forests

At the Henry Wright Preserve we found a total of 17 plant species in our three plots and 12 plant species in our plots at Kelsey Trail (Table 1, Fig. 1). At the Henry Wright Preserve we found a total of nine American chestnut (*Castanea dentata*) tree stems, 14 blackgum (*Nyssa sylvatica*) tree stems, and no dog hobble (*Leucothoe fontanesiana*) stems within our plots. At Kelsey Trail we found no American chestnut, only one blackgum tree, and 34 dog hobble stems. Rosebay rhododendron (*Rhododendron maximum*) was the most abundant species (by stem) at each site but was almost five times more abundant at Kelsey Trail than at the Henry Wright Preserve (Table 1). These are a few examples of the major differences in plant diversity between the two sites. The total plant stem abundance observed in our plots was greater at Kelsey Trail with 100 more plants stems than at the Henry Wright Preserve.

FIG 1. Species diversity at each of the four Highlands Cashiers Land Trust sites studied.

Salamanders

At the Henry Wright Preserve we found four salamander species throughout our three plots and found one species outside of our plots. We observed a total of five different species of

3

salamanders (Table 2). At Kelsey Trail we found two salamander species within our plots and found two outside of our plots for a total of four different species of salamanders. One major difference

TABLE 1. Acidic cove plant stem abundance between the Henry Wright Preserve and the Kelsey Trail.

Plant Species	Henry Wright Preserve	Kelsey Trail
Acer pensylvanicum	0	1
Acer rubrum	6	4
Amelanchier laevis	1	0
Betula lenta	0	3
Castanea dentata	9	0
Fraxinus americana	2	0
Hamamelis virginiana	11	2
Kalmia latifolia	3	1
Leucothoe fontanesiana	0	34
Liriodendron tulipifera	6	0
Nyssa sylvatica	14	1
Pinus sroba	5	0
Prunus serotina	1	0
Quercus alba	0	1
Quercus prinus	3	1
Quercus rubra	2	1
Rhododendron calendulaceum	2	0
Rhododendron maximum	32	157
Smilax	7	0
Tsuga canadensis	0	3
Tsuga caroliniana	3	0
Vaccinium stamineum	2	0
Total	109	209

between the sites is the abundance of seal salamanders (*Desmognathus monticola*) at the Henry Wright Preserve and ocoee salamanders (*Desmognathus ocoee*) at Kelsey Trail (Table 2). We found seven gray-cheek salamanders (*Plethodon metcalfi*) at the Henry Wright Preserve that were not in our plots and counted these as our incidentals. At Kelsey Trail we found a red-backed salamander (*Plethodon serratus*) and a black-bellied salamander (*Desmognathus quadramaculatus*) that were not in our plots and counted these as our incidentals. This makes the species diversity at both sites similar with five species at Henry Wright and four species at Kelsey Trail. The total salamander abundance was greater at the Henry Wright Preserve (Table 2).

TABLE 2. Salamander species and abundance in our stream plots at our two acidic cove sites.

Species	Number of individuals at Henry Wright	Number of individuals at Kelsey Trail
D. monticola	5	1
D. monticola (juvenile)	4	1
D. ocoee	0	7
D. ocoee (juvenile)	0	1
E. wilderae	1	0
G. porphyriticus	1	0
unidentified	6	0
Total	18	10

Granitic Dome Outcrop

At Satulah Mountain we found a total of 13 plant species in our plot and 16 plant species at Sunset Rock (Table 3, Fig. 1). At Satulah Mountain we found 54 mountain laurel (*Kalmia latifolia)* stems, three service berry (*Amelanchier laevis*) trees, nine white oak (*Quercus alba*) trees, and one white pine (*Pinus strobus*) tree. At Sunset Rock we found three mountain laurel stems, 12 service berry trees, one white oak tree, and eight white pine trees. Like the acidic cove forest sites, there are occasions when a species is only found at one site at a small abundance and never found at the other site. We also anecdotally noticed that the amount of bare rock showing appeared to be much greater at Sunset Rock where more people visit than at Satulah Mountain, which had more lichen and mosses covering the rock. The total plant abundance was greater at Satulah Mountain with 91 stems being observed over 49 at Sunset Rock (Table 3).

TABLE 3. Granitic dome outcrop plant comparison between Satulah Mountain site and Sunset Rock site.

Plant Species	Satulah	Sunset
Acer rubrum	1	4
Amelanchier laevis	3	12
Cyperaceae	0	1
Galax urceolata	1	0
Hamamelis virginiana	2	0
Kalmia latifolia	54	3
Liriodendron tulipifera	0	2
Pinus rigida	1	5
Pinus strobus	1	8
Polygonatum	0	2
Quercus alba	9	1
Quercus prinus	0	1
Quercus rubra	4	1
Rhododendron Maximum	1	0
Rubus fruticosus	3	1
Solidago	5	4
Tsuga caroliniana	0	2
Vaccinium	6	1
Vine (common name)	0	1
Total	91	49

Other Fauna

At Henry Wright Preserve, we found various signs of fauna about which we took note. We saw black bear (*Ursus americanus*) scat and paw prints, we heard an eastern screech owl (*Megascops asio*) and we heard a great horned owl (*Bubo virginianus*). At Kelsey Trail, we also saw black bear scat. At Satulah Mountain, we found signs of a black bear along the trail to our site and we found coyote (*Canis latrans*) scat at our site. At Sunset Rock, we found black bear scat adjacent to our plot.

Acidic Cove

At the Henry Wright Preserve we found 17 plant species with 109 individuals and at Kelsey Trail, we found 12 plant species with 209 individuals. This suggests that Kelsey Trail has higher plant abundance and fewer plant species than at the Henry Wright Preserve. The higher abundance may be a direct cause of the lower number of species. At Kelsey Trail, we found the majority of plants to be rosebay rhododendron (*R. maximum)* and dog hobble (*L. fontanesiana*), both of which create a dense cover of understory plants and are clonal in that one plant may have multiple above ground stems come up from the rhizome below ground (Kroons and Hutchings 1995). The clonal nature of these species allows them to form very dense thickets, which can shade out many plant species that require a minimum amount of sunlight. The presence of these two plants is likely a reason for the low species richness observed.

At Henry Wright, we observed less plant stem abundance than at Kelsey Trail but higher species richness. The greater species diversity at the Henry Wright Preserve may result in higher competition between species. This could explain the lower plant abundance as the individual species partition the available resources.

The presence of rosebay rhododendron and dog hobble might be related to the frequent use of this trail. Humans may inadvertently trample small saplings along the trail as they walk. Also in the past, plant species may have been cut down for the construction of the cart trail and rosebay rhododendron and dog hobble could have been the first to colonize the area. Both species are also vigorous growers, are very common in the area, and do well in disturbed areas.

Salamanders

The salamander abundance at the Henry Wright Preserve stream was not only larger than the abundance found at the Kelsey Trail stream, but it was also more species rich. According to Pechmann (1994), human impact affects stream salamanders. This may explain why the stream at the Henry Wright Preserve has a higher species richness than the stream at Kelsey Trail. While it appears that higher salamander diversity is present at the more remote, less human-impacted stream, we cannot say for certain that there is a direct correlation. There are many factors, abiotic and biotic, that affect salamander population richness and evenness in streams (Harper and Guynn 1999). It is inaccurate to assume that one element, human disturbance, is the underlying factor.

When incidental salamander findings were added to our stream plot findings, we saw a different pattern. At Henry Wright Preserve, we found five species with 25 total individuals. At Kelsey Trail, we found four species with 12 total individuals. This shows a higher abundance of salamanders at the Henry Wright Preserve than at Kelsey Trail, but a similar species richness. The higher species abundance at the Henry Wright Preserve may be the result of a more desirable forest community because there are more habitats available for salamanders. Mackey (2012) found salamander abundance was highest in the less disturbed sections of streams in area golf courses. In addition, the variation in slope at the Henry Right Preserve likely impacts the abundance found at the site. With similar species richness at the two sites, it might appear that the Henry Wright Preserve and Kelsey Trail have similar diversity. However, of the four species found at Kelsey Trail, 75% of the individuals were *Desmognathus ocoee*. This shows a low

species evenness, which is a major component of species diversity. It is important to keep monitoring each site over time to observe any changes in salamander abundance.

Granitic Dome Outcrop

At Satulah Mountain, there were 13 plant species and 91 individual stems while at Sunset Rock, there were 16 plant species and 49 individual stems. Our Satulah Mountain plot had higher plant stem abundance but a slightly lower species richness than our plot at Sunset Rock. This may be the result of the remoteness of the Satulah Mountain site relative to Sunset Rock, which has a history of heavy public use and impact. Satulah Mountain has almost twice as many individual stems as Sunset Rock, but a slightly smaller number of observed species. The high plant stem abundance is likely due to the lack of human disturbance. With little to no human disturbance, the plants at Satulah Mountain are not being trampled on or interfered with, which maintains a relatively stable plant environment.

At Sunset Rock, there are fewer individual stems, which results in a more open canopy. The open canopy could allow many smaller plant species to flourish with the direct sunlight (Chazdon 1988). In addition, human disturbance can alter what species are found in any given spot and lead to a high species turnover. This creates an environment where plant species are constantly fluctuating in species type and species abundance (Francl 2000).

The number of species found at the two sites is similar, which suggests that while the stem abundance is impacted, the diversity is less impacted. However, this may be an artifact that the higher disturbance at Sunset Rock allows for other, more disturbance tolerant species to colonize. This is verified in part since some of the species present at Sunset Rock but not Satulah Mountain, such as tulip poplar (*Liriodendron tulipifera*), tend to be associated with disturbed sites (Kota et al. 2007). Long-term data collecting at these sites can look into species turnover rates, which can tell us more about the potential impacts public activity has on the species diversity and colonization of these two sites. Granitic dome outcrop communities house rare and fragile plant species, therefore the presence of human activity can greatly effect these communities (Bolas 2006).

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AN ASSESSMENT OF THE HYBRIDIZATION OF *CASTANEA DENTATA* AS A METHOD OF RESISTANCE TO *CRYPHONECTRIA PARASITICA*

Mariah Goodman

Abstract. The American chestnut tree (*Castanea dentata*) was once a keystone species for many environments throughout North America. The introduction of the chestnut blight fungus (*Cryphonectria parasitica*) in the early 20th century managed to wipe out the majority of chestnut trees throughout North America. I participated in a long-running project with the United States Forest Service, which uses modern hybridization methods in an attempt to produce a blight resistant American chestnut tree with primarily blight-resistant genes. I assessed six plots of American, Chinese, and hybrid chestnut trees for the presence of blight in order to determine the current success rate of hybridization. I found that while Chinese chestnuts still had the highest survival rates, the hybrid trees showed significant resistance to the blight fungus.
    Key words: American chestnut; Castanea dentata; *chestnut blight fungus;* Cryphonectria parasitica; *hybridization; southern Appalachians*

Introduction

The American chestnut tree (*Castanea dentata*) was a vital component of many American landscapes up until the early 20th century (Frienkel 2007). These trees played an essential role in both the ecology of their natural habitat, and in the lives of early settlers, particularly those of the southern Appalachian region. In some regions, up to one of every four trees was an American chestnut (Frienkel 2007). Their trunks were recorded to have reached diameters of over nine feet, and the namesake fruit was said to carpet the forest floor at harvest time (Brown 2000). These nuts were an important food source for humans and native wildlife alike. The wood of these trees was also vital to the early American lumber industry, and was very often used in the construction of the homes and furniture of the people living in their native range (Brown 2000). Unfortunately, this all radically changed within the span of just 30 years, with the introduction of *Cryphonectria parasitica*, the chestnut blight fungus (Frienkel 2007).

Although the exact year of the introduction of *Cryphonectria* (chestnut blight) is unknown, it is suspected that it arrived in the United States on a variety of Japanese chestnut in the early 1900's. The native *Castanea dentata* had little to no natural resistance to this fungal invader, and the majority of the American chestnut population was quickly decimated. By the early 1940's, next to none of the once-grandiose chestnut trees remained standing (Frienkel 2007). In modern times, chestnut stumps remain and consistently re-sprout, but these sprouts can only survive for so long before succumbing to *Cryphonectria*. (Clark et al. 2014).

Although there have been numerous efforts to restore the American chestnut to its former glory, none have succeeded thus far. Scientific efforts in this field are most often funded by The American Chestnut Foundation (TACF), a nonprofit group based out of Asheville, North Carolina. These projects range from genetic to environmental methods of blight control. For example, scientists with the New York College of Environmental Science and Forestry have been working on developing a fungus-resistant American chestnut using transgenic technology (Wines 2014). The American Chestnut Foundation has also been working with the United States Forest Service (USFS) on the project from which my study is derived. Over the past decade, the USFS has been working in conjunction with TACF in order to genetically develop an American-Chinese chestnut hybrid. While *C. dentata* (the American chestnut) is incredibly sensitive to blight, *C. mollossoma* (the Chinese chestnut) shows a high level of resistance and survival in the

presence of *Cryphonectria*. The overarching goal of TACF's project is to develop a Chinese-American hybrid strain with as much American chestnut genetic material as possible, yet exhibits the same level of resilience as the Chinese chestnut (Clark et al. 2014). Our hope is that these trees can eventually be used to repopulate some USFS lands with American chestnut in order to restore them to their former magnificence.

During the course of this study, we assessed six separate plots of hybrid, Chinese, and American plantings on USFS land ranging from western North Carolina to eastern Tennessee. The trees within these plots were planted in 2009, 2010, and 2011, and we assessed each year as its own independent set of data. Our ultimate goal is to gain an understanding of the extent of blight resistance in current third generation third backcross (B3F3) hybridized trees, especially in comparison to American and Chinese controls planted within the same plots.

METHODS

The first step in developing a blight-resistant American chestnut hybrid was conducted by the American Chestnut Foundation in the early 2000's. Chinese and American trees were first crossed to produce an F1 generation, which was then selectively backcrossed with American chestnut trees in order to boost American genetic material within these hybrids. After three backcross events, the trees were then intercrossed twice more with themselves in order to produce a hopefully blight-resistant B3F3 generation, which is 15/16ths American chestnut (TACF 2014; Fig. 1).
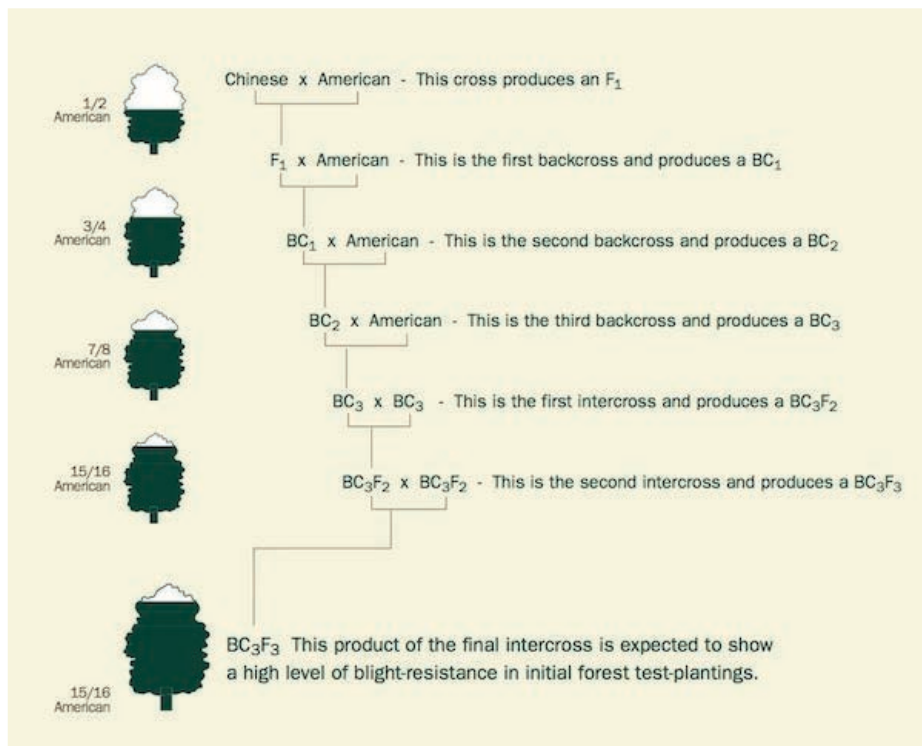

FIG 1. The production of the B3F3 generation.

Many separate B3F3 families were generated by this method, each derived from a separate American "mother tree", and were labeled in the format of D1 through D20. However, many sites contained a limited number of hybrid families. For example, the 2009 planting known

as TRD, located at Eagle Fork in North Carolina, had only D2, D4, and D5 B3F3 families.

We sampled six sites derived from three different years of planting: 2009, 2010, and 2011. For the sake of age consistency, we grouped the data from these sites into three categories by the year in which they were planted. The site names, locations, year of planting, and number of planted trees are outlined in Table 1.

TABLE 1. Summary of sites used.

Site Name	Location	Year Planted	No. Planted Sites
TRD, Eagle Fork	Clay, NC	2009	444
WRD, Iron Mountain	Unicoi, TN	2009	444
NRD; Allen Branch	Cocke, TN	2010	525
WRD Midstory; Wagner Branch	Carter, TN	2011	289
WRD Shelterwood; George Creek	Carter, TN	2011	304
CHRD Shelterwood; Locust Cove	Graham, NC	2011	302

Each plot followed a generally consistent layout of planting. Within a plot, trees were planted in a grid-like fashion, with roughly 12 feet of space between each tree in a row. Each plot contained a randomized mix of Chinese chestnut, multiple American chestnut families, multiple B3F3 families, and a number of intermediate hybrids. The intermediate hybrids included strains such as B2F3, B3F2, and B1F3. These intermediate hybrid trees were ultimately not included in our results because their survival is much less relevant to our goal.

We sampled these sites in a fairly uniform manner, in which we systematically assessed a single tree for the presence or absence of blight, and marked it on a datasheet. Many of the trees we sampled had blight in the previous year's observations, which was already noted on our datasheets. If a blight canker was found on a tree, we assigned a "canker rating" to the affected area. A "2" denotes that a canker is present on a tree, but is primarily on superficial tissue, while a "3" indicates that a virulent canker is present, which typically means that there is notable sunken or swollen tissue, and many visible sporulating fruiting bodies of *Cryphonectria* present. If a tree was found to have no blight present, or was dead but from factors other than blight, we ranked it as a "1". If we found a tree that had been killed due to blight, we ranked it as a "4".

If a tree was found to have living blight on living tissue, a number of other actions were set into place. First, we measured the height of the bottom and top segments of the canker. We then observed the circumference of the infected area, and noted the percentage of the trunk which was circumscribed. We then looked for the presence of the bright orange fruiting bodies of *Cryphonectria* and made note of their proliferation. If a tree was an American chestnut, Chinese chestnut, or B3F3 with blight, we took a sample of the canker present using a bark punch. We sent these samples to Dr. Richard Baird (Mississippi State University) to be plated out to confirm the presence of *Cryphonectria*.

We categorized each of the three years' B3F3, American chestnut, and Chinese chestnut plantings into four survival classes; alive with no blight, alive with a superficial canker, alive with a virulent canker, dead from blight, or dead from other factors.

RESULTS

Overall, there was a distinctly higher number of B3F3's planted in each site across each age group, with various levels of survival success as compered to American and Chinese chestnut trees. The B3F3's recorded in the 2009 plantings showed a roughly 10% greater

survival (with no blight) rate than American or Chinese chestnuts (Table 2, Fig. 2).

TABLE 2. Total health statistics of trees found in 2009 plantings- TRD (Eagle Fork), WRD (Iron Mtn)

	Alive	Alive w/ superficial blight	Alive w/ virulent blight	Dead- blight	Dead-other	Total
B3F3	148	23	16	4	55	246
American	80	8	30	8	28	154
Chinese	65	7	2	0	49	123
Total	293	38	48	12	132	523
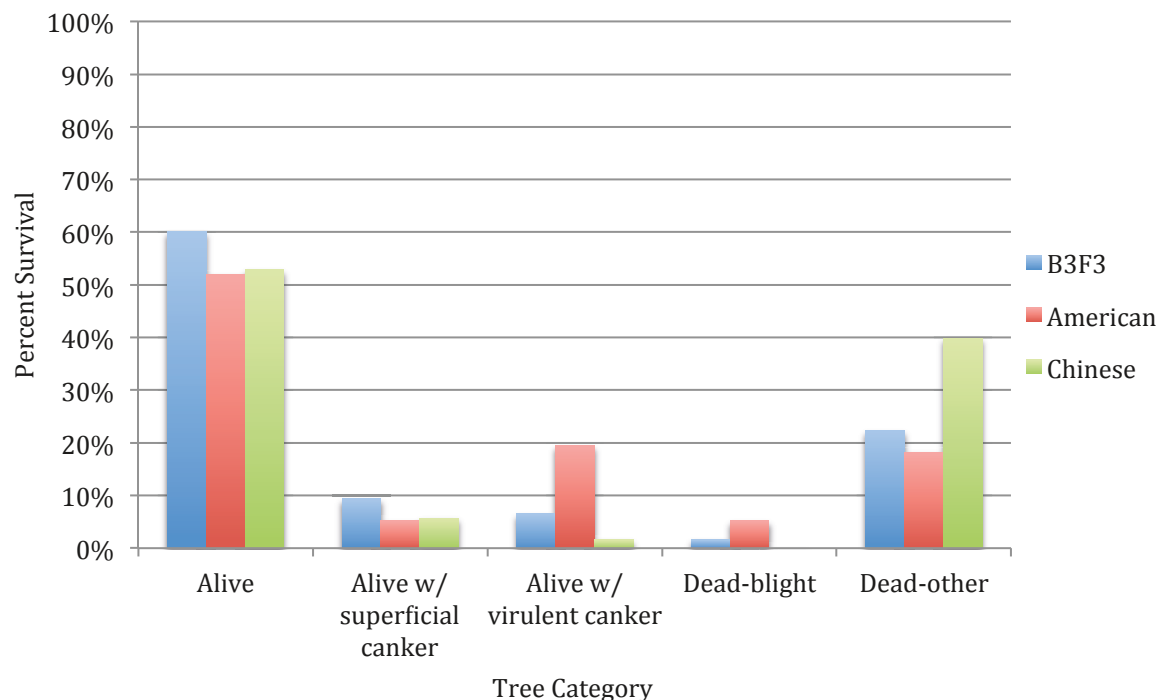
Fig 2. Percent survival rates of American chestnut, Chinese chestnut, and hybrid trees in 2009 plantings.

In the 2010 planting, Chinese chestnuts were found to have a very high rate of survival, while B3F3's still showed lower die-off rates than American chestnuts due to blight alone. American trees also exhibited higher incidences of living blight infection within this site (Table 3, Fig. 3).

TABLE 3. Total health stats of trees found in 2010 planting- NRD (Allen Branch).

	Alive	Alive w/ superficial canker	Alive w/ virulent canker	Dead- blight	Dead- other	Total
B3F3	59	4	8	5	81	157
American	41	4	10	9	50	114
Chinese	43	0	0	0	1	44
Total	143	8	18	14	132	315

The plantings observed in the 2011 sites repeated the general trend found across all the others. American trees generally exhibited the greatest rates of infection and death from blight, while Chinese chestnut showed the lowest rate. Hybrid trees in these plots in particular had low blight incidences (Table 4, Fig. 4).
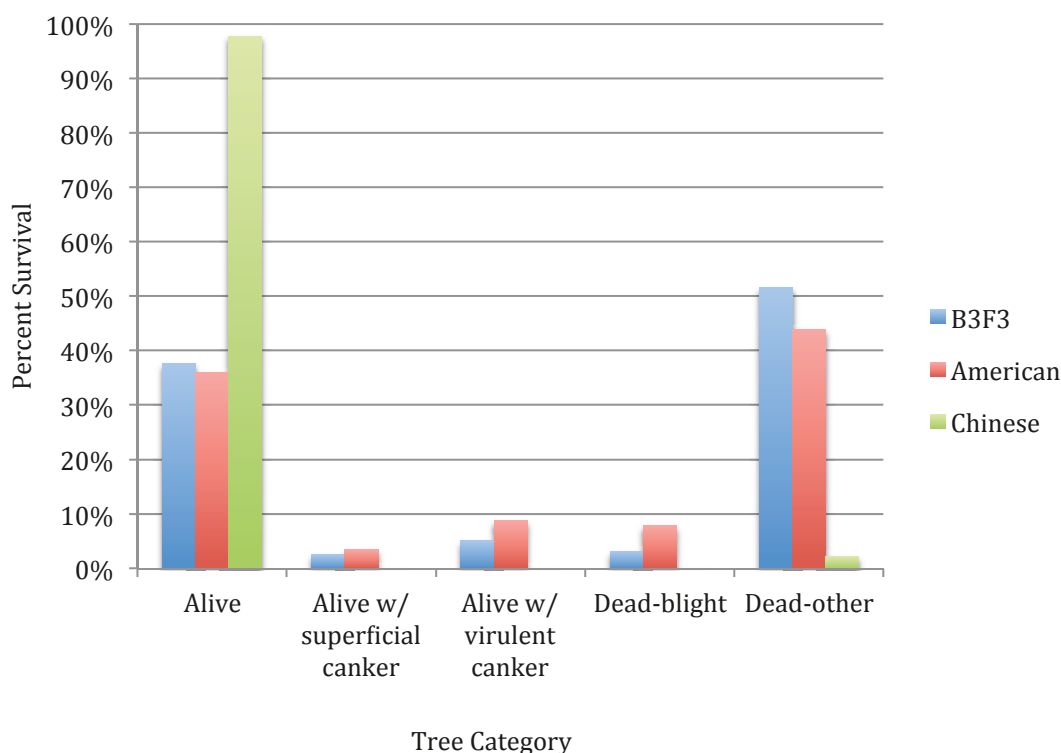
Fɪɢ 3. Percent survival rates of American chestnut, Chinese chestnut, and hybrid trees in 2010 plantings.

Tᴀʙʟᴇ 4. Total health stats of trees found in 2011 plantings- WRD-Mid (Wagner Branch),WRD-Shelt (George Creek), CHRD (Locust Cove).

	Alive	Alive w/ superficial canker	Alive w/ virulent canker	Dead- blight	Dead- other	Total
B3F3	332	18	12	16	183	561
American	50	7	11	18	52	138
Chinese	66	5	2	1	27	101
Total	448	30	25	35	262	800

The plantings observed in the 2011 sites repeated the general trend found across all the others. American trees generally exhibited the greatest rates of infection and death from blight, while Chinese chestnut showed the lowest. Hybrid trees in these plots in particular had low blight incidences (Table 4, Fig. 4).

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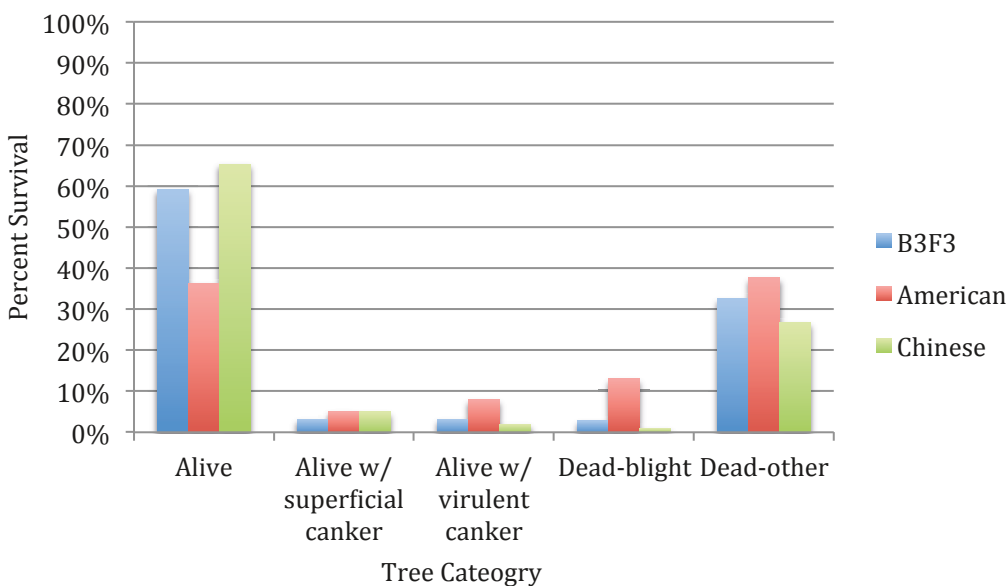
F<small>IG</small> 4. Percent survival rates of American chestnut, Chinese chestnut, and hybrid trees in 2011 plantings.

D<small>ISCUSSION</small>

These results are consistent with the intended goal of The American Chestnut Foundation and the United States Forest Service's chestnut hybridization project thus far. As previous research indicates, our Chinese chestnut trees showed the greatest overall resistance to blight, with very few incidences of infection in the 2009 and 2011 plantings, and none whatsoever in the 2010 planting. To compliment this, American trees have been showing the greatest rates of blight infection across the board, as would be suggested by the plant's unfortunate history. The B3F3 hybrids have varying success rates, but have fared better than the American chestnuts overall.

When considering these results, it is important to factor in the environmental variability of the sites. Some sites resulted in variable growth success for each respective type of chestnut, which is very well demonstrated by the incredibly high survival rate of Chinese chestnut in the 2010 plot. Factors such as temperature, sunlight availability and moisture are all very important when considering the growth of *Cryphonectria parasitica*, and it will likely be important to look more closely at these factors in future studies. It should also be noted that some sites were affected by factors entirely outside of our influence, such as a large amount of deer rub in one of the 2011 plots.

Although this study represents a small portion of a much more far-reaching, long term project, my observations still bode well for the future of the American chestnut. They indicate that this form of hybridization will likely be a viable option for reforestation. Nonetheless, extended observation of these and the other plots will further indicate the validity of my claims here.

A<small>CKNOWLEDGEMENTS</small>

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FLORA HABITAT DEVELOPENT IN LANDSLIDES AND THEIR ROLE IN THE DEVELOPMENT OF EARLY SUCCESSTIONAL FOREST

Jordan Hessler

Abstact. Landslides have the force to change forest ecosystems and alter the geology of the Southern Appalachian Mountains. This destructive force can rewind forest succession from late succession to early succession (Geertsema and Pojar 2007). This study on the effect of landslides on the development of early successional forest was conducted by examining flora species richness in plots in landslide areas. The objective of the study was to determine if landslides create early successional forest habitat. I found a correlation between early successional flora taxa and early successional habitat. Early successional species were found at relatively low frequencies in landslide areas so it was not possible to determine if landslides created early successional habitat.

Key Words: early successional forest, Landslides, Southern Appalachian Mountains

INTRODUCTION

Landslides are natural disturbances that play an ecological role in maintaining biological diversity. Landslides can affect diversity within the landslide by altering moisture, inundating the system with mud or stone, increasing or decreasing the perviousness of the soil, and increasing exposure to climate conditions. These conditions provide substrates that turn back the ecological clock due to the exposure of unweathered rock that leads to primary and secondary succession (Geertsema and Pojar 2007).

Western North Carolina is prone to landslides because of the steep mountain slopes and the track of cyclone events that hit the coast and work their way into this area after landfall. Western North Carolina can be hit from tropical cyclone events coming from the Gulf of Mexico and the Atlantic Ocean which can drop massive amounts of precipitation on the side of mountains causing them to slide. This area is the second most hit area in the United States when it comes to cyclone events and explains why slides are abundant in Western North Carolina (Greenberg et al. 1997).

There are three types of slides that commonly disturb areas in Macon County, North Carolina. The first type is a rockslide that consists entirely of large and small rocks or boulders. The second type called an earth slide and this contains only soils. The third type of landslide is called a debris flow, and these contain rocks, soils, and trees. All slides are not equal, as some slides slowly move down the hill over a period of time, while other slides require a high precipitation event and begin to slide down a steep slope at a high velocity destroying everything in their paths. These sliding events can change plant communities over night because of their destructive force, leveling hundred year old trees in seconds and removing the understory vegetation (Fuhrmann et al. 2008). The slides can leave behind large boulders and completely strip the land of soils making it hard for vegetation to establish in the affected area. The affected area may revert back to an early successional forest.

Early successional forest habitats are becoming rare due to the minimal amount of natural events that lead to their development in forest ecosystems in the southern Appalachians (Hull and Scott 1982). In addition, farmland that was once open and provided for these early successional forests is now filling in with large woody tree species rather than herbaceous flora and small shrubs. Landslides are important for generating early successional forest in the southern Appalachians.

There are just two studies that have examined plant succession on landslides in the southern Appalachian Mountains despite the high number of siding events. Hull and Scott (1982) found that the adjacent forest composition is a major factor contributing to the rate of recovery and species composition in the landslide. They also found that individual plant species (e.g. R. americanus) were important to a recovering slide in Nelson County, Virginia. Elliot et al. (1997) looked at the re-vegetation of a clearcut watershed in the southern Appalachian Mountains and found the woody species diversity remained constant while the abundance of opportunistic woody species abundances increased after the clearcut. The herbaceous species biomass of the watershed decreased throughout their study. The results of the study showed the clearcut area regenerated as forest intermediate to early and late successional forest.

The objective of this study was to make observations on the composition of the flora in landslides in the southern Appalachians. I hypothesized that the flora in the landslide should resemble the composition of the flora found in typical early successional forest since soil conditions on the landslide would resemble soil conditions typical of an early successional forest.
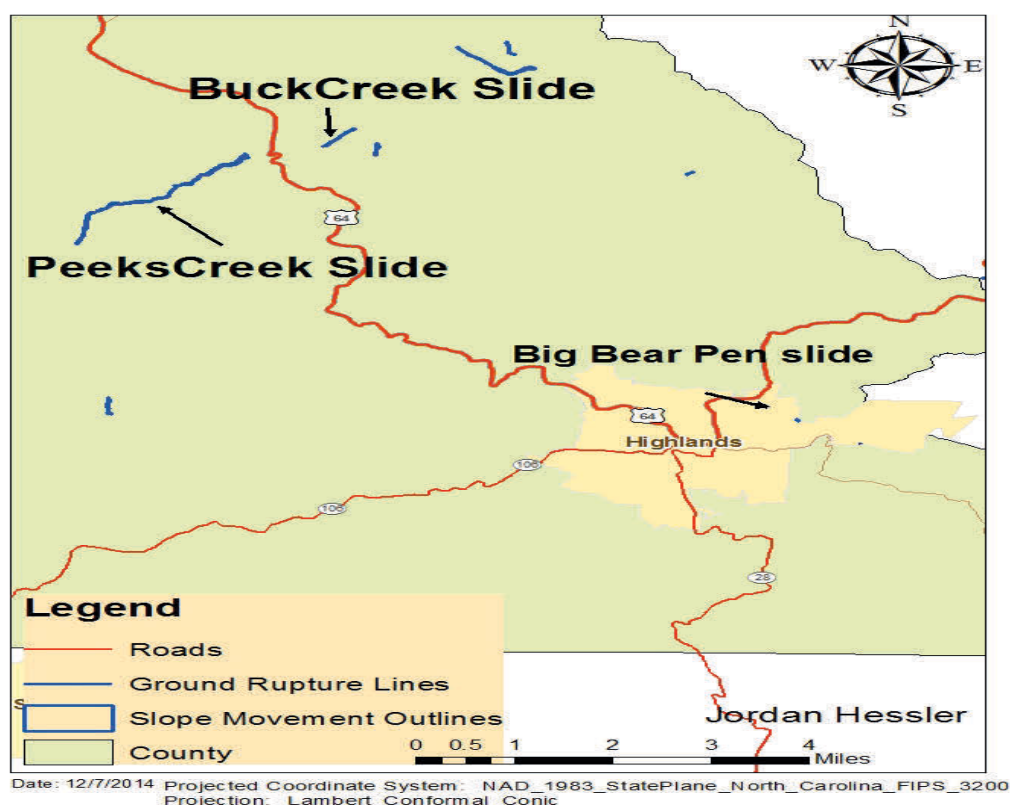
FIG. 1 Map of eastern Macon County, showing sampled landslides.

METHODS

I sampled three landslides in Macon County, North Carolina during the fall of 2014. These landslides were the same age and easy to access, and were named for the roads that were closest to them. Peeks Creek slide was a debris flow and was the most destructive of the three, Big Bear Pen slide is an earth slide and covered the least amount of area. Buck Creek slide was

intermediate in size and was a debris flow (Fig. 1). All three slides independent in terms of slide area and type of slide material.

Dr. Rick Wooten, a geologist working for the North Carolina Department of Environment and Natural Resources, provided satellite imagery which was loaded into ArcGIS (2012) with layers for slide area and ground movement. Additionally, I used layers for Macon County data, provided by Dr. Gary Wein of the Highland Cashiers Land Trust.

The area over which a slide travels made sampling the whole slide impossible. For each slide in the accessible areas I placed plots at the top, middle, and bottom of the accessible areas in the slide. Buck Creek and Big Bear Pen slides were completely accessible, and I placed Peeks Creek Top, Middle, and Bottom plot areas in the accessible area. I set up the plots by running three transects that were 20 meters apart horizontally and ten meters apart vertically. This resulted in 6 plots, for which 10 sq. meters of each plot were in the slide area and ten sq. meters were in an undisturbed area next to the slide; these are the large plots. The large plot area was further broken down into 12 micro plots by running a line vertically down each transect at five, ten, and 15 meters. This produced six micro plots inside the slide and six micro plots outside the slide. The micro plots were assigned a random number, using a random number generator, to indicate each plot as woody or herbaceous. Thus, each of the six micro plots in and outside the slide contained three woody plots, and three herbaceous plots. Flora in each plot was identified and its presence was recorded. If a species could not be identified in the field, a sample or picture was taken to later identify the plant species. The grass species could not be identified so they were labeled grass 1, grass 2, grass 3, and grass 4.

Data Analysis

I analyzed the data using MDS (Nonmetric Multidimensional Scaling), which scales plots in graphical space based on similarity of species composition. Plants significantly associated with each individual slide were identified by indicator analysis based on relative importance (frequency) in each site. The input for the MDS was only for flora species found within all three slide areas and excludes the flora found outside the slide areas.

RESULTS

Non-Metric Multidimensional Scaling

Nonmetric Multidimensional Scaling produces a graph of the relationship of flora taxa in each landslide in graphical space. The ordination for Fig. 2 represents herbaceous species in a visual matrix. This ordination shows in landslides 2 and 3 are more related to each other in terms of herbaceous species than they are related to slide 1 (Fig 2). The ordination also shows two outliers in the data set in landslide 3. The Jaccard Index of stress on the ordination for herbaceous flora shows a final stress of 22.50 which suggests that this ordination represents the data in a reasonable way. The "woody in" ordination shows that the three landslides had species in common as well as species of flora that are representative of that particular slide (Fig 3). The "woody in" ordination has one outlier in the data on landslide 2. The Jaccard Index of stress on the ordination for woody flora shows a final stress of 22.82, and again this suggests that these areas are reasonably presented by the ordination.

Several individual species were particularly important to the three landslides (Table 1). For example, Grass 1, *Solidago* spp., and two species of *Quercus* were particularly important in landslide 1.

Table 1. Significant *p* values from the "herb in" and 'woody in" MDS. Slide 1 is Buck Creek, slide two is Big Bear Pen, and slide three is Peeks Creek slide.

Species	Slide	P value	Species	Slide	P value
Grass 1	1	0.0006	*Tsuga caroliniana*	2	0.0016
Rubus americanus	2	0.0022	*Quercus rubra*	1	0.0054
Solidago spp.	1	0.0076	*Kalmia latifolia*	1	0.011
Eutrochium fistulosum	3	0.0226	*Quercus prinus*	1	0.0264
Echinacea spp.	3	0.0228	*Sassafras albidum*	1	0.0272
Asteraceae spp.	2	0.0256	*Carya tomentosa*	1	0.0312
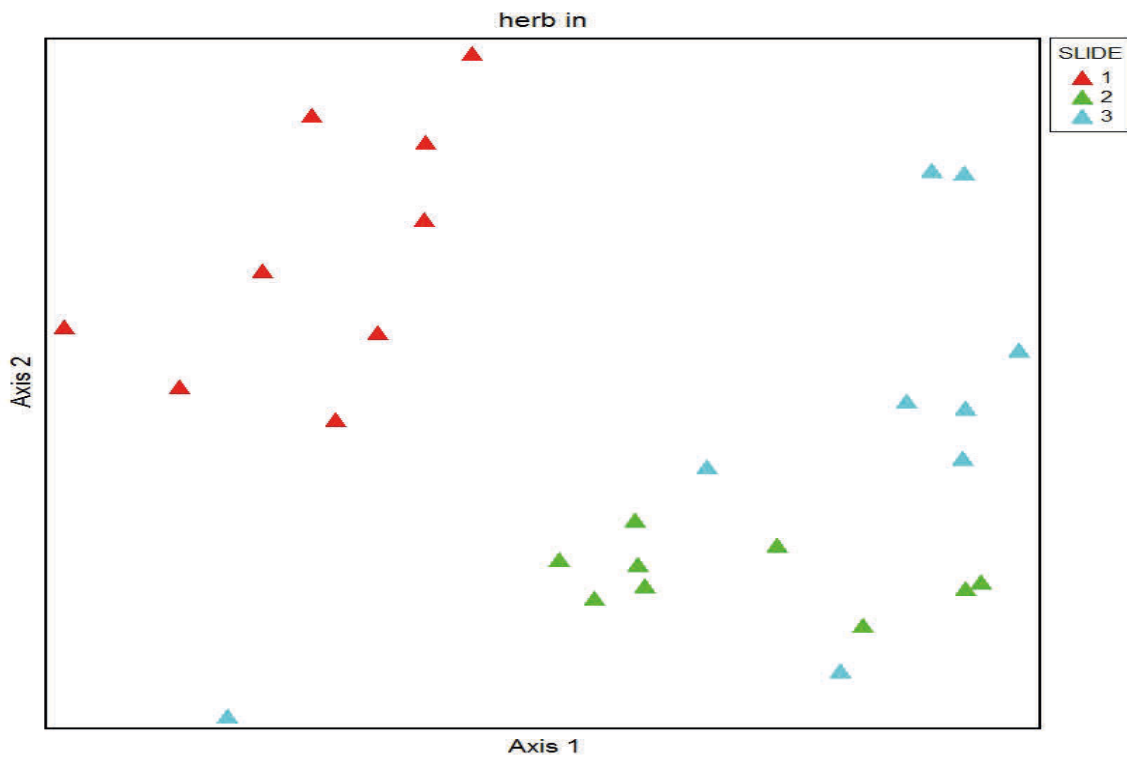
FIG 2. Herbaceous species MDS, where slide 1 is Buck Creek slide, slide 2 is Big Bear Pen slide, and slide 3 is Peeks Creek slide.
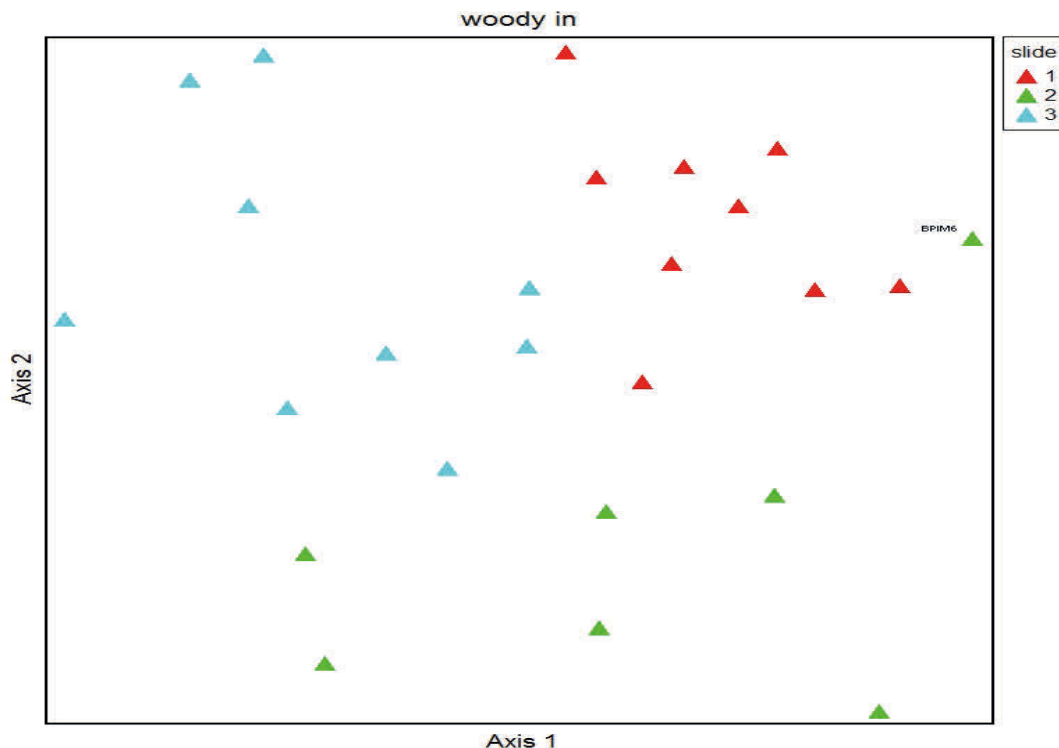
FIG 3. Woody species MDS where slide 1 is Buck Creek slide, slide 2 is Big Bear Pen slide, and slide three is Peeks Creek slide.

Woody Flora

The three landslides had many species of woody flora in common, including *A. rubrum, L. tulipifera, O. arboreum, K. latifolia, P. strobus, R. maximum*, and *S. albidum* (Table 2). These are woody flora commonly found in early successional forest. The three most frequent early succession woody flora were *K. latifolia* (26%), *O. arboretum* (26%), and P. *strobus* (24%). The three woody flora with the highest frequencies that are not considered early succession woody species were *Q. rubra* (17%), *Q. alba* (13%), and *C. tomentosa* (11%). The three most abundant early succession woody flora are present in higher frequencies compared to the three most abundant non-early succession woody species.

Herbaceous Flora

There are several species of herbaceous flora that were found in each of the three landslides (Table 3). There are five taxa that are not indicators of early successional forest, nine indicators for successional forest, and one taxon with an unknown significance. The three early successional herbaceous flora with high frequencies include *R. americanus* (27%), *S. retroflexum* (21%), and Grass 1 (13%). The three most frequently plotted non-early successional indicators include *D. punctiloblua* (11%), *S. tundifolia* (11%), and *G. pubescens* (5%). The three early successional forest indicators show up in higher frequencies compared to the non-early successional indicators.

Table. 2. Fifteen most abundant woody species found in plots. The column called indicator of early succession was developed by crossing my data with Elliot et al. (2007) and Hull and Scott's (1982) data to compare frequencies (%) to flora found commonly in early successional forest due to disturbance.

Species	Number of times plotted	Frequency (%)	Indicator of early succession(Y/N)
Kalmia latifolia	14	26%	Y
Oxydendrum arboreum	14	26%	Y
Pinus strobus	13	24%	Y
Acer rubrum	11	20%	Y
Rhododendron maximum	11	20%	Y
Liriodendron tulipifera	9	17%	Y
Quercus rubra	9	17%	N
Quercus alba	7	13%	N
Carya tomentosa	6	11%	N
Clethra acuminata	6	11%	N
Acer pennsylvanicum	4	7%	N
Tsuga caroliniana	4	7%	N
Quercus prinus	4	7%	N
Sassafras albidum	4	7%	Y
Betula alleghaniensis	2	4%	N

Table 3. Fifteen most abundant herbaceous flora found in plots. The column called indicator of early succession was developed by crossing my data with Elliot et al. (2007) and Hull and Scott's (1982) data to compare frequencies (%) to flora found commonly in early successional forest due to disturbance.

Species	Number of times plotted	Frequency (%)	Indicator of early succession(Y/N)
Rubus americanus	15	27%	Y
Symphyo trichumretroflexum	12	21%	Y
Grass1	7	13%	Y
Asteraceae spp.	6	11%	Y
Dennstaedtia punctilobula	6	11%	N
Smilaxro tundifolia	6	11%	N
Solidago spp.	5	9%	Y
Echinacea spp.	4	7%	Y
Eutrochium fistulosum	4	7%	Y
Goodyera pubescens	3	5%	N
Grass 4	3	5%	Y
Micranthes petioaris	3	5%	N
Grass 2	2	4%	Y
polygonum spp.	2	4%	N/A
Toxicodendron radicans	2	4%	N

DISCUSSION

Several taxa that I observed were indicator taxa for early successional forest because they showed up in higher frequencies than the non-successional taxa. *Kalmia latifolia, O. arboreum,* and *P. strobus* were the most frequently found woody taxa within the plot. Other woody taxa were found in low frequencies but still indicated early successional forest (e.g. *S. albidum* which

was found in only 7% of the time). According to Hull and Scott (1982) *S. albidum*, *L. tulipifera*, *B. lenta*, and R. *pseudo-acacia* were the most important woody taxa found in their study of landslides in Nelson County, Virginia. Elliot at el. (2007) also found similar significant woody taxa in their study of re-establishment of a watershed after a clear-cut in the southern Appalachians. Taxa they found in abundance include, *L. tulipifera*, *K. latifolia*, *A. rubrum*, and *R. maximum*. The suite of taxa that found in my study suggest there is a relationship between the woody taxa found in both landslides and early successional forest. The three taxa that showed up frequently in my plots that were not considered early successional taxa are *Q. rubra*, *Q. alba*, and *C. tomentosa*. It is unclear why these three taxa were so predominant in the landslides I studied.

There was a correlation between herbaceous taxa found in the landslides I studied and early successional forest development. *Rubus americanus*, *S. retroflexum*, and Grass 1 were the three most common herbaceous taxa that I found in the landslides, and all three species are indicators of early successional forest. I cannot identify grasses to species, so grasses were identified by morphotype, and thus labeled Grass 1, Grass 2, etc. According to Smith at el. (2007) grasses help characterize early successional habitat. Hull and Scott (1982) also found *Rubus* spp. to be an important part of revegetation of landslides, and *R. americanus* was the most frequent herbaceous species in the landslide plots. Elliot et al. (1997) found Aster species and *Rubus*. spp to be abundant in a clear-cut watershed. In my landslide plots, S. *retoflexum*, which is in the aster family, was the second most frequent species after *Rubus spp*. Like the woody taxa, there are taxa of herbaceous species that are found in high frequencies in the plots that are considered non-early successional flora. *Dennstaedtia punctiloblua* and *Smilax rotundifolia* were found in 11% of the plotted areas, which is a higher frequency than that of some of the taxa that are considered to be early successional flora. I do not know the significance of these herbaceous taxa and, as with the woody taxa, this is an area that requires further research.

CONCLUSION

The most severe natural disturbances which disturb the canopy, understory, and soil will open up patches large enough for early successional vegetation (Peter et al. 2011). Landslides have the potential to cause that much destruction when they slide. This study showed that there is a correlation between the frequencies of observed taxa in landslides and in early successional forest.

The addition of more plots to each landslide area, a survey of species abundance within plots, and measurement of soil depth and light availability would be three ways to improve this current study, and contribute to our understanding of early successional forest created by landslides. There will need to be a more in-depth study done on landslides to determine if slides create early successional forest.

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THE HABITAT OF *SARRACENIA PURPUREA* VAR. *MONTANA*, THE MOUNTAIN PURPLE PITCHER PLANT

ZACHARY MACKENZIE

Abstract. With the possible addition of *Sarracenia purpurea* var. *montana*, a rare plant species of the southern Appalachian Mountains, to the U.S. Fish and Wildlife Service's Endangered Species List, research on the habitat of the plant will be very important in the near future to help inform the decision to place the species on the list and to aid conservation and habitat restoration efforts. The greatest threat to *S. purpurea* is most likely habitat loss due to encroachment of woody vegetation into the high-elevation bogs in which the species is naturally found. The purpose of this study is to aid future conservation and habitat restoration by providing a better understanding of the habitat required for a healthy population of *S. purpurea* and to determine how the proximity of large woody-vegetation impacts the pitcher plants. My hypotheses that the abiotic factors of habitats with large woody vegetation will vary greatly from those of a bog-like habitat and that the morphological variations of the individuals of *S. purpurea* will show signs of adaptations to a less-suitable habitat were supported by the results of the study. Soil moisture content and organic matter decreased between habitats as the amount of woody vegetation increased. The morphological variations of the sampled individuals of *S. purpurea* indicated that the plants have adapted to existing in unsuitable habitats filled with woody vegetation. The individuals located in habitats with greater amounts of woody vegetation exhibited larger keels and longer pitchers as relative to the pitcher width, which are all indicators that the plants are growing in a dry and shaded habitat. I propose that the undesirable morphology of small pitchers and large keels in a shaded environment is created naturally in a habitat with large amounts of woody vegetation. The results of this study indicate that the greatest conservation and habitat restoration efforts will most likely need to focus on controlling the encroachment of woody vegetation into bog habitats because the presence of woody vegetation negatively alters the habitat of *S. purpurea*, and in turn causes undesirable responses in the morphology of the plant.

Key words: morphological variation; Sarracenia purpurea *var.* montana*; soil pH; soil moisture content; soil organic matter; woody vegetation.*

INTRODUCTION

The southern Appalachians are home to numerous rare plant species that are threatened by further depletion due to various natural and unnatural causes. One such rare plant is *Sarracenia purpurea* var. *montana*, the mountain purple pitcher plant. *S. purpurea* is a perennial, rosette-forming, carnivorous plant species that is naturally found in high-elevation bogs, which contain very nitrogen-poor soil. *S. purpurea* is competitive in these nitrogen-poor soils because it exhibits modified leaves that are curled into the shape of a pitcher. These pitchers fill with rainwater and allow the plant to capture insects, which become a source of nitrogen for the plant (Ellison et al. 2004, USDA Forest Service).

A unique characteristic of *S. purpurea* is that its many morphological variations can be used to predict the environmental factors and the suitability of its habitat (Ellison et al. 2004). Because the pitchers of *S. purpurea* are actually modified leaves, the photosynthetic capability of an individual with pitchers is limited. Attached to the pitcher is a piece of tissue known as a keel, or wing, which resembles a more traditional leaf. The purpose of the keel is to perform photosynthesis as well as to hold the pitcher upright (Shreve 1906, Ellison and Gotelli 2002). Carnivory in plants is believed to have originated in nutrient-poor habitats with plenty of sunlight, such as bogs, because the benefits of the nutrients obtained by carnivory exceed the "marginal photosynthetic cost" of creating carnivorous organs (Givnish et al. 1984). Larger

keels, as relative to pitcher diameter, are more efficient at photosynthesizing (Ellison and Gotelli 2002). When there are large amounts of nitrogen in the soil, individuals of *S. purpurea* will exhibit narrower pitchers relative to the keel, because it is not as necessary to focus on prey capture in a nitrogen rich habitat, and it is more beneficial to maximize rates of photosynthesis. Ellison and Gotelli (2002) found that the morphological response of *S. purpurea* to its environment is so sensitive that the plant can be used to predict the availability of nitrogen in the soil. Individuals will also exhibit larger keels and narrower and longer pitchers when the habitat is drier and more shaded (Shreve 1906, Ellison et al. 2004). Ne'eman et al. (2006) simulated this particular morphology by covering individuals of *S. purpurea* to mimic a shaded habitat and simulated a smaller pitcher size by covering the opening of the pitcher. They found that a narrow pitcher relative to the keel is not an optimal morphological variation for survival in a nitrogen-poor habitat and that reproductive success is limited by this morphology.

Research on *S. purpurea* and its habitat is of special concern because the species is a "petition species" that could be added to the U.S. Fish and Wildlife Service's Endangered Species List (Alexander, pers. comm.). A possible threat to *S. purpurea* could be habitat loss due to the encroachment of woody vegetation into the normally very moist environment of the high-elevation bogs. Natural causes and human activity (e.g. climate change, dredging, and alterations to the flow of water and the water table) have caused some bog habitats to become drier, altering the plant communities within the bogs (Collins et al. 1982, Weltzin 2000, Langanke et al. 2007). As woody-vegetation, especially large trees, moves into the bogs, further loss of soil moisture occurs and the habitat may begin to transition into more of a forest rather than a bog. The purpose of this study is to aid future conservation and habitat restoration efforts by providing a better understanding of the habitat required for a healthy population of *S. purpurea* and to determine how the proximity of large woody-vegetation impacts the pitcher plants. I sampled individual pitcher plants at three populations of visibly varying habitat types in Jackson and Macon Counties, North Carolina. I completed the field work for this study during the months of September and October 2014 before the plants began senescing. Because of the rarity of *S. purpurea* and the threat of poaching, the exact location of these populations will not be disclosed. I hypothesize that the abiotic factors of the habitats with large amounts of woody vegetation will vary greatly from those of a bog-like habitat and that the morphological variations of the individuals of *S. purpurea* will show signs of adaptations to a less-suitable habitat.

MATERIALS AND METHODS

Plant Sampling

I sampled 20 individuals of *S. purpurea* at each of the three populations in Jackson and Macon Counties, North Carolina. Population 1 was in a sphagnum bog very suitable for *S. purpurea*. Population 2 was also in a bog-like habitat near the edge of a forested lake. The habitat of Population 3 was a less suitable habitat, as it was much drier and surrounded by a forest dominated by white pine (latin namae) and red maple (latin name). I transected each population with a meter tape so that about half of the population was on either side of the tape. Using the meter tape, I found the length of the transect. I then divided the meter tape into 20 equal lengths. A plant was sampled at each of the 20 lengths along the meter tape. In order to ensure a random sampling, I rolled a die to determine the distance from the meter tape at which

the sampled plant would be chosen. For Population 1, where the plants were more spread out, the number on the die represented the distance in feet from the meter tape at which the plant would be sampled. At Populations 2 and 3, the number on the die corresponded to how many half-feet away from the meter tape the plants would be sampled. The side of the meter tape at which the plants were sampled was alternated with each plant. Because Population 3 covered a large area, I divided the population into five distinct groups. I found the total number of individuals in the population and calculated the percentage of total individuals contained in each group. This percentage was used to find how many of the 20 samples would be taken from each group.

Understory and Canopy Light Availability

At each plant sampled, I measured percentages of understory and canopy light availability using a densiometer. I held the densiometer just above the plant in order to measure the understory light availability. To measure the canopy light availability, I held the densiometer at elbow height. For measuring both the understory and canopy light availabilities, I calculated the percentage of available light while facing each of the four cardinal directions and then calculated an average of the percentages.

pH

At each individual of *S. purpurea* sampled, I measured the pH of the soil using a Kelway Soil Tester®, Model HB-2. As per the soil tester instructions, I left the tester in the soil for three minutes before taking a reading of pH. After each sampling, I used Kelway Conditioning Film to chemically clean the tester plates.

Soil Analysis (Moisture Content and Organic Matter)

At each plant sampled, I used a 15 cm soil corer to obtain a soil sample adjacent to each plant. A single procedure was used to find both the moisture content and organic matter of each soil sample. This procedure is known as "Weight Loss-on-Ignition" (LOI) as described by the Soil Testing Laboratories of the University of Wisconsin-Madison (2004). I filled small glass jars with about five grams of soil and dried the soil for two hours at 105 ^{o}C. After two hours, I measured the mass of the soil again. I used this mass to calculate the percent moisture content by mass using the formula:

Moisture Content (%) = 100 * ((original mass – mass at 105 ^{o}C) / original mass)

Then, I heated the soil at 290 ^{o}C for two hours. Although the procedure described by the Soil Testing Laboratories calls for heating the soil at 360 ^{o}C, the oven I used reached a maximum temperature of only 290 ^{o}C. I used the mass measured after the second two hours to calculate the percent organic matter by mass of the soil by using the formula:

Organic Matter (%) = 100 * ((mass at 105 ^{o}C – mass at 290 ^{o}C) / mass at 105 ^{o}C)

I heated the soil samples in batches of 10. With each batch, I included a sample of soil of known LOI as a quality control measure.

Morphological Variations

In order to determine if the abiotic factors of each habitat, such as light availability, soil pH, moisture content, and organic matter, are suitable for a healthy population of *S. purpurea*, I measured certain physical properties of each plant. The physical properties measured included the number of leaves, the length of the longest pitcher, the width of the largest mouth, and the width of the largest keel in each rosette. I measured the widths of the mouth and wing to the nearest 0.1 mm at their widest points using a caliper. I measured the pitcher length to the nearest 0.1 cm using a ruler.

RESULTS

Understory and Canopy Light Availability

The populations in the bog-like habitats (Populations 1 and 2) had the greatest amounts of available light in the canopy (Fig. 1). Population 3 had the smallest amount of available light in the canopy. Although Population 2 had a greater range of values for understory light availability, there was not much variation in the median percentages of understory light availability among the populations (Fig. 2).
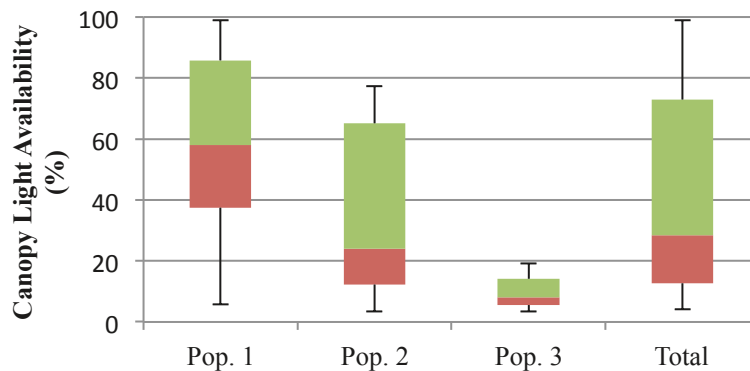
FIG. 1. Variation in canopy light availability among the populations. Percent canopy light availability decreased from Population 1 to Population 3. The last plot includes the data from all three populations. For all such plots in this paper, the lines extending off the box show the minimum and maximum values, the bottom and top box edges represent the first and third quartile values, respectively, and the line within the box represents the median value.
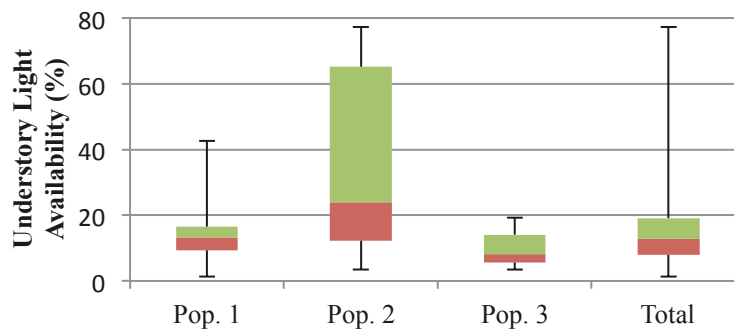
pH

The soil pH did not appear to show large variation or trends among the populations. The pH values taken at each population were largely all within the same range (Fig. 3).
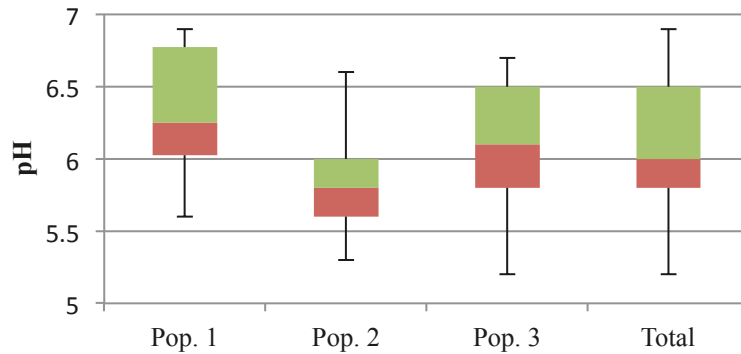
FIG. 3. Variation in soil pH values among the populations. The pH values at each population were largely within the same range.

Soil Analysis (Moisture Content and Organic Matter)

Both aspects of the soil analysis showed variations and trends among the populations. Population 1 had much greater values for organic matter content by mass, while Populations 2 and 3 had very similar, smaller values (Fig. 4). The moisture content by mass followed a comparable trend to the organic matter content. Population 1 had larger values, while Populations 2 and 3 had very similar, smaller values (Fig. 5).

I found that increased canopy light availability, indicating lower presence of woody vegetation, was correlated with an increase in soil moisture content (Figs. 6 and 7). No such trends were observed in understory light availability.
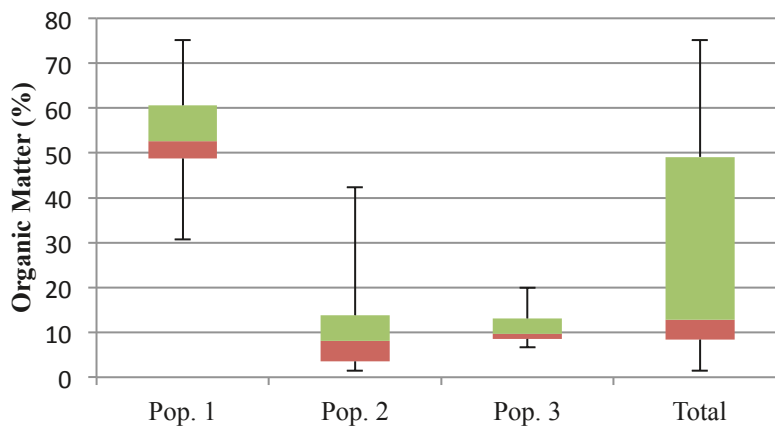
FIG. 4. Variation in soil organic matter among the populations. Percent organic matter decreased from Population 1 to Population 3.
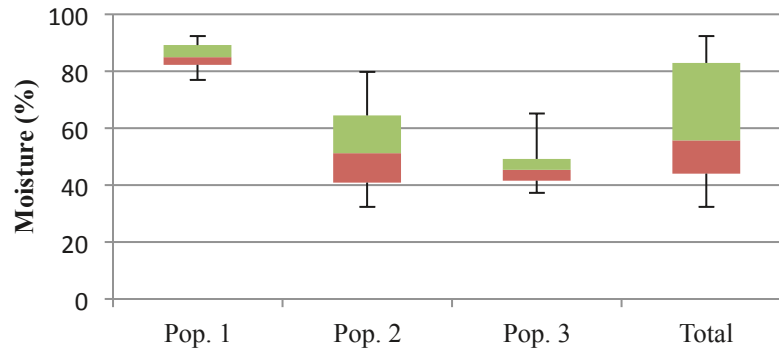
FIG. 5. Variation in soil moisture content among the populations. Percent moisture content decreased from Population 1 to Population 3.
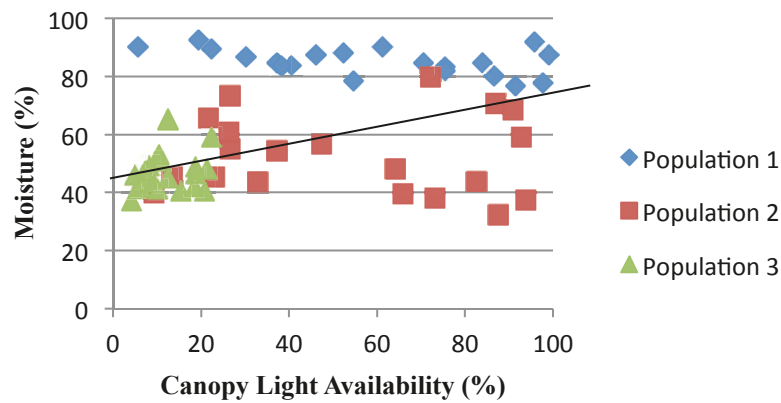
FIG. 6. The relationship between canopy light availability and soil moisture content. Percent moisture content increased with increasing percent canopy light availability. Slope = 0.2835. r^2 = 0.2398.
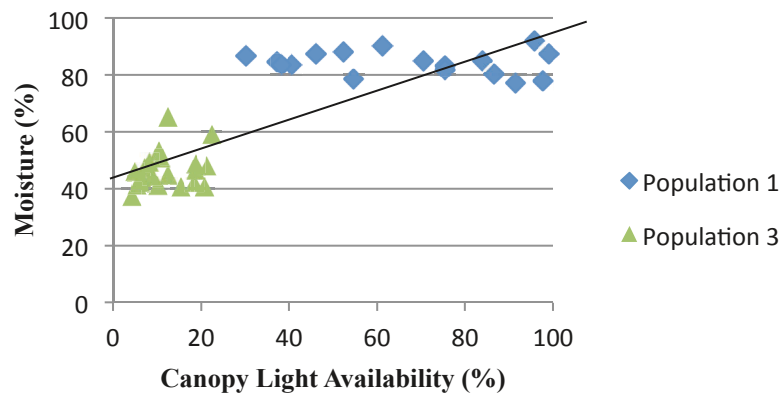
FIG. 7. The relationship between canopy light availability and soil moisture content (adjusted). This figure depicts the same increasing trend between canopy light availability and soil moisture content as does Fig. 6. Because the soil at Population 2 was rocky and less likely to retain moisture, I removed its data values from this figure. I also excluded the first three data points from Population 1 because, although these plants were within the bog, they were located near the edge of the surrounding forest and have smaller values for canopy light availability, which skew the data. Slope = 0.5092. r^2 = 0.6762.

The morphology of the plants responded to the varying abiotic factors of their habitats. There was a slight trend in the number of leaves per plant (Fig. 8). The individuals in Populations 1 and 2, the populations with more light availability and soil moisture content, tended to have slightly more leaves per plant than Population 3. There was a distinct trend in the morphologies that indicate a dry and shaded environment. Populations 1 and 2 both had higher values for the pitcher width to pitcher length ratio (Fig. 9), meaning that the individuals in Populations 1 and 2 had larger pitchers relative to pitcher length than Population 3. There was also a decreasing trend in the ratio of pitcher width to keel width from Population 1 to Population 3 (Fig. 10).
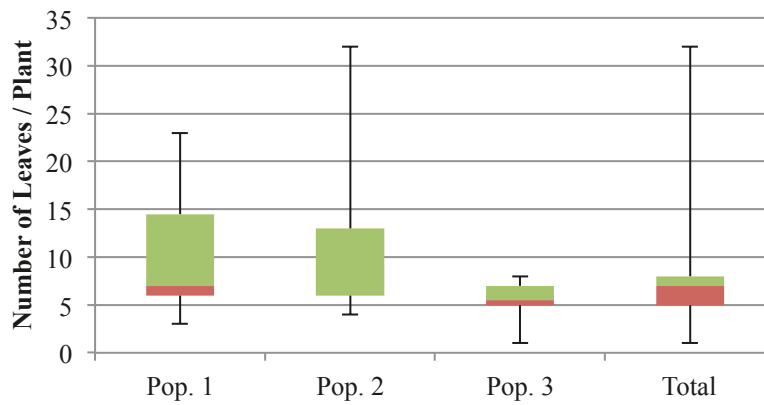
FIG. 8. Variation in the number of leaves per plant among the populations. There is a slight decrease in the number of leaves per plant from Population 1 to Population 3, with Population 3 having a much smaller range of values. Note: The median value and the value of the first quartile were the same for Population 2.
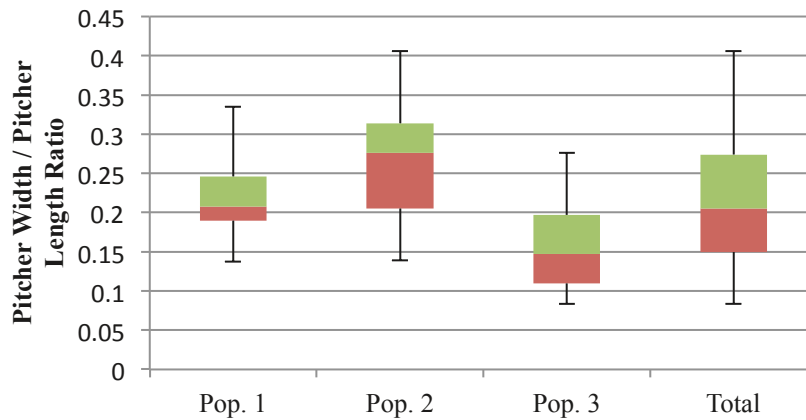
FIG. 9. Variation in pitcher width to pitcher length ratios among the populations. Populations 1 and 2 had higher values for pitcher width to pitcher length ratio.
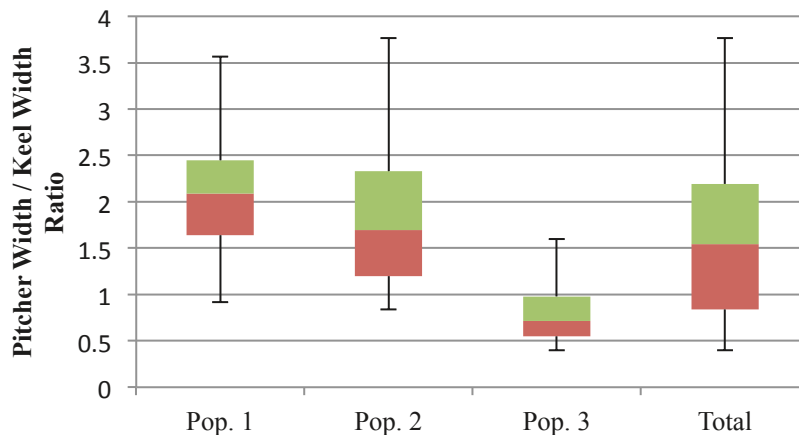
FIG. 10. Variation in pitcher width to keel width ratios among the populations. The pitcher width to keel width ratio decreased from Population 1 to Population 3.

DISCUSSION

Overall, the results of this study supported my hypotheses that the abiotic factors of the habitats with large woody vegetation will vary greatly from those of a bog-like habitat and that the morphological variations of the individuals of *S. purpurea* will show signs of adaptations to a less-suitable habitat. The amount of woody vegetation varied visibly between the habitats of the three populations. This variation in woody vegetation was also confirmed by the results of measuring canopy light availability, with Population 1 having the greatest values for canopy light availability, and Population 3 having the smallest values. Although canopy light availability varied greatly among the habitats, understory light availability did not. The soil pH also did not vary greatly among the habitats. Other abiotic factors of the habitats were more affected by the proximity of woody vegetation. Soil moisture content and organic matter both decreased from Population 1 to Population 3. Population 2 had smaller values for moisture content than I expected. This can be explained by the results of the organic matter analysis. As can be seen in Fig. 4, the soil of Population 2 contained little organic matter and was actually quite rocky, meaning that it is less likely to retain moisture. The large range in the values of Population 2 for both organic matter and moisture content is most likely explained by the population's location extending from the edge of a forest to the edge of a lake. This observed trend of decreasing moisture content returns us to the question: How does large woody vegetation affect the abiotic factors of the habitat of *S. purpurea*? The positive correlation found between canopy light availability and soil moisture content suggests that woody vegetation does in fact decrease soil moisture content.

The morphological variations of sampled individuals of *S. purpurea* also indicated that the plants have adapted to existing in unsuitable habitats filled with woody vegetation. The individuals of Population 3 (located in the drier and more shaded habitat) exhibited larger keels and longer pitchers as relative to the pitcher width, indicators that the plants are growing in a dry and shaded habitat. I propose that the undesirable morphology of small pitchers and large keels in a shaded environment, as described by Ne'eman et al. (2006), is created naturally in a habitat with large amounts of woody vegetation. Because nitrogen availability can also alter the morphology of *S. purpurea*, it may be beneficial in future studies to measure the levels of nitrogen in the soils of the three habitats.

The results of this study may aid the U.S. Fish and Wildlife Service in deciding if *Sarracenia purpurea* var. *montana* should be placed on the Endangered Species List and in determining the abiotic requirements of the habitat of a healthy *S. purpurea* population. Although the values shown for all three populations show the various ranges of the abiotic factors in which *S. purpurea* can exist, it is important to note that Population 1 was found in the most suitable habitat, and that these values act as a better guideline for the requirements of a suitable habitat for *S. purpurea*. The greatest conservation and habitat restoration efforts will most likely need to focus on controlling the encroachment of woody vegetation into bog habitats because the presence of woody vegetation negatively alters the habitat of *S. purpurea*, and in turn causes undesirable responses in the plant's morphology. I believe that if any restoration efforts are made to control woody vegetation in any of the three habitats, it would be very beneficial to the preservation of *Sarracenia purpurea* var. *montana* to study how the populations respond to such efforts.

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GROWING CONDITIONS AND SEED SET IN *STEWARTIA OVATA*

KATIE RUSSELL AND ELLA WICKLIFF

Abstract. Stewartia ovata (Theales: Theaceae) is a rare small tree found primarily in the southern Appalachians and nearby regions. We studied the environmental and growing conditions that affect *S. ovata* seed set in order to aid both gardening and conservation efforts. The specific parameters we studied include light availability, slope aspect, and tree size. We collected data and fruit from both cultivated and wild populations of *S. ovata.* The average seed yield for cultivated *S. ovata* of 45% was greater than the average seed yield for the wild *S. ovata* of 39%. Based on the observed growing conditions in wild populations, we also studied cultivation techniques that allow for the greatest viability when growing *S. ovata* from seed, cutting, and plantings. Correlations between seed set and environmental conditions in the wild may be helpful to gardeners and to conservationists working with this rare species who hope to maximize seed yield. However, we were unable to find correlations between environmental conditions and the number of capsules produced by the tree or the total number of seeds taken from those capsules. There are many other factors, yet unexplored, that may affect fertility in these trees.
Key words: Stewartia ovata; Theaceae; light availability; rare plant conservation; seed set.

INTRODUCTION

Stewartia ovata, or mountain camellia (Theales: Theaceae; Table 1) is found primarily in the southern Appalachian mountains, but it is also found as far north as Virginia and as far west as Mississippi (Boland et al. 2008; Fig. 1). In the southern Appalachians, *S. ovata* grows at relatively low elevations (343-788.5 meters), often where a steep slope meets a level area (Thurman-Irons 2013). It prefers moist but well-drained soil (Johnston, pers. comm.) and is nearly always found in river valleys and along rivers (Thurman-Irons 2013). *Stewartia* grows particularly well in areas with exposure to large amounts of sunlight such as riverbanks, road sides, or bluffs. Populations are also frequently found in canopy gaps, many of which have been recently created by fallen hemlock trees that allow light to reach the understory. *Stewartia ovata* do well in acidic soils, and also grow well with rock fragments that have acidic composition (Jack Johnston, pers. comm.). Associated species include *Acer rubrum*, *Tsuga canadensis*, *Pinus strobus*, and other species found in the acidic cove plant community (Deans 2012).

Stewartia ovata is a bushy deciduous shrub or small tree, which reaches about 5.5 meters in height and usually has multiple stems. Its leaves are 12.7-15.2 cm long, ranging in shape from ovate to elliptic, with broadly winged leafstalks (Fig. 2). In the fall, the leaves turn orange, red or deep purple. *Stewartia ovata* blooms for approximately two weeks from mid-June to the first week of July in our study area (Little 1980). It is well known for its white five-petaled flowers, which can be up to 6.5 cm across (Spongberg 1975) with ruffled margins and orange anthers. The flowers are pollinated by beetles, bees, and flies (Ley 2014). The fruits are relatively large 1.5–2.2 × 1.4–1.6 cm (Prince 2009), round or conical, and five-ribbed. The woody fruits develop from the compound ovaries at the center of the flowers. The fruits, initially green, turn brown as they mature and open between mid-September and mid-October. The seeds are released from five internal compartments and dispersed mainly by the wind (Spongberg 1975). In general, the fruits contain a maximum of 10 seeds, or two per compartment. It is possible for a compartment to have more than two seeds per compartment or less than two seeds per compartment due to abortion. Abortion may occur if the ovule is not fertilized or fails to develop after fertilization. The seeds are doubly dormant, which means that they have seed coat (external) and internal

dormancy, and require scarification as well as stratification before they will germinate. In the wild they require two years to germinate (Spongberg 1975).

Table 1. Taxonomic information for *Stewartia ovata.*

Classification Rank	Scientific Name and Common Names
Kingdom	Plantae
Subkingdom	Tracheobionta – Vascular plants
Superdivision	Spermatophyta- Seed plants
Division	Magnoliophyta- Flowering plants
Class	Magnoliopsida- Dicotyledons
Subclass	Dilleniidae
Order	Theales
Family	Theacea- Tea family
Genus	Stewartia L. - Stewartia
Species	*Stewartia ovata*- mountain camellia
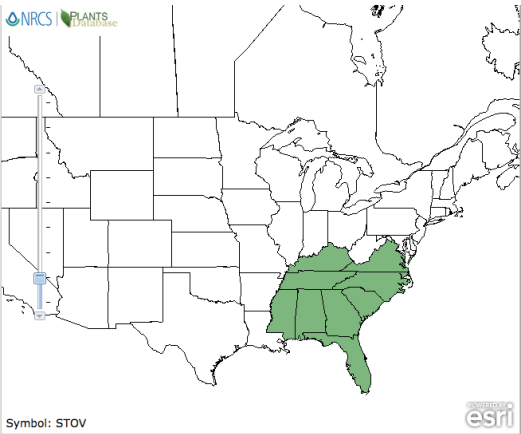
Fig. 1. Range map of *S. ovata* (Britton et al. 1913).   Fig. 2. *S. ovata* seeds, leaves and flower (USDA, NRCS 2014).
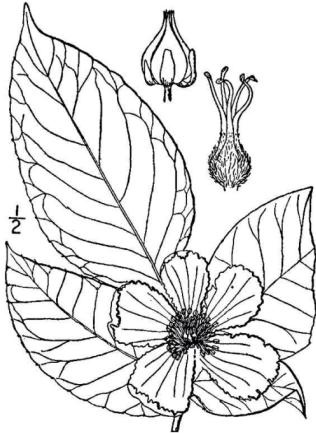
We collected *S. ovata* seeds between late August and mid-November of 2014. We collected data on environmental conditions at *S. ovata* sites to look for a correlation between growth conditions and seed yield, and were especially interested in sites showing evidence of greater seed yield - that is, more seeds per fruit or a lower rate of abortion. We also planted our collected seeds in a seedbed in order to quantify seed viability, which is another measure of fertility and reproductive success of plants. Viability cannot be quantified for several years. Correlations between seed set and environmental conditions would be helpful to gardeners who wish to plant trees for great reproductive success and to conservationists working with this rare species, who hope to have the greatest seed yield possible.

METHODS AND MATERIALS

Site Selection

We surveyed four locations with known wild *S. ovata* populations. The first wild site was along Needmore Road, adjacent to the Little Tennessee River, Macon County, North Carolina. The second wild site was on Rattlesnake Road, which branched off of Needmore Road, also in

Macon County, NC. The other two wild sites were located in Rabun County, Georgia: one on the Tallulah River, and the other at a drainage of the Chattooga River called Sarah's Creek.

We also examined three sites under active management and cultivation within the natural range of *S. ovata*. The first cultivated site we examined was Jack Johnston's garden at his home in Clayton, GA (Rabun County). There we observed an example of long-term successful propagation techniques and cultivation. The garden is located near a natural spring and is laid out as a series of flattened ridges, which allows plants to grow on a slope and still be accessible. The garden contains mature trees as well as numerous seed beds and allows for decreased competition from overstory trees and abundant sunlight for *S. ovata* growth. The second cultivated site, also property of Jack Johnston, was located in Otto, NC (Macon County). This population was planted within the past 20 years. The population was near a stream, but had limited light availability due to the established canopy trees. The third cultivated site, called Snow Hill Falls, was also in Macon County, near Franklin, NC. The trees at this site grew on a slope had abundant sunlight, and were downslope from a pond and near a small stream.

Measurements Taken to Assess Growing Conditions

Within each wild population we recorded associated plant species within a 1.5 meter radius of each *S. ovata* tree. We recorded associates in order to determine the common forest communities in which *S. ovata* grows as well as other species that benefit from growing in the same environmental conditions. GPS locations and elevation information were recorded for each wild population using a Garmin® GPSmap 60CSx GPS unit. At each wild site we used a compass to take slope aspect at each tree. We stood near the tree and faced the direction of the downward slope, then used our compass to determine our direction. We collected fruit from three to six *S. ovata* plants per population at both our wild and cultivated sites. The seeds were removed from the tree by gently twisting the fruit off of the main stem. On average, approximately 40-100 or more fruits were collected per tree within each population.

At each tree from which we collected fruit we also measured light availability using a spherical densiometer, a pocket-sized instrument that has a gridded mirror with concave curvature enabling the reflection of a large area overhead (California Department of Pesticide Regulation 2004). On the concave mirror, there was a grid of 24 squares and in each square we imagined four dots in order to obtain the number of dots that showed where open canopy was present. Measurements for each tree were taken outside of that tree's canopy. We held the densiometer flat and level at waist height, approximately 12 inches from the body and took four densiometer readings at each tree – one in each cardinal direction. We took the average number of dots that indicated light availability at each tree and we multiplied this number by 0.26 to get percent of sunlight available.

For each tree from which we collected fruit, we also collected the Diameter at Breast Height (DBH), a standard metric of tree size. DBH is measured at 1.37 meters from the base of the tree. We used a diameter tape that indicated the circumference of the trunk on one side of the tape and the calculated diameter on the other side based on the pre-defined conversion factor, assuming a circular cross-section (AFCDCB 2006). If more than one stem was present, we recorded the largest stem's DBH.

After collection, we cracked open the fruits to collect the seeds inside. We used pliers to open the fruits because the fruits were closed at the time of collection. Once we opened each compartment within a fruit, we recorded the total number of seeds as well as the total number of

fruits. We calculated the average seeds per fruit by dividing the total number of seeds by the total number of fruits. We also calculated the percent yield by dividing the actual number of seeds collected by the maximum possible yield (assuming a maximum of 10 seeds per fruit). We also recorded the mode, or the most frequent number of seeds per fruit.

Seed Propagation Techniques for Cultivation Purposes

Propagating *S. ovata* from seed requires imitating its optimal natural growing conditions. First we placed rotting wood-chips in the bottom of a pot and then we added a layer of approximately two inches of decomposing leaves to establish mycorrhizal associations. Next we sprinkled seeds evenly into the pot, followed by another layer of leaves. To the top we added ground pine bark to the pot to ensure that the moisture content within the pot was not too high. We then added the newly potted seeds to a cage for protection, where it may take up to seven months to three years for the seeds to germinate (Spongberg 1975, Jack Johnston, pers. comm.).

Building a Seed Bed

Another technique we used in a gardening setting to propagate *S. ovata* under natural conditions was to build a seed bed. We selected the site of the seed bed by considering the factors known to influence growth in *S. ovata* including moisture availability and drainage, light availability, aspect, and soil composition. We then leveled the ground on which the bed was to be built. The bed had three cross beams separating it into four sections. We assembled the seed beds by nailing and attaching all four sides together, and we added a wire mesh to the bottom of the bed to keep out predators that may eat the seeds, such as voles. We moved the bed into place and added mushroom fertilizer to the bottom of the bed and spread a high number of seeds evenly around the bed. Next we added finely crushed leaves to allow for mycorrhizal associations and to allow water to drain rather than accumulate in the bed, which may cause root rot. Then we added more mushroom fertilizer and mixed sand over top to create a substrate for the seedlings to take root in for the next three to five years. Finally, we added wire mesh to the top of the bed to protect from adverse environmental factors and predation.

Proper Planting Technique

For an outplanting to survive, *S. ovata* seedlings should be planted at three to five years of age. Usually the larger and older the seedling is, the better it will be able to adapt to the change in environment caused by planting. The seedling should ideally be planted in an area with good drainage, on a slope, and with consistent light availability without competition from tall canopy species. Red maples are common associates, but these outcompete *S. ovata,* and their growth in the immediate vicinity around the seedling is harmful. Once planted, ample water should be applied. Common mistakes in propagation include overly moist soils which causes root-rot, wrong soil composition, and excessive shading which prevents growth.

Garlon 3A specialty herbicide can be used for foliar inhibition of competing vegetation, a trademark of Dow AgroSciences LLC. Seacom-PGR made by Saltwater Farms is a liquid plant growth regulator that provides a high concentration (400 ppm) of the natural plant growth hormone cytokinin, and leads to enhanced crop growth, increased yields, and healthier, more

productive plants. SeaCom is added 1 tsp. per gallon of water. Spray-N-Grow micronutrients is also a foliar spray used in a concentration of 10 oz per two gallons water.

Humidity Dome Cutting Propagation

Humidity Domes are used to encourage rooting in cuttings. The cutting is inserted into soil and covered by a large glass jar. The jar holds humidity, allowing the cutting to root in a moist environment without the base of the cutting being wet, which would cause root rot.

RESULTS

We collected data from 27 trees at six sites. Four of the sites (Needmore, Rattlesnake, Tallulah and Chattooga River) were located on public land, not maintained, and were considered sites of wild populations. Two sites were under active cultivation (Snow Hill Falls and Otto). Slope aspect data were not collected at the sites under cultivation (Table 2). To generate each figure for the studied environmental factors affecting seed yield we used MS Excel to plot and calculate the correlation coefficient for each graph. Each graph shows the environmental factor (light availability, DBH, Slope aspect) compared to calculated percent yield.

Because *S. ovata* need abundant light for healthy growth, we predicted that higher light availability would lead to a higher seed set. No such trend was found (Fig. 3). We also found no correlation between the size of the tree, measured as diameter at breast height, and the seed yield (Fig. 4). *S. ovata* were found over a wide range of slope aspects. Our analysis could discern no relationship between slope aspect and seed yield (Fig. 5).

Fifteen plant species or species groups were frequently found near *S. ovata*. The most common associated species found within 1.52 m of *S. ovata* individuals from which we collected fruit were ferns (*Pteridophytes*), especially Christmas ferns (*Polystichum acrostichoides*) and red maples (*Acer rubrum*; Table 3).
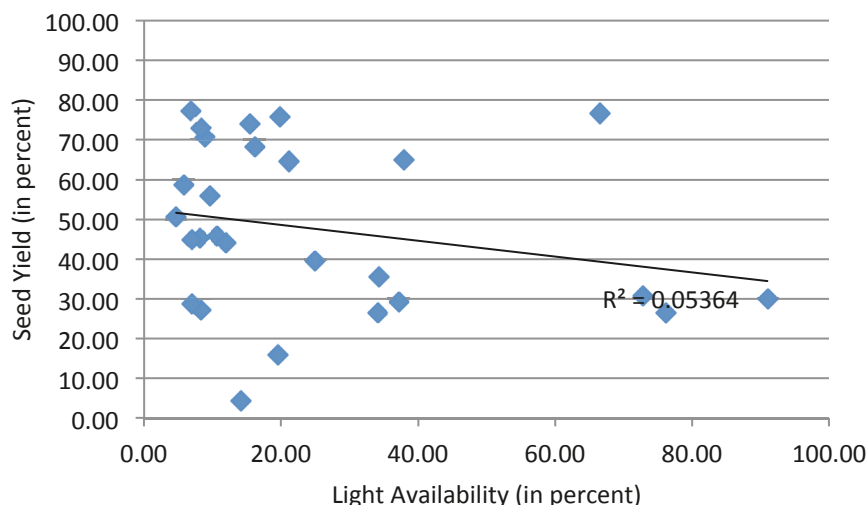
Fig 3. Percent light availability and percent seed yield.
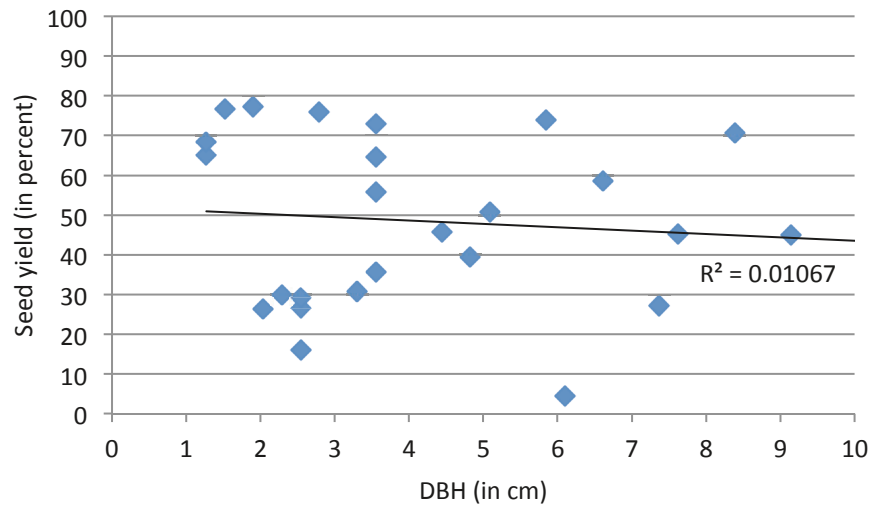
Fig 4. Diameter at Breast Height in cm and percent seed yield.
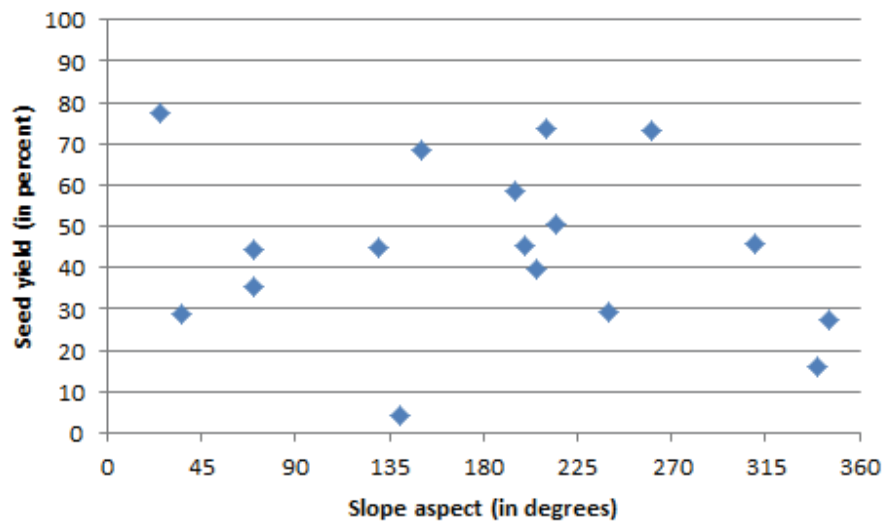
Fig 5. Slope Aspect in degrees and Seed Yield in percent.

DISCUSSION

Our sites were selected by Jack Johnston based on populations that he knew of previously. The growth and survival of *S. ovata* are dependent on light availability (Jack Johnston, pers. comm.), but light availability has no effect on the reproductive capacity of an individual tree (Fig. 3). Tree size and slope aspect also have no bearing on reproductive capacity in our study populations.

Because we did not collect every fruit from the trees that we sampled, nor did we collect a standard proportion (e. g. ⅔ of the fruit from each tree), we were unable to draw correlations between environmental conditions and the number of fruits produced by the tree or the total number of seeds taken from those fruits. In order to mitigate the lack of standardization, we based our trends on seed yield as a percentage.

Table 2. Seed yield, stem count, Diameter at Breast Height, light availability, and slope aspect for all individuals.

Name of Site	% Yield	Number of Stems	DBH (cm)	% Sun	Aspect
Snow Hill Falls	26.50	2	2.54	34.06	
	65.00	2	1.27	37.96	
	26.48	2	2.03	76.18	
	30.83	1	3.30	72.80	
	76.62	2	1.52	66.56	
	30.00	3	2.29	91.00	
Needmore	16.00	1	2.54	19.50	340
	27.19	1	7.37	8.32	345
	45.71	1	4.45	10.66	310
Rattlesnake	70.67	1	8.38	8.84	No slope
	45.33	1	7.62	8.06	200
	28.86			7.02	35
Tallulah	4.29	1	6.10	14.04	140
	44.89	3	9.14	7.02	130
	39.50	1	4.83	24.96	205
	68.33	8	1.27	16.12	150
	44.19	4	10.16	11.96	70
	73.88	1	5.84	15.34	210
Otto	75.94	1	2.79	19.76	
	64.55	1	3.56	21.06	
	55.83	3	3.56	9.62	
Chattooga River	58.62	2	6.60	5.72	195
	50.68	1	5.08	4.68	215
	73.00	1	3.56	8.32	260
	29.12	1	2.54	37.18	240
	77.31	1	1.91	6.76	25
	35.56	1	3.56	34.32	70

Table 3. Common associated species found within 1.52 meters of a *S. ovata* with fruit in a wild population.

Associated Species	% Occurrence in Total Number of Populations	Number of Sites Where Present
Pteridophytes spp.	94.7368	5
Acer rubrum	89.4736	5
Rhododendron maximum	84.2105	5
Pinus strobus	73.6842	3
Liriodendron tulipifera	63.1578	4
Tsuga canadensis	57.8947	5
Hexastylis arifolia	52.6315	4
Leucothoe fontanesiana	52.6315	3
Oxydendrum arboreum	52.6315	4
Quercus alba	47.3684	5
Carya spp.	47.3684	4
Kalmia latifolia	36.8421	3
Vitis rotundifolia	36.8421	4
Ilex opaca	31.5789	3
Gaylussacia baccata	31.5789	2

There were some random errors present in our data collection. Because the trees were often on steep slopes or bent, DBH was not always taken at precisely 1.37 meters from the root system. Densiometer readings were taken without the inclusion of branches or leaves from the trees in question and inclusion of leaves of the trees in question would have resulted in lower light availability than what was recorded in our data. For trees with multiple stems, we recorded only the DBH of the largest stem instead of an additive measure of all stems. However, even without these errors and assumptions, it is unlikely that any pattern would emerge because these would change the raw data only slightly.

The environmental conditions that have been shown to be important for *S. ovata* growth (Deans 2012) appear to have no relation to seed output in a single season. It is possible that over a larger sample size or over many years, the growth factors we observed may be important to seed set in *S. ovata* populations. Future studies that observe seed set over multiple seasons may be able to identify factors that determine annual seed production. Climatic factors such as rainfall and temperature variability may affect seed set annually. Differences in soil moisture may be a factor in seed set because *S. ovata* are very sensitive to soil moisture, thriving in moist but well-drained soils (Deans 2012). High quality soil, from a steady supply of decomposing organic material, results in higher growth rates and could also lead to increased reproductive output. Another factor we did not take measurements on was the proximity of the trees to one another. According to Osada et al. (2004), who studied Theaceae in Japan, fruit set was dependent on how close individual trees were to each other because pollinators were more likely to visit trees in larger populations than isolated trees. Higher pollination rates due to spatial distribution resulted in higher fruit set in high density areas. Reproductive success may also vary with interspecific and intraspecific competition, damage to the plant, predation, and genotypic variation.

To further explore such factors on seed set and viability in *S. ovata* a study could be performed by collecting fruit from a variety of trees and data on soil moisture, quality, pollination and proximity to other *S. ovata*, possible competitors, damage, predators, and genetic material. A long-term study might compare seed yield to these factors in hopes of shedding further light on what contributes to seed yield and viability. We established seed beds over the course of this study for future research to quantify the viability of the seeds we collected during the Fall of 2014. The viability of these seeds will reflect the fertility and reproductive success of the plant, and may correlate with the environmental conditions discussed above.

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AN ASSESSMENT ON WHOLE TREE REVETMENT CONDITION IN THE LITTLE TENNESSEE RIVER BASIN

JAN SITTERSON

Abstract. Whole tree revetments are used to control bank erosion along the Little Tennessee River and surrounding tributaries. Whole tree revetments reduce the water velocity along the bank and catch suspended sediments and floating woody debris for recolonization of woody vegetation. A revetment and riparian zone assessment, better known as a Stream Visual Assessment Protocol (SVAP), was conducted by canoe to determine the success of 15 year-old revetments. This study found that the percentage of revetment remaining after 15 years is lower in large streams than in small tributaries. There was no significant difference in average SVAP score between the larger river and smaller creeks. Higher SVAP scores were correlated with a higher percentage of revetment remaining. In general, wider streams had larger decreases in the percentage of revetment length over time. Larger streams have more sediment load and are more likely to capture and colonize vegetation but they have a greater chance of being blown out due to the current and rise in water levels. The data suggest that revetments in large streams have shorter life spans than in smaller streams because the percentage of revetment left is larger in smaller streams. This assessment showed that whole tree revetments are not 100% successful in the Little Tennessee River basin. The results of this assessment did not support my hypothesis that whole tree revetments are more successful in larger streams than in smaller streams.

Keywords: Whole tree revetment; Stream Visual Assessment Protocol (SVAP); Little Tennessee River; Cartoogechaye Creek; Tessentee Creek; Franklin North Carolina; southern Appalachian Mountains.

INTRODUCTION

Whole tree revetments consist of whole trees that are cabled together on the bank to capture sediment and colonize vegetation along a river bank for stabilization. Revetments are usually built on the outside of a bend in the channel near the thalweg which is characterized by faster currents that increase the risk of bank erosion (Alaska 2009). Whole tree revetments reduce the water velocity along the bank and catch suspended sediments and floating woody debris. The revetment catches enough sediment to allow vegetation to grow and create a more stable bank. The network of roots that develops over time protects the integrity of the bank structure. I hypothesized that streams larger in width have more successful revetments due to the higher amount of sediment load in these wider streams. Whole tree revetments have advantages and disadvantages. The revetments look natural, provide habitats above and below the water line, protect the bank from erosion, self-repair, and can be created from recycled trees (Ellis 2002). Unfortunately, they have negative downstream effects and have a limited lifetime due to natural causes. A revetment may increase rates of erosion downstream, making the placement of revetments crucial (Alaska 2009).

Streams naturally sculpt their channels but the banks of the Little Tennessee River and its tributaries in Macon County, North Carolina are eroding at an enhanced pace due to livestock entering the stream, high sediment loads that scour the riverbanks, and loss of vegetative cover (Love 2003, McLarney 2003). As a result, the North Carolina Clean Water Management Trust Fund in 1995 funded the Macon County Soil and Water Conservation District to restore eroding banks along the Little Tennessee River and its surrounding tributaries. Section 404 of the Clean Water Act, 33 USC 1344 was used in conjunction with section 401 of the NC Division of Water Quality Certification to implement the revetment implantation on the banks of the Little Tennessee River and its tributaries until 2002 (Johnson pers. comm.). Whole tree revetments

were built and seedlings of woody vegetation were planted to stop the accelerated erosion of the stream banks with the funds provided. In the fall of 2014 I assessed 54 of these revetments along the Little Tennessee River and two of its tributaries (Cartoogechaye and Tessentee Creek) in Macon County built from 1995 to 2002 in order to quantify their success and their lifetime.

METHODS

Study Area

The Little Tennessee River is located in the southern Appalachian Mountains originating within Georgia and joining the Tennessee River in Tennessee (Betcher et al. 2011). For this assessment, I selected three different sized streams within Macon County, North Carolina for comparison of revetment success. The assessment sites on the Little Tennessee River stretched from the North Carolina/Georgia border to the southwestern part of the Franklin City limits. A total of 54 revetments were assessed in the fall of 2014 in the Little Tennessee River and two of its tributaries, Cartoogechaye Creek and Tessentee Creek. I could not assess more revetments than this due to limited time and access to the revetments on the river.

FIG. 1. Digital photograph of a successful revetment on Cartoogechaye Creek.

Revetment Assessment

I found information on the original revetments in the Macon County Soil and Water District data files. I converted parcels with revetment work along the river to GPS points with one point at the beginning of the parcel and one at the end. It is easiest to travel by canoe for the investigation on revetments. Revetments are identified by embedded logs in the bank, defoliated branches facing upstream and sawed edges. I conducted a visual assessment at each revetment

using the Stream Visual Assessment Protocol (SVAP) which scores the revetment and bank from one to four on revetment condition, bank condition, riparian zone quantity, riparian zone quality, and livestock access. SVAP requires the user to note the presence of deep pools, wood accumulation, and undercut banks near or on the revetment. At the beginning and the end of each revetment I took a GPS waypoint using a Garmin® (Garmin® 2010). In addition, I took digital photographs of each revetment to compare them to photographs taken when the revetments were built. Additional information on the presence of cables and exotic vegetation was noted. The lengths of the revetments were found by uploading the GPS waypoints to ArcGIS™ (ESRI 2014) and running a geometry calculation. Average widths of the river at the assessment sites were found by using aerial photography by ArcGIS™ (ESRI 2014). The SVAP score, revetment length remaining, parcel information, photos from 2014 and original photos from 1995 were converted to one geodatabase for further studies.
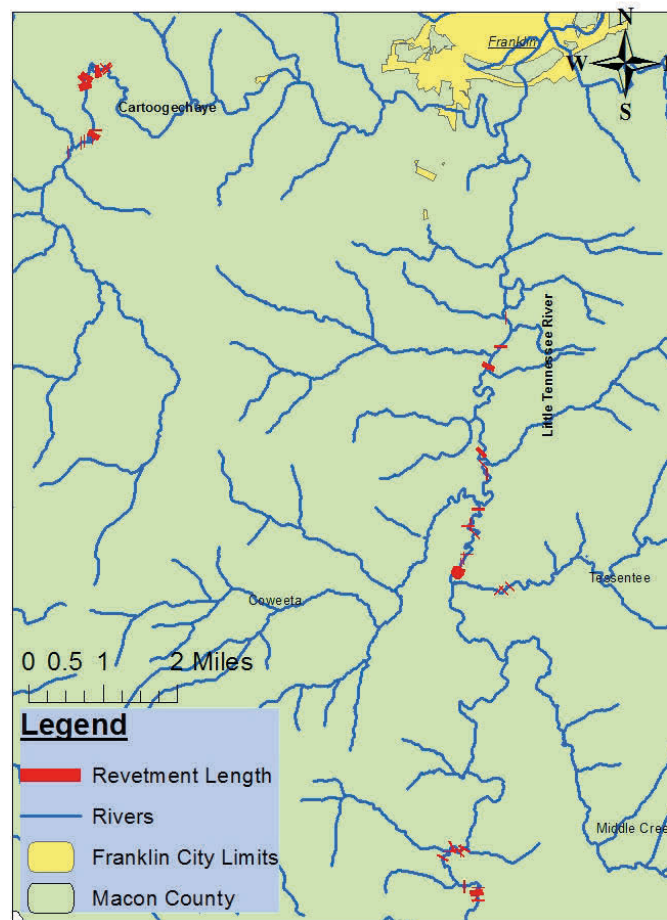
FIG 1. 2014 Revetment assessment site locations in Macon County.

TABLE 1. The average percent of the revetments left and the average Stream Visual Assessment Protocol (SVAP) score of the studied sites within the Little Tennessee River and two of its tributaries, Cartoogechaye Creek and Tessentee Creek. A perfect SVAP score is 20.

River	Average Percentage of Revetment Left	Average SVAP Score
Little Tennessee River	63.95	14.43
Cartoogechaye Creek	82.58	14.46
Tessentee Creek	98.96	15.25

RESULTS

The section of the Little Tennessee River assessed was a sixth and seventh stream order with an average stream width of 75 feet (Wallace 1993). The average width of Cartoogechaye Creek and Tessentee Creek were 45 feet and 25 feet respectively. The average percentage of revetment left for the Little Tennessee River, Cartoogechaye Creek and Tessentee Creek were 63.94 %, 82.58 %, and 98.95 % respectively (Table 1). On average the smaller streams, Cartoogechaye and Tessentee Creeks, have higher percentages of revetment remaining than do the larger Little Tennessee River after about ten years of damage due to flooding events and other natural forces. There was no significant difference in average SVAP score between the Little Tennessee River (14.43), Cartoogechaye Creek (14.46) and Tessentee Creek (15.25; Table 1; Fig. 2). There is a positive correlation between the percentage of revetment remaining and the SVAP score (Fig 2). There is a slight positive correlation between percent of revetment length left and SVAP score (Fig. 2). Of the 54 revetments assessed, 26 of the sites had undercut banks on or around the revetment.
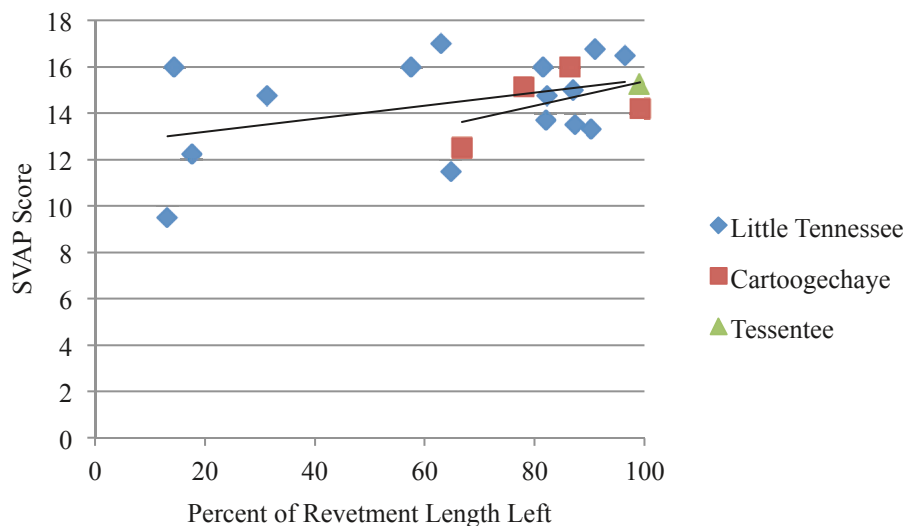
FIG. 2. Percent of revetment length remaining from each site compared to SVAP score for the three streams observed in the Little Tennessee River basin. There is a slight positive correlation between percent of revetment length left and SVAP score as shown by the trend line of the Little Tennessee River and Cartoogechaye Creek.

TABLE 2. Original revetment length compared to percent of revetment length remaining in 2014. River name and average SVAP score noted for each revetment site.

Owner Code	Original Revetment Length (Feet)	New Revetment Length (Feet)	%Revetment Remaining	Average SVAP Score (out of 20)	River
KFarm1	50	43.22	86.44	16	Cartoogechaye
11	1655	1641.04	99.16	14.2	Cartoogechaye
Slag1	1080	842.26	77.99	15.13	Cartoogechaye
LE1	100	87	87	15	Little Tennessee
021	1255	181.2	14.44	16	Little Tennessee
Lois1	160	28.05	17.53	12.25	Little Tennessee
Web1	800	721.83	90.23	13.31	Little Tennessee
BH1	410	337.18	82.24	14.75	Little Tennessee
And1	200	162.9	81.45	16	Little Tennessee
Riv1	350	337.5	96.43	16.5	Little Tennessee
LTLT	495	405.95	82.01	13.7	Little Tennessee
Pot1	790	454.29	57.51	16	Little Tennessee
Tes1	270	267.19	98.96	15.25	Tessentee
Deal	125	78.79	63.03	17	Little Tennessee
Gal1	145	131.8	90.89	16.75	Little Tennessee
Neil1	190	59.55248	31.34	14.75	Little Tennessee
Jes1	240	160.2	66.75	12.5	Cartoogechaye
Dor1	390	51.42	13.18	9.5	Little Tennessee
Bus1	270	174.66	64.69	11.5	Little Tennessee
Teag1	190	165.8	87.26	13.5	Little Tennessee

DISCUSSION

Larger streams have a greater sediment load which provides a base for riparian zone habitats on revetments, but the revetments have a greater chance of being blown out due to the current and variation of the flow of the water. The results of this study, based on the percentage of revetment remaining, indicate that large stream revetments have shorter life spans than those within small streams. I used the SVAP scores to assess the integrity of the revetment and surrounding riparian zones. Wood accumulation and deep pools provide habitat for fish and macroinvertebrates (McLarney 2003). The deposition of large woody debris on the revetment over time alters the stream structure to prevent further bank erosion (Wallace 1993). The overall success of the revetments to stabilize the river banks was the same between a large river and a smaller creek. The positive correlation between the percent of the revetment remaining and SVAP score suggests that successful revetments retain most of their original length (Fig. 2). Revetments in Cartoogechaye Creek and Little Tennessee River and the related SVAP scores both show this trend. I did not have enough revetment data on Tessentee Creek for this comparison because there was only one parcel with revetment work. A revetment that has retained most of its original length will allow recolonization of the bank with vegetation and successfully stabilize the bank. Banks with increased amounts of woody vegetation are less susceptible to erosion. Revetments with low percentages of length remaining were blown out due to erosion by high water velocities and unstable banks. From my survey, I found that whole tree revetments reduce erosion along the banks of streams.

There are disadvantages to using revetments to stabilize stream banks. They can colonize nonnative species including Princess tree (*Paulownia tomentosa),* and Johnson grass (*Sorghum halepense*) (Love 2003, Monar 2011). Revetments are not 100% successful because none of the

revetments in this study received a perfect score of 20 on the SVAP assessment and none retain their original length (Table 2). The 26 revetments that had the presence of undercut banks show that the revetments were not fully effective at preventing erosion. A revetment is difficult or impossible to identify when either it is exceptionally successful with ample amounts of vegetation on top or when it is completely blown out. That being said, there is a measurement error of up to 10 feet due to GPS satellite configuration when the revetment length is calculated, so the success rate of revetments may be higher or lower than the results of this assessment. The line of best fit used by ArcGIS™ to calculate distances is another source of error in the calculations.

Recommendations for Future Assessments

A larger comparison of all of the streams and revetments in the near future would be needed to complete this assessment on the Little Tennessee River and surrounding tributaries. It is important to maintain the revetments and surrounding riparian zone in order to further reduce erosion in the Little Tennessee River basin. This can be done by adding more trees and woody vegetation to the existing revetment.

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LONG-TERM CHANGES IN VEGETATION DYNAMICS FOLLOWING PARTIAL HARVEST IN THE SOUTHERN APPALACHIANS

ANDREA N. STEWART

Abstract. Prior to the 1950s, common partial harvesting operations in the southern Appalachians involved the removal of logs by ground-skidding and the construction of steep access roads and skid trails along stream channels. Little is known about how these historical practices have affected long-term vegetation changes. An experimental watershed in the Coweeta Hydrologic Laboratory (Macon County, North Carolina) was partially harvested in the years 1942 to 1956 according to these practices. Using long-term vegetation surveys, I analyzed patterns in composition, abundance, and diversity of the woody species in this watershed 62 years after treatment. I calculated Shannon's index of diversity based on species biomass (Hˈbiomass) and species richness (S) for each survey and I grouped species into diffuse-porous or ring-porous xylem anatomy categories. I also compared the partially harvested watershed to a watershed that was clearcut in 1977 and to an untreated watershed (mature forest). The partially harvested forest was not different from the mature forest in terms of species composition, total aboveground biomass, basal area, density, and the distribution of xylem anatomy groups. The partially harvested forest had lower Hˈbiomass than the mature forest in the last survey period (2010s). However, the partial harvest did not alter long-term vegetation dynamics as significantly as clear cutting. These results could help forest managers, conservationists, and hydrologists understand the long-term effects of partial harvesting on a forest and allow them to make informed decisions when designing a management strategy.

Key words: *Coweeta Hydrologic Laboratory; diversity; forest composition; forest vegetation; logging; partial harvest.*

INTRODUCTION

Vegetation structure and function in southern Appalachian forests have been highly influenced by human disturbance, particularly logging, over the past several centuries. From the late 1800s to the early 1900s, widespread and intensive logging was common in the region (Brose et al. 2001, Elliott and Swank 2008). Millions of board feet of timber were cut and skidded with cattle and horses, creating vast and desolate landscapes (Douglass and Hoover 1988). This logging disturbance has affected the species composition and diversity of present forests (Elliott and Vose 2011). Chestnut blight (*Cryphonectria parasitica* (Murr.) Barr) also impacted southern Appalachian forests by killing virtually all *Castanea dentata* by the 1940s (Douglass and Hoover 1988, Elliott and Swank 2008). This invasive pest caused a fundamental shift in population and community dynamics by devastating the former oak-chestnut forest association (Braun 1950) with the elimination of the dominant or co-dominant *C. dentata*.

Logging in the southern Appalachians continued throughout the 1900s, and in the 1940s typical harvesting operations removed logs by ground-skidding with horse teams. Steep access roads and skid trails were placed along the path of least resistance adjacent to stream channels (Lieberman and Hoover 1948). Although studies have examined the effects of partial harvest on vegetation (McGee et al. 1999, Schuler and Gillspie 2000, Schuler 2004, Jones et al. 2009, D'Amato et al. 2011, Keyser and Loftis 2012), few have examined the long-term changes in vegetation due to this type of partial harvesting. An experimental watershed in the Coweeta Basin, southwestern North Carolina, was partially harvested in the period 1942 to 1956 to simulate the traditional harvesting practices of the 1940s. The purpose of the original treatment at that time was to examine the effects on stream water

quality (Lieberman and Hoover 1948). In this study, I analyzed patterns in composition, abundance, and diversity of the woody species in this watershed 62 years since the partial harvest.

The primary goal of this study was to evaluate the long-term effect of partial harvesting on vegetation dynamics in the Appalachian Mountains. The primary objectives were to (1) determine the changes in forest composition and diversity in the partially harvested watershed and (2) compare the vegetation changes in a partially harvested forest to a forest with a more severe cutting disturbance and a mature forest. I conducted vegetation surveys August to October 2014 and used historical survey data from three watersheds within the Coweeta Basin, one that was partially harvested in 1942-1956, one that was clearcut in 1977, and one that was untreated (mature forest) since 1923 that serves as a reference watershed.

METHODS

Study Area

Coweeta Hydrologic Laboratory is an experimental forest of the Southern Research Station, USDA Forest Service. Coweeta is located in the Nantahala National Forest, Macon County, North Carolina, in the Blue Ridge Physiographic Province of the southern Appalachian Mountains (latitude 35°03ʹ N, longitude 83°25ʹ W). The elevation ranges from 675 to 1592 m, and slopes range from 30 to over 100 percent. Mean annual temperature is 12.6 °C and mean annual precipitation is 1800 mm (Douglass and Hoover 1988). Soils are deep sandy loams underlain by folded schist and gneiss (Thomas 1996). Vegetation is southern mixed deciduous forest. I selected three watersheds within the Coweeta Basin for this study (Table 1).

TABLE 1. Description of watersheds used in study area (Douglass and Hoover 1988).

Watershed	Treatment	Forest Age (yrs)	Size (ha)	Elevation range (m)	Aspect
WS 10	Partial harvest	62	86	742 – 1159	SSE
WS 7	Clearcut	31	59	722 – 1077	S
WS 14	Reference	88	61	707 – 992	NW

Disturbance History

Prior to the establishment of Coweeta Hydrologic Laboratory, the Coweeta Basin was disturbed by burning and grazing by the Cherokee Indian Nation and European settlers (Douglass and Hoover 1988). Additional disturbances after settlement included Dogwood anthracnose fungal disease *Discula destructiva* Redlin affecting *Cornus florida* (Chellemi et al. 1992), a major drought from 1985 to 1988 (Swift et al. 1989), the southern pine beetle (*Dendroctonus frontalis* Zimmerman) attack on *Pinus rigida* (Smith 1991), large blowdowns caused by hurricanes (Greenberg and McNab 1998), landslides (Hales et al. 2009), and hemlock woolly adelgid (*Adelges tsugae* Annand) (Elliott and Vose 2011). The most dramatic disturbances, however, were logging in the early 1900s and chestnut blight. The J. A. Porter Logging Company held the rights for timber over 38 cm (15 in) and performed heavy selective logging from 1919 to 1923. When the Forest Service took over administration of the Coweeta Basin in 1923, eight million board feet of timber had been removed (Douglass and Hoover 1988). Chestnut blight, common across the region in the early 1900s, was first reported in

Coweeta in 1926. *C. dentata* was the most abundant species in the Coweeta Basin in 1934, contributing 22% of the total density and 36% of the total basal area (Elliott and Swank 2008). By 1940, virtually all *C. dentata* trees were killed (Douglass and Hoover 1988).

While Watershed 14 (WS14) was affected by these disturbances alone, Watershed 10 (WS10) and Watershed 7 (WS7) were subjected to additional disturbance treatments by researchers at Coweeta. WS10 was partially harvested from 1942 to 1956 according to the typical, local logging practices of the time (Lieberman and Hoover 1948). Thirty percent of the total basal area was removed (Douglass and Hoover 1988). WS7 was clearcut in 1977 over a six-month period by tractor skidding and yarding with a mobile cable system. All stems ≥2.5 cm diameter at breast height (dbh, 1.37 m above ground level) were cut and logging debris was left in place with no further site preparation (Elliott et al. 1997).

Vegetation Surveys

I used permanent 0.08 ha (20 m x 40 m) plots to study changes in vegetation patterns in all three watersheds. The plots were established in 1934 along 13 parallel, approximately north-south transects (330°) at 200 m intervals spanning the Coweeta Basin (Douglass and Hoover 1988). All the permanent plots in the Coweeta Basin were originally inventoried in 1934 and three survey periods afterwards (1969-1972, 1988-1993, and 2009-2014 (hereafter 2010s)) (Table 1). All woody stems (trees and shrubs) ≥2.5 cm dbh were measured to the nearest 0.1 cm and recorded by species in the permanent plots. Inventory data from the earlier survey periods has been published for WS7 (Elliott et al. 1997, Elliott et al. 1999). Inventory data for WS10 has never been published and the most recent survey of the permanent plots in WS10 was conducted for this study in 2009 and August to October 2014 (Appendix A). WS14 was inventoried as part of the overall Coweeta Basin surveys (Elliott, unpublished). Locations of WS10, WS7, and WS14 and their permanent plots in the Coweeta Basin are outlined in Figure 1.

Data Analysis

For all species and inventories in each watershed, I calculated density (number of stems per hectare), basal area (m^2 ha^{-1}), foliage biomass (kg ha^{-1}), and total biomass (kg ha^{-1}). I used species-specific allometric equations from Martin et al. (1998) to calculate aboveground biomass (foliage and total) of deciduous trees, equations from Santee and Monk (1981) and Miniat (unpublished) for *Tsuga canadensis*, and equations from McGinty (1972) for *Rhododendron maximum* and *Kalmia latifolia*. To evaluate changes in diversity, I used species richness (S, number of species) and Shannon's index of diversity (H') (Shannon and Weaver 1949), a simple quantitative expression that incorporates both species richness and the evenness of species abundance. H' was calculated on the basis of total biomass (H'biomass). I compared density basal area, foliage biomass, total biomass, S, and H' across watersheds in each year and across time within individual watersheds using the general linear model procedure (PROC GLM, SAS 2010). In order to evaluate the implications of species composition changes on water use and water yield, species were also classified into xylem anatomy groups (ring-porous, diffuse-porous, evergreen, *Robinia pseudoacacia*, and tracheid) to compare the distribution of these groupings over time in each watershed (Taneda and Sperry 2008).

RESULTS

In 1934, the aboveground biomass across all watersheds was statistically similar (Fig. 2). In the 2010s, the aboveground biomass across all watersheds was also similar; analysis of variance revealed that small differences in the 2010s biomass measurements were not significant. Between these years, decreases in biomass were observed in all watersheds due to chestnut blight. Additionally, the biomass decreased in WS10 due to the partial harvest in 1942 to 1956 and in WS7 due to the clearcut in 1977.
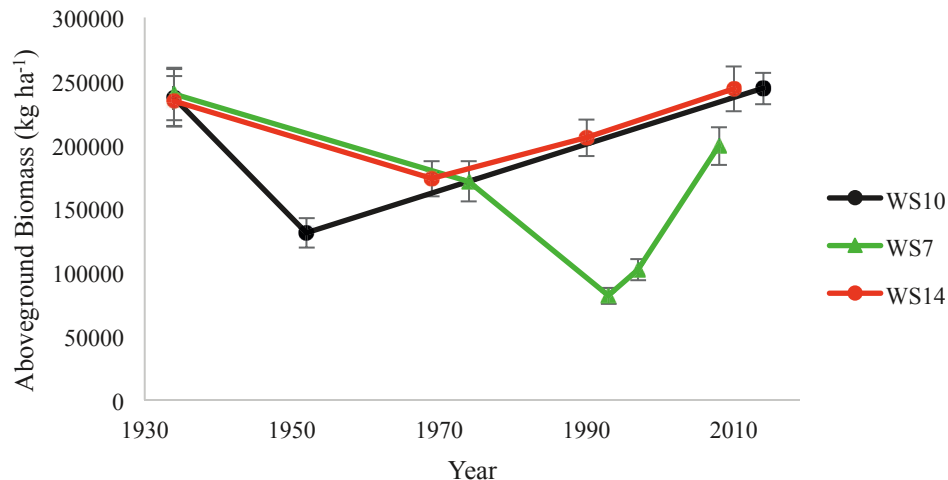
FIG. 2. Change in aboveground biomass for all watersheds since 1934.

H'biomass increased over time in WS10 and WS14 (Fig. 3). H'biomass in WS14 was significantly greater than in WS10 and WS7 in the 2010s based on analysis of covariance. The

increased diversity in WS14 over time can be attributed to its greater species evenness (Fig. 4) rather than changes in S (Table 2).
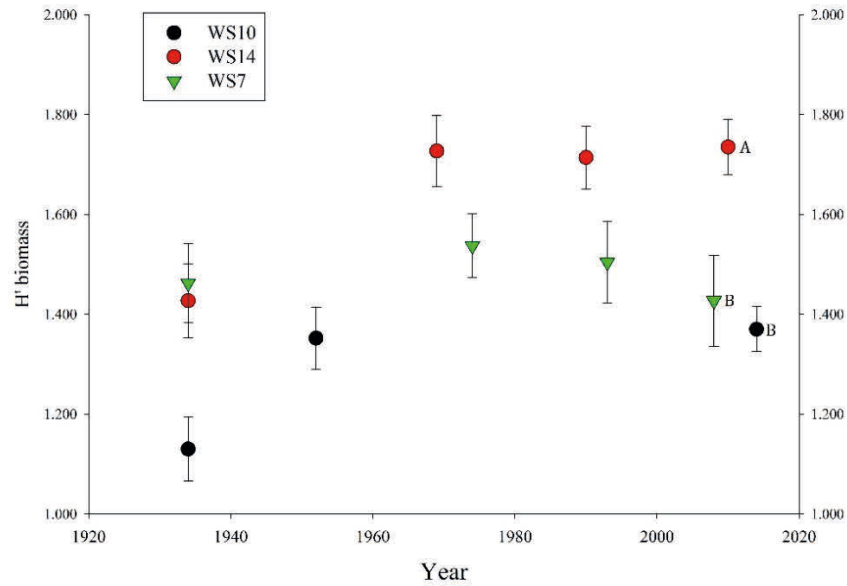
FIG. 3. Changes in diversity (Shannon's index based on total biomass, H˙biomass) for all watersheds since 1934. In the 2010s, values followed by different letters are significantly different (p ≤ 0.05) based on analysis of covariance, where the covariate is initial H˙biomass in 1934 (PROC GLM, SAS 2010).
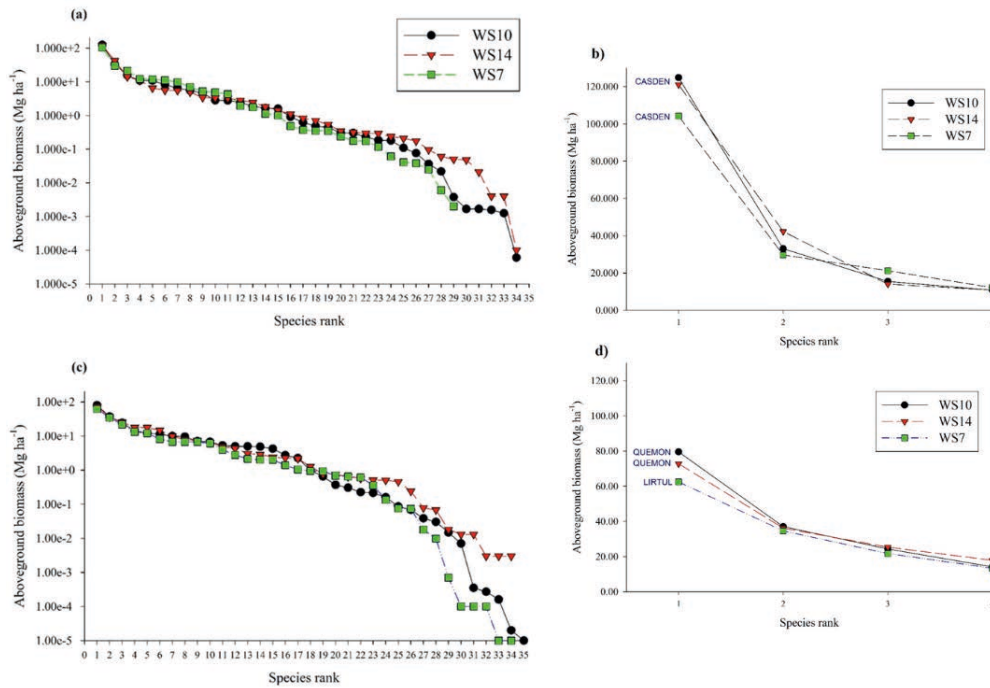
FIG. 4. Dominance diversity curves for all species in all watersheds in a) 1934 and c) the 2010s. Dominance diversity curves for the highest-ranked species in b) 1934 and d) the 2010s.
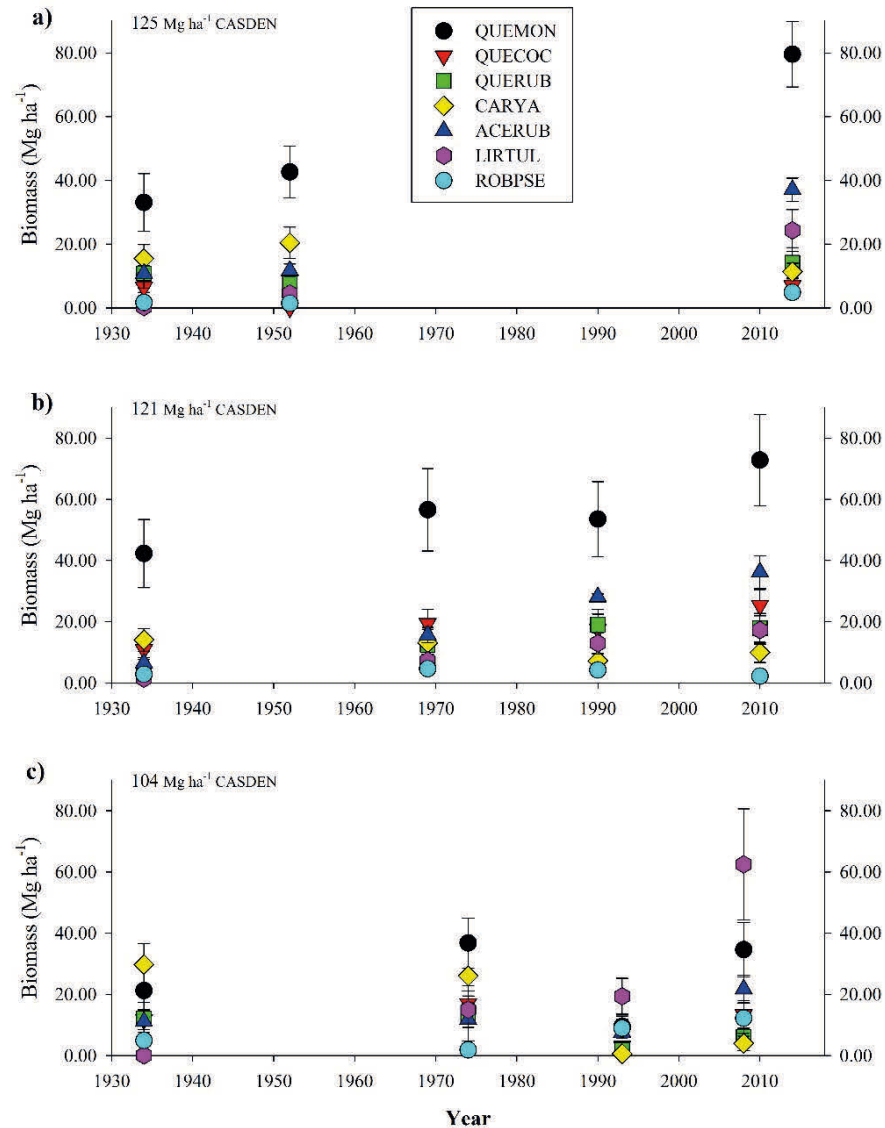
FIG. 5. Species composition over time for all watersheds: a) WS10, b) WS14, and c) WS7. Species codes: CASDEN = *Castanea dentata*; QUEMON = *Quercus montana*; QUECOC = *Quercus coccinea*; QUERUB = *Quercus rubra*; CARYA = *Carya* spp.; ACERUB = *Acer rubrum*; LIRTUL = *Liriodendron tulipifera*; ROBPSE = *Robinia pseudoacacia*.

Dominance diversity curves revealed that the species evenness was lower in 1934 than in the 2010s for all watersheds due to the dominance of *C. dentata* (Fig. 4a,b). All watersheds had greater species evenness in the 2010s due to the replacement of the highly dominant *C. dentata* by other species, including *Quercus montana* and *Liriodendron tulipifera* (Fig. 4d). In the 2010s, the diversity was greatest in WS14 because the least abundant species were more evenly distributed (Fig. 4c).

In 1934, *C. dentata* was the most abundant species in all watersheds (Fig. 5). WS10 and WS14 had a similar pattern of species composition over time; *Q. montana* became the most abundant species after 1934 in both watersheds. The only difference between these two

watersheds was that *L. tulipifera* became slightly more abundant in WS10 than WS14. In WS7, *L. tulipifera* became the most abundant species by the 2010s, with *Q. montana* being the second most abundant species. Species-specific biomass data for WS10 is available in Appendix A.

WS10 was the only watershed that had a greater number of species (S) in the 2010s than in 1934 (Table 2). Based on analysis of covariance, WS10 had significantly fewer species than WS14 (p < 0.05) in the 2010s, but there was no significant difference between WS10 and WS7.

Basal area increased in all three watersheds from 1934 to the 2010s (Table 2). Based on analysis of variance, the 2010s basal area in WS10 was not significantly different from WS14. There was, however, a significant difference between the 2010s basal area in WS7 and WS14, with WS7 having greater basal area than WS14.

The density in WS10 and WS14 did not significantly change from 1934 to the 2010s (Table 2), whereas the density in WS7 significantly increased over time. In the 2010s, the density in WS10 and WS14 was not significantly different, but there was a significant difference between WS7 and WS14, with WS7 having greater density than WS14.

The biomass of diffuse-porous species increased and ring-porous species decreased over time in all three watersheds (Fig. 6). WS10 and WS14 had similar increases in diffuse-porous biomass, while WS7 had even more diffuse-porous biomass. This increase can be attributed to the dominance of *L. tulipifera* in WS7 (Fig. 5), a diffuse-porous species. In the 2010s, both WS10 and WS7 had greater evergreen, *Robinia pseudoacacia*, and tracheid biomass than WS14. WS7 had greater evergreen, *R. pseudoacacia*, and tracheid biomass than WS10 in the 2010s.

TABLE 2. Average density, basal area, foliage biomass, total aboveground biomass (leaf + wood), species richness (S), and Shannon's index of diversity (H') for total biomass for each inventory in each watershed. Standard errors are in parentheses.

WS	Year	No. of Plots	Density (stems ha^{-1})	BA (m^2 ha^{-1})	Foliage Biomass (kg ha^{-1})	Total Biomass (kg ha^{-1})	S	H' Total Biomass
WS 10	1934	44	3010 (128)	29.86 (1.53)	4,490 (216)	236,842 (22512)	11.5 (0.4)	1.130 (0.064)
	1952	40	3762 (249)	21.23 (1.24)	2,658 (160)	130,998 (11637)	13.0 (0.3)	1.352 (0.062)
	2014	43	3378 (306)	36.11 (1.14)	4,255 (157)	244,282 (12423)	12.6 (0.4)	1.370 (0.045)
WS 14	1934	31	3653 163)	29.95 (1.49)	4,997 (308)	234,455 (19613)	16.3 (0.6)	1.427 (0.074)
	1969	31	5979 (498)	25.92 (1.44)	3,501 (203)	173,536 (13798)	17.2 (0.5)	1.727 (0.071)
	1990	31	3835 (558)	29.82 (1.51)	3,857 (192)	205,724 (14441)	14.7 (0.6)	1.714 (0.063)
	2010	30	3672 (322)	36.93 (1.89)	4,157 (227)	243,807 (17569)	14.3 (0.6)	1.735 (0.056)
WS 7	1934	23	2781 (150)	30.12 (1.60)	4,025 (217)	239,871 (20626)	15.5 (0.5)	1.462 (0.079)
	1974	24	2810 (419)	25.31 (1.52)	3,216 (203)	171,419 (15647)	12.7 (0.6)	1.537 (0.064)
	1993	24	18649 (2335)	23.07 (1.36)	3,787 (298)	81,729 (6321)	14.6 (0.7)	1.504 (0.082)
	1997	24	11790 (1293)	27.47 (1.43)	4,101 (335)	102,125 (8277)	13.2 (0.7)	1.478 (0.085)
	2008	24	11989 (149)	42.51 (2.09)	5,808 (370)	199,126 (14756)	13.2 (0.7)	1.427 (0.091)

The partially harvested forest was not different from the mature forest in terms of total aboveground biomass, species composition, diversity in terms of aboveground biomass, basal area, density, and the distribution of xylem anatomy groups. The partially harvested forest was less diverse in terms of total biomass than the mature forest over time. Clear cutting, however, does significantly affect long-term vegetation dynamics in terms of species composition and the distribution of xylem anatomy groups.
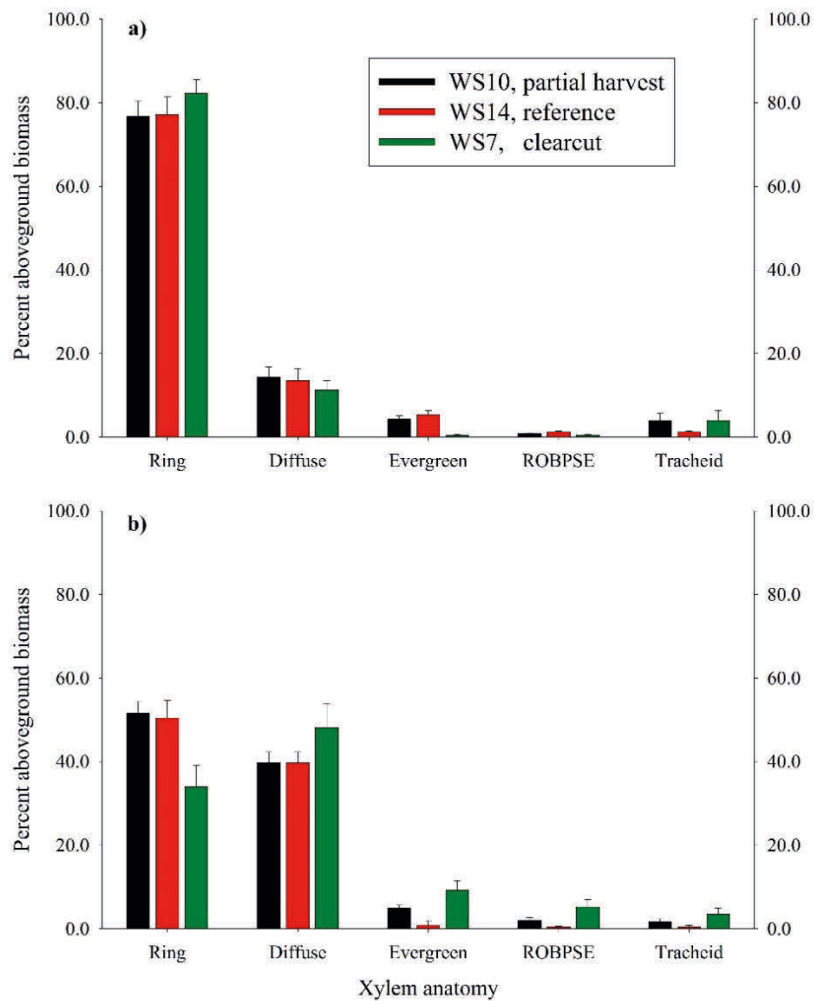
FIG. 6. Distribution of xylem anatomy groups by aboveground biomass in a) 1934 and b) the 2010s for all three watersheds.

In terms of aboveground biomass, WS10 had recovered from the partial harvest, indicating that forests that are partially harvested according to these practices are able to regenerate in a relatively short period of time. This finding is important for forest managers and conservationists because it means that a portion of timber can be removed from a forest by this same method of partial harvesting without affecting its long-term tree viability. Schuler (2004) also found that diameter-limit harvesting in the Appalachians produced average annual productivity rates that were similar to even-aged stands.

Biomass diversity (H'biomass) was higher in the partially harvested forest than in the clearcut forest. Though species evenness increased over time in all watersheds due to the reduction of *C. dentata* biomass from chestnut blight, the mature forest had the highest H' and species evenness in the 2010s. In the partially harvested and clearcut watersheds, the less abundant species were not as evenly distributed. In WS10, these less abundant species included vine and shrubs such as *Lindera benzoin*, *Parthenocissus quinquefolia*, *Vaccinium corymbosum*, *Vaccinium stamineum*, and *Toxicodendron radicans*. Due to the lower abundance of vine and shrub species in WS10, we can perhaps expect partially harvested forests to exhibit less forest structural diversity. One reason this is an important finding is because structural complexity in forest stands has become a recent focus of forest management paradigms related to climate change (Puettmann et al. 2009). However, I did not examine structural diversity in this study. These results differ from Schuler (2004), who found that H' decreased in both a diameter-limit harvested forest and a reference forest, and Schuler and Gillespie (2000), who found that H' increased in a diameter-limit harvesting but decreased in an uncut forest due to decreased species evenness.

Species composition was fairly unaffected by the partial harvest, yet highly affected by the clearcut. Based on my results, the rankings of the most dominant species in a forest can be expected to be similar to a mature forest after partial harvest. If managers partially harvest a forest instead of clear cutting, they can expect to maintain similar species over time. The species composition patterns observed differ from those found by Schuler and Gillespie (2000), who found that partial harvesting caused an increased dominance in shade-tolerant species in the overstory. Abrams and Downs (1990) and Stapanian et al. (1997) observed similar patterns to Schuler and Gillespie (2000), but differences in results may be attributed to different disturbance regimes, harvesting methods, and vegetation communities. These comparison studies also did not incorporate long-term data. My results are similar to Schuler (2004), who also observed a decrease in abundance of *C. dentata*, an increase in abundance of *Acer rubrum*, and an increase in abundance of *Acer saccharum*.

Tree species richness was not diminished by partial harvesting. These results partially agree with Schuler (2004) and Schuler and Gillespie (2000), who also found decreases in S in an uncut forest but instead found increases in S in a diameter-limit harvested forest. The results are similar to Schumann et al. (2003), who found that a partially harvested oak-pine forest in Maine exhibited relatively greater species richness when compared to a control forest. According to my results, forest managers may need not be concerned about partial harvesting reducing the number of species in a forest. Instead, forest managers may need to be concerned with the evenness of the species present.

Implications

Changes in species composition over time due to harvesting has the potential to alter productivity, biogeochemistry, and hydrology. For example, the amount of water used by a plant varies based on its xylem anatomy type, so changes in species composition can potentially alter water use and subsequently alter water yield. Ring-porous wood, for example, has large conduits that embolize in freeze-thaw events, so it has less functional sapwood area and lower water use. Diffuse-porous wood has conduits that retain function year-around, so it has more functional sapwood area and greater water use (Taneda and Sperry 2008). I observed that partial harvesting, unlike clear cutting, does not highly impact the distribution of biomass in xylem anatomy groups,

suggesting that partial harvesting would not reduce the water yield as much as clear cutting. Similarity between WS10 and WS14 indicates that the species composition driving evapotranspiration in these watersheds is similar. This is an important finding because changes in water yield due to forest treatment have significant implications for climate change and domestic water use (Ford et al. 2011). Because climate change impacts on streamflow can be either mitigated or exacerbated by forest management practices involving changes in species composition (Ford et al. 2011), future research should further establish the relationship between partial harvest and water yield within the Coweeta Basin and the greater southern Appalachians.

CONCLUSION

This study revealed that partial harvesting in the southern Appalachians does not alter long-term vegetation dynamics as significantly as clear cutting. Total aboveground biomass, species composition, basal area, density, and the distribution of xylem anatomy groups by biomass (and thus, potentially water yield) were unaffected by partial harvesting, while H' based on biomass decreased relative to the mature forest. Compared to clear cutting, partial harvesting had a lesser long-term impact on the forest. These results may help us understand how historical harvesting practices affected present-day forest composition and diversity, and this understanding could aid forest managers, conservationists, and hydrologists in making informed decisions when designing management strategies for southern Appalachian forests. Researchers at Coweeta should continue to survey the vegetation in these watersheds to understand the impact of partial harvesting over an even longer period of time.

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DISTRIBUTION OF *CRYPHONECTRIA PARASITICA* WITHIN PLOT PLANTINGS OF CHINESE, HYBRID, AND AMERICAN CHESTNUT

JAKE WARD

Abstract. The American Chestnut Foundation has partnered with the United States Forest Service to plant American-Chinese chestnut hybrids that are supposedly resistant to the chestnut blight, *Cryphonetria parasitica*. Several plots of trees were planted in forest clearings from 2009-11, and these trees have been monitored since planting. *Cryphonetria parasitica* reproduces by means of wind dispersed spores. If spores are dispersed over very short distances, then the pattern of *C. parasitica* within a plot should be clumped. Instead, I found a random distribution of *C. parasitica* in all six of the plots that I surveyed.

Key words: American chestnut; Castanea dentata; *Chinese chestnut;* Cryphonectria parasitica.

INTRODUCTION

The American chestnut (*Castanea dentata)* once dominated the well-drained sandy soil slopes of the southern Appalachian Mountains. In the early 20th century, a fungal disease, *Cryphonectria parasitica,* came from Asia to the United States on seedlings for nursery stock of ornamental Japanese chestnut trees (Smith 2012). The disease spread through the populations of American chestnut very rapidly, and by 1950, virtually all of the canopy chestnut trees in the natural range from Maine to Alabama had succumbed to the blight. (Griffin 2000, Ronderos 2000). All that remained were coppiced stumps that could barely grow tall enough to reach the mid-story before being destroyed by the blight (Baird *pers. comm.*).

Cryphonectria parasitica is the fungus that causes chestnut blight and can be identified by its orange-hued fruiting bodies protruding through the outer bark, usually around a large swollen canker on the tree. The fungus produces an acid, to which the tree reacts by swelling and cracking, and the acid causes the demise of the tree. *Cryphonectria parasitica* normally attacks and damages American chestnut but also colonizes many other hardwoods, including *Quercus rubra* (red oak) (Baird 1991). When infected, red oaks may show symptoms by the presence of fruiting bodies, but the result is never as catastrophic as those in the chestnut. Stressed, dying or dead oak trees are more susceptible to chestnut blight than healthy trees (Baird 1991).

In 1983, The American Chestnut Foundation (TACF) was created with the mission to restore the American chestnut back to its former state before the devastating blight took hold (TACF 2014). It was discovered that the blight susceptible American chestnut could be crossed with the fairly resistant, although still vulnerable, Chinese chestnut. While the Chinese chestnut serves as a donor for genetic material to improve resistance, the Chinese trees do not grow as vibrantly as their American counterpart (Knapp 2014). The Wagner Research Farm was established in Meadowview, Virginia in 1989 with the purpose of producing hybrid trees and testing blight resistance (Clark *pers. comm.*). These hybrids were interbred with one another through three generations, and then crossed with another American tree, selecting for blight resistance each time. These back-crossed trees were then interbred and crossed with an American chestnut, once again selecting for blight resistance. After crossing the third backcross trees through three generations, these trees, referred to as B3F3 trees, were 15/16 American chestnut trees, and relatively resistant to *C. parasitica* (Smith 2012). TACF funded the U.S. Forest Service (USFS) and provided them with 20 "families" of B3F3 trees to plant at different experimental plots in the southern Appalachians to determine resistance to blight under natural

conditions and for later use in a possible restoration project (Clark *pers. comm.*). To date, the USFS has planted about 4,500 chestnut trees in three national forests within the Appalachian Mountains of North Carolina, Virginia and Tennessee (Clark et al. 2014).

An infected American chestnut tree will exhibit characteristic symptoms of chestnut blight. The fungus resides just under the outer layer of bark, but as a tree reacts, orange discoloration around the infection is a good indication of blight (Smith 2012). A tree normally reacts to the blight by either swelling - causing the infamous canker - or sinking, which is an indicator of a severe and deadly infection. An infection becomes covered in stromata that appear as orange specks puncturing through the bark of the tree. These stromata contain two types of spores that are necessary for *C. parasitica* to reproduce. Sexual spores called ascospores are forcibly ejected from black, vase-like structures called perithecia in vast quantities and carried by the wind (Smith 2012). Asexual spores called conidia ooze out of round, fruiting bodies called pycnidia after rain or moisture events. Conidia cling to insects, birds or anything mobile in which they come into contact, or simply ooze over the existing canker and undergo self-fertilization (Smith 2012).

I hypothesized that because *Cryphonetria parasitica* reproduces by sporulation and spores are both dispersed by the wind and by oozing, the distribution of blight within a plot should be clumped. If there is a clumped pattern, then it may be easy to assess which American chestnuts are resistant to blight; resistant trees are those that should have been exposed to blight but do not show symptoms or show only superficial symptoms.

<center>METHODS</center>

Sites for planting chestnut seedlings were selected from national forests in the southern Appalachian Mountains in North Carolina, Tennessee, and Virginia (Clark et al. 2014). Most sites were logged before planting, using a shelterwood method where large shelter trees (mostly oak) were left behind for seed and protection from the elements. Native *Castanea dentata* sprouts inhabited nearly all plots, indicating an appropriate location for chestnut seedlings to grow. The layout of the plot depended on available space and the presence of roads, skid trails, rocks, or any other barriers that would prevent a successful or safe seedling planting. A crew used compasses and measuring tapes to set up equally spaced grids at 8 or 12 ft. intervals. At each plot, different stages of hybrids were planted (B1F3-B3F3) as well as different families of hybrids and American chestnut. Only one family of Chinese chestnut was used for this experiment. Tree "family" indicates a common mother tree shared by all saplings of a particular "family". In some cases, plastic mesh cages were used to control deer browsing.

We surveyed several of the USFS experimental plots that were planted in 2009, 2010, and 2011, focusing on plots in North Carolina and Tennessee. These plots contained American, Chinese, and B3F3 chestnut trees, as well as some of the intermediate hybrids (B3F2, B1F3, etc.) We examined each tree in each plot thoroughly for blight symptoms, taking into account any abnormalities, wounds, severe insect damage, or fungi to assign a canker ranking (Table 1). If the tree was free of all blight symptoms, it received a canker rank of one. A small canker, or a questionable area on the bark of the tree that had only a chance of being chestnut blight was also given the rank of 1. A canker rank of 2 meant the canker was definitely affecting the tree but was not fatal; the tissue surrounding the canker was mostly alive, and the canker had not killed many, if any of the stems. A canker rank of 3 was the most severe canker ranking for a living tree, and the canker was very swollen or possibly sunken. A tree with a canker rank of 3 has much dead

TABLE 1. Canker classifications.

Classification	Characterization
1	No blight or not suspected to be blight. Could be from other injury (e.g., cicada, mechanical).
2	Cankered, but not killing the stem, not sunken and sporulation is not very evident, Could be rather superficial, possibly swollen.
3	Sunken and sporulating/dead stem from blight above canker, could be some swelling and cracking but overall canker is covered by sporulation.
4	Tree dead from blight.

tissue around the canker and has possibly experienced major die-back or top-break above the canker. If a tree was killed by chestnut blight, with no sprouting, it received a ranking of 4. We also noted whether the blight was on dead or living portions of the tree. If we gave the tree a ranking of 4 or the tree was dead, we determined if there were fruiting bodies on the corpse, and if there was evidence of reaction to the blight before the tree died (such as swelling or breaking). Reaction to blight before death is important because it can be used to determine if the tree actually died from blight or if blight appeared after the tree had already died.

If blight symptoms were present on a tree, we measured the top and bottom edges of the canker using a telescoping height pole. We estimated the amount of stromata present around the canker or blighted area by percent coverage. We recorded categories of 0 %, <10 %, >10 %, <50 %, or >50 % coverage of fruiting bodies present. We also visually measured the percent of the circumference of the tree that the canker encircled. We took bark samples of the infected trees to confirm or deny the presence of chestnut blight on that particular tree.

In order to assess the distribution of blight within a plot, I mapped each plot in Microsoft Excel, by assigning one tree to each cell and labeling it with its corresponding family (Figs. 1-6). I then assigned a color to the cell according to the following color scheme: white cells represent no blight symptoms, gray cells represent a tree dead from causes other than *C. parasitica*, red cells represent a tree exhibiting blight, and dark red cells represent a tree dead from blight. I examined each map visually to determine the distribution pattern. Each cell in the map is labeled with the type of tree (American chestnut, Chinese chestnut, or a hybrid).

RESULTS

At the Nollichucky Ranger District 2010 site, 228 out of 525 trees died (Fig. 1). This plot had the highest level of infection of any plot we visited. Based on visual inspection, the overall distribution of chestnut blight was random rather than clumped and did not appear to originate at any one point. However, some trees appeared to be clumped, possibly by chance. Not much blight occurred along the long narrow panhandle of the WRD 2009 site, but at the end of the plot where there are more trees there were more chestnut blight infections. There was also a much lower percentage of tree survival on the narrow side of the plot compared to the opposite side. At the NRD 2011 plot (Fig. 2) there is a semicircle on the downhill, eastern side of the plot where blight was absent. There was no apparent difference in unrelated tree death between this section of the plot and the entire plot. The TRD 2009 site (Fig. 3) was the most diseased site with *Cryphonectria parasitica*, although there does not appear to be a pattern associated with the infestation.
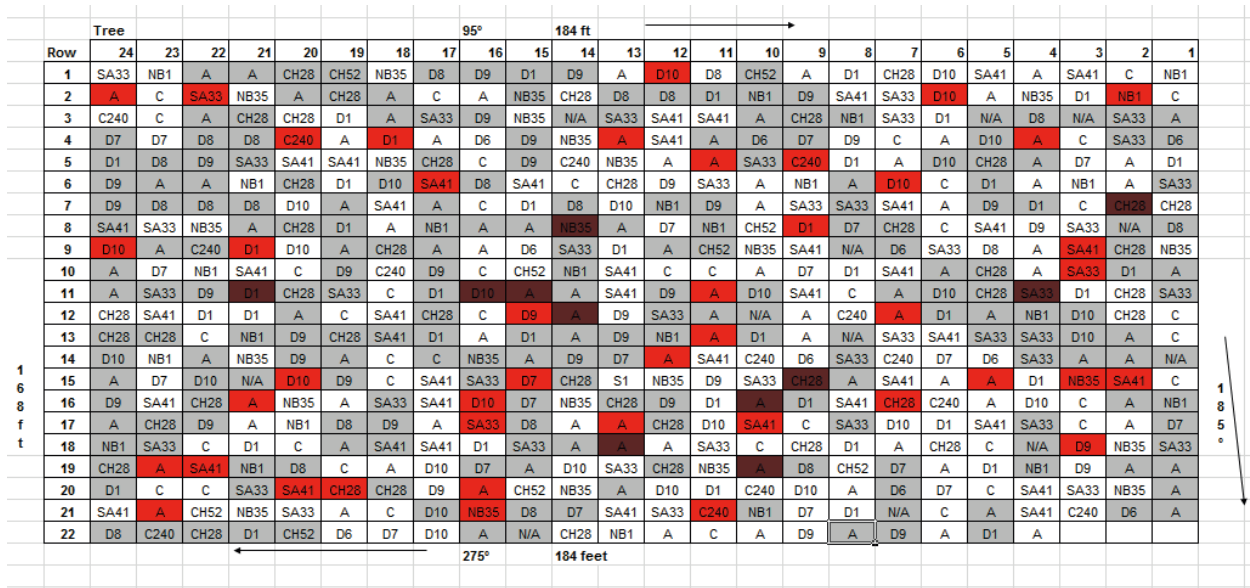
FIG. 1. Nolichucky Ranger District (NRD) 2010 planting. Each cell represents a planted seedling. White cells represent healthy trees, red cells indicate a blight infected tree, dark red cells indicate a tree that has died from blight, and gray cells represent trees that are dead from other causes. The azimuths are noted as well as the row and tree locations.

Tree — 95° — 184 ft

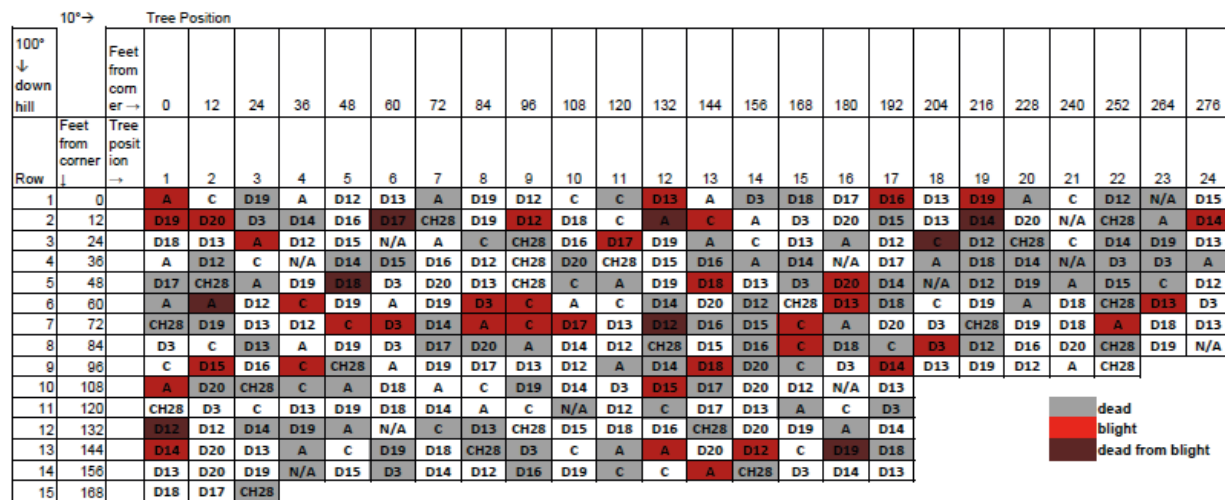| Row	24	23	22	21	20	19	18	17	16	15	14	13	12	11	10	9	8	7	6	5	4	3	2	1
1	SA33	NB1	A	A	CH28	CH52	NB35	D8	D9	D1	D9	A	D10	D8	CH52	A	D1	CH28	D10	SA41	A	SA41	C	NB1
2	A	C	SA33	NB35	A	CH28	A	C	A	NB35	CH28	D8	D8	D1	NB1	D9	SA41	SA33	D10	A	NB35	D1	NB1	C
3	C240	C	A	CH28	CH28	D1	A	SA33	D9	NB35	N/A	SA33	SA41	SA41	A	CH28	NB1	SA33	D1	N/A	D8	N/A	SA33	A
4	D7	D7	D8	D8	C240	A	D1	A	D6	D9	NB35	A	SA41	A	D6	D7	D9	C	A	D10	A	C	SA33	D6
5	D1	D8	D9	SA33	SA41	SA41	NB35	CH28	C	D9	C240	NB35	A	A	SA33	C240	D1	A	D10	CH28	A	D7	A	D1
6	D9	A	A	NB1	CH28	D1	D10	SA41	D8	SA41	C	CH28	D9	SA33	A	NB1	A	D1	C	D1	A	NB1	A	SA33
7	D9	D8	D8	D8	D10	A	SA41	A	C	D1	D8	D10	NB1	D9	A	SA33	SA33	SA41	A	D9	D1	C	CH28	CH28
8	SA41	SA33	NB35	A	CH28	D1	A	NB1	A	A	NB35	A	D7	NB1	CH52	D1	D7	CH28	C	SA41	D9	SA33	N/A	D8
9	D10	A	C240	D1	D10	A	CH28	A	A	D6	SA33	D1	A	CH52	NB35	SA41	N/A	D6	SA33	D8	A	SA41	CH28	NB35
10	A	D7	NB1	SA41	C	D9	C240	D9	C	CH52	NB1	SA41	C	C	A	D7	D1	SA41	A	CH28	A	SA33	D1	A
11	A	SA33	D9	D1	CH28	SA33	C	D1	D10	A	A	SA41	D9	A	D10	SA41	C	A	D10	CH28	SA33	D1	CH28	SA33
12	CH28	SA41	D1	D1	A	C	SA41	CH28	C	D9	A	D9	SA33	A	N/A	A	C240	A	D1	A	NB1	D10	CH28	C
13	CH28	CH28	C	NB1	D9	CH28	SA41	D1	A	D1	A	D9	NB1	A	D1	A	N/A	SA33	SA41	SA33	SA33	D10	A	C
14	D10	NB1	A	NB35	D9	A	C	C	NB35	A	D9	D7	A	SA41	C240	D6	SA33	C240	D7	D6	SA33	A	A	N/A
15	A	D7	D10	D10	D9	C	SA41	SA33	D7	CH28	D9	SA33	D9	SA33	D9	A	CH28	A	SA41	A	A	D1	NB35	SA41
16	D9	SA41	CH28	A	NB35	A	SA33	SA41	D10	D7	NB35	CH28	D9	D1	A	D1	SA41	CH28	C240	A	D10	C	A	NB1
17	A	CH28	D9	A	NB1	D8	D9	A	SA33	D8	A	A	CH28	D10	SA41	C	SA33	D10	D1	SA41	SA33	C	A	D7
18	NB1	SA33	C	D1	C	A	SA41	SA41	D1	SA33	A	A	A	SA33	C	CH28	D1	A	CH28	C	N/A	D9	NB35	SA33
19	CH28	A	SA41	NB1	D8	C	A	D10	D7	A	D10	SA33	CH28	NB35	A	D8	CH52	D7	A	D1	NB1	D9	A	A
20	D1	C	C	SA33	SA41	CH28	CH28	D9	A	CH52	NB35	A	D10	D1	C240	D10	A	D6	D7	C	SA41	SA33	NB35	A
21	SA41	A	CH52	NB35	SA33	A	C	D10	NB35	D8	D7	SA41	SA33	C240	NB1	D7	D1	N/A	C	A	SA41	C240	D6	A
22	D8	C240	CH28	D1	CH52	D6	D7	D10	A	N/A	CH28	NB1	A	C	A	D9	A	D9	A	D1	A			

275° — 184 feet

FIG. 2. Cheoah Ranger District (CHRD) 2011 planting. Each cell represents a planted seedling. White cells represent healthy trees, red cells indicate a blight infected tree, dark red cells indicate a tree that has died from blight, and gray cells represent trees that are dead from other causes. The azimuths are noted as well as the row and tree locations.
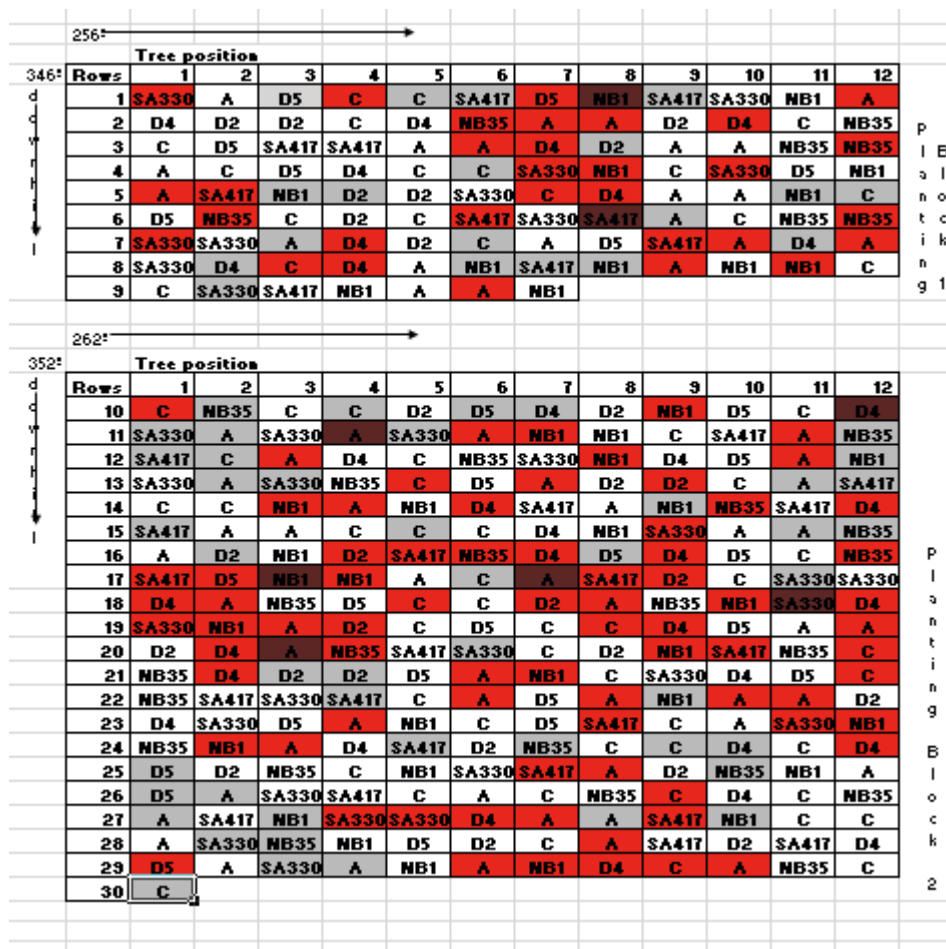
100° down hill — 10° → Tree Position

Row	Feet from corner	0	12	24	36	48	60	72	84	96	108	120	132	144	156	168	180	192	204	216	228	240	252	264	276
	Tree position →	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24
1	0	A	C	D19	A	D12	D13	A	D19	D12	C	C	D13	A	D3	D18	D17	D16	D13	D19	A	C	D12	N/A	D15
2	12	D19	D20	D3	D14	D16	D17	CH28	D19	D12	D18	C	A	C	A	D3	D20	D15	D13	D14	D20	N/A	CH28	A	D14
3	24	D18	D13	A	D12	D15	N/A	A	C	CH28	D16	D17	D19	A	C	D13	A	D12	C	D12	CH28	C	D14	D19	D13
4	36	A	D12	C	N/A	D14	D15	D16	D12	CH28	D20	CH28	D15	D16	A	D14	N/A	D17	A	D18	D14	N/A	D3	D3	A
5	48	D17	CH28	A	D19	D18	D3	D20	D13	CH28	C	A	D19	D18	D13	D3	D20	D14	N/A	D12	D19	A	D15	C	D12
6	60	A	A	D12	C	D19	A	D19	D3	C	A	C	D14	D20	D12	CH28	D13	D18	C	D19	A	D18	CH28	D13	D3
7	72	CH28	D19	D13	D12	C	D3	D14	A	C	D17	D19	C	D17	D13	C	A	D20	D3	CH28	D19	D18	A	D18	D13
8	84	D3	C	D13	A	D19	D3	D17	D20	A	D14	D12	CH28	D15	D16	C	D18	C	D3	D12	D16	D20	CH28	D19	N/A
9	96	C	D15	D16	C	CH28	A	D19	D17	D13	D12	A	D14	D18	D20	C	D3	D14	D13	D19	D12	A	CH28		
10	108	A	D20	CH28	C	A	D18	A	C	D19	D14	D3	D15	D17	D20	D12	N/A	D13							
11	120	CH28	D3	C	D13	D19	D18	D14	A	C	N/A	D12	C	D17	D13	A	C	D3							
12	132	D12	D12	D14	D19	A	N/A	C	D13	CH28	D15	D18	D16	CH28	D20	D19	A	D14							
13	144	D14	D20	D13	A	C	D19	D18	CH28	D3	C	A	A	D20	D12	C	D19	D18							
14	156	D13	D20	D19	N/A	D15	D3	D14	D12	D16	D19	C	C	A	CH28	D3	D14	D13							
15	168	D18	D17	CH28																					

Legend: dead, blight, dead from blight

256⇒

Tree position											
Rows 1	2	3	4	5	6	7	8	9	10	11	12
1 SA330	A	D5	C	C	SA417	D5	NB1	SA417	SA330	NB1	A
2 D4	D2	D2	C	D4	NB35	A	A	D2	D4	C	NB35
3 C	D5	SA417	SA417	A	A	D4	D2	A	A	NB35	NB35
4 A	C	D5	D4	C	C	SA330	NB1	C	SA330	D5	NB1
5 A	SA417	NB1	D2	D2	SA330	C	D4	A	A	NB1	C
6 D5	NB35	C	D2	C	SA417	SA330	SA417	A	C	NB35	NB35
7 SA330	SA330	A	D4	D2	C	A	D5	SA417	A	D4	A
8 SA330	D4	C	D4	A	NB1	SA417	NB1	A	NB1	NB1	C
9 C	SA330	SA417	NB1	A	A	NB1					

262⇒

Tree position											
Rows 1	2	3	4	5	6	7	8	9	10	11	12
10 C	NB35	C	C	D2	D5	D4	D2	NB1	D5	C	D4
11 SA330	A	SA330	A	SA330	A	NB1	NB1	C	SA417	A	NB35
12 SA417	C	A	D4	C	NB35	SA330	NB1	D4	D5	A	NB1
13 SA330	A	SA330	NB35	C	D5	A	D2	D2	C	A	SA417
14 C	C	NB1	A	NB1	D4	SA417	A	NB1	NB35	SA417	D4
15 SA417	A	A	C	C	C	D4	NB1	SA330	A	A	NB35
16 A	D2	NB1	D2	SA417	NB35	D4	D5	D4	D5	C	NB35
17 SA417	D5	NB1	NB1	A	C	A	SA417	D2	C	SA330	SA330
18 D4	A	NB35	D5	C	C	D2	A	NB35	NB1	SA330	D4
19 SA330	NB1	A	D2	C	D5	C	C	D4	D5	A	A
20 D2	D4	A	NB35	SA417	SA330	C	D2	NB1	SA417	NB35	C
21 NB35	D4	D2	D2	D5	A	NB1	C	SA330	D4	D5	C
22 NB35	SA417	SA330	SA417	C	A	D5	A	NB1	A	A	D2
23 D4	SA330	D5	A	NB1	C	D5	SA417	C	A	SA330	NB1
24 NB35	NB1	A	D4	SA417	D2	NB35	C	C	D4	C	D4
25 D5	D2	NB35	C	NB1	SA330	SA417	A	D2	NB35	NB1	A
26 D5	A	SA330	SA417	C	A	C	NB35	C	D4	C	NB35
27 A	SA417	NB1	SA330	SA330	D4	A	A	SA417	NB1	C	C
28 A	SA330	NB35	NB1	D5	D2	C	A	SA417	D2	SA417	D4
29 D5	A	SA330	A	NB1	A	NB1	D4	C	A	NB35	C
30 C											

FIG. 3. Tellico Ranger District (TRD) 2009 planting. Each cell represents a planted seedling. White cells represent healthy trees, red cells indicate a blight infected tree, dark red cells indicate a tree that has died from blight, and gray cells represent trees that are dead from other causes. The azimuths are noted as well as the row and tree locations.
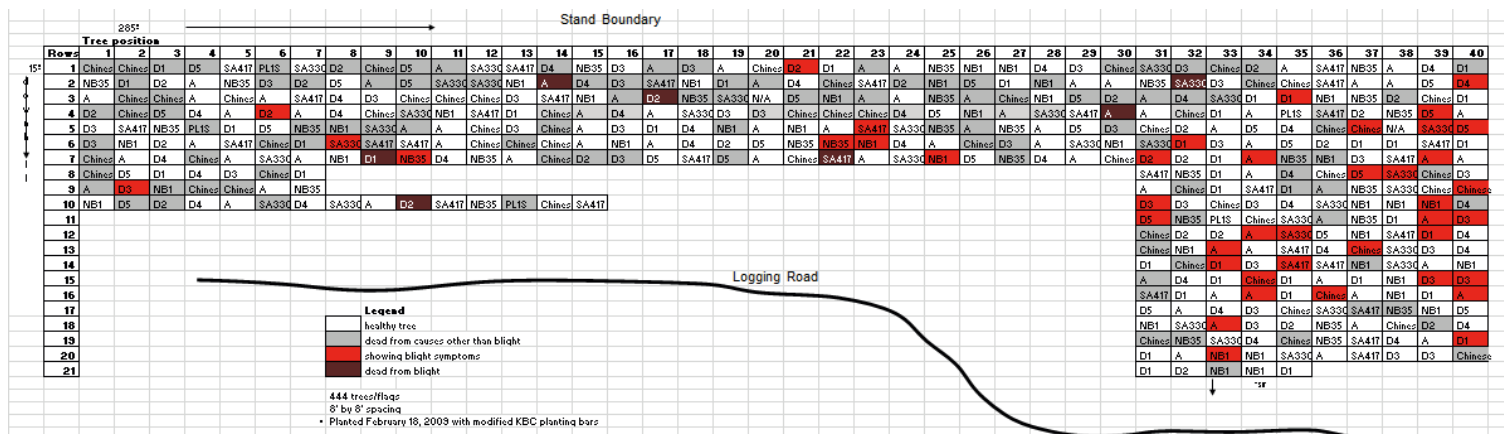
FIG. 4. Watauga Ranger District (WRD) 2009 planting. Each cell represents a planted seedling. White cells represent healthy trees, red cells indicate a blight infected tree, dark red cells indicate a tree that has died from blight, and gray cells represent trees that are dead from other causes. The azimuths are noted as well as the row and tree locations.
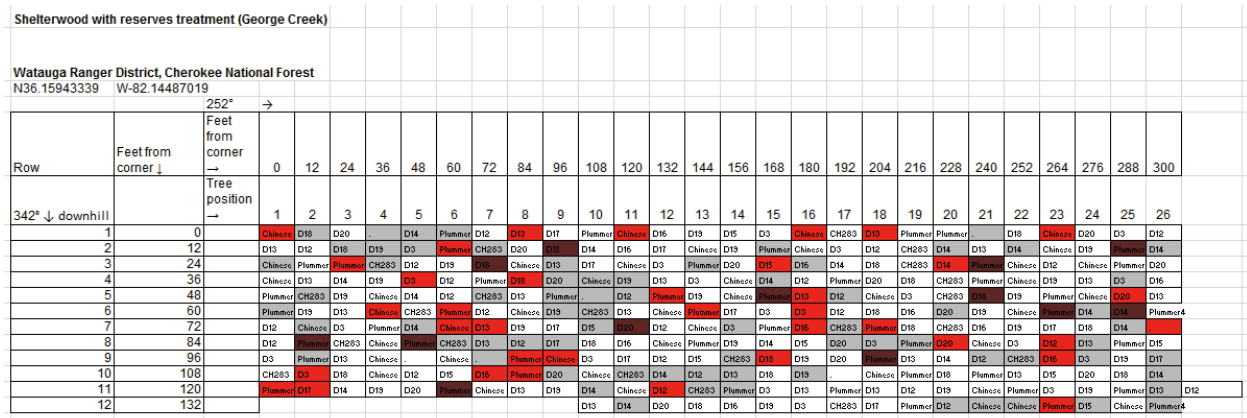
FIG. 5. Watauga Ranger District (WRD) Shelterwood 2011 planting. Each cell represents a planted seedling. White cells represent healthy trees, red cells indicate a blight infected tree, dark red cells indicate a tree that has died from blight, and gray cells represent trees that are dead from other causes. The azimuths are noted as well as the row and tree locations.

DISCUSSION

The TRD 2009 site (Fig. 3) site was split into two sub-plots on either side of a skid trail left behind by logging operations. I was particularly curious about the results from this site because I wanted to see the differences, if any, between the two separated plots. These were not any differences in the level and distribution of blight infestation between these two sub-plots.

The pan-shaped WRD 2009 plot was shaped like this to avoid planting near the logging road near the site (Fig. 4). It has an asymmetrical distribution of blight from the left side of the plot to the right, but there is not a clumped distribution. The amount of blight, however, is proportional between both sides when the number of dead trees are taken into account. The left side of the plot has many more dead trees than the right side of the plot, and also more trees that have died from blight. The portion of the plot with a greater surface area has more deaths related

to causes other than *C. parasitica*. One of the most prevalent causes of tree dieback across most plots is deer browse (Clark *pers. comm.*). Since deer are forest edge creatures, they prefer the openness created during the timber harvest and the narrow strip of vegetation that is the skinny part of the planting plot.

At the NRD 2011 plot, there is a noticeable gap in blight infection on the eastern downhill side of the plot. A follow-up study might be able to determine if this unblighted part of the plot is due to chance or due to particularly resistant trees in this area of the plot. This one site, however, may provide evidence for a complicated distribution where the blight creeps downhill and has just yet to infect trees in this area. Many trees have cankers at the very base of the main stem in this site. The downhill aspect to this hypothesis could be explained by the oozing conidia that can be carried by water during rain (Smith 2012). Further research and experimentation is needed to test this hypothesis.

The number of tree deaths at NRD 2010 exceeds that of any other plot (Fig. 2). The large numbers of tree death in this plot made it difficult to ascertain any pattern of *C. parasitica* distribution. Dead trees could not be examined, and since they never had a chance to be infected by chestnut blight it made gaps in the distribution charts that could not be analyzed.

When I first began this study, I was curious about whether dead or dying trees served as barriers to blight dispersal to the living trees, but I do not believe that they do. Many of the dead trees can be attributed to deer browse or ink disease by the parasitic fungus *Phytopthera cinnamomi* (Clark et. al 2014). Ink disease is named for the black color it turns the roots of the plants when it kills the plant. Because *P. cinnamomi* destroys root stock, the trees cannot recover or resprout when the disease strikes (Clark et al. 2012). *P. cinnamomi* is an aggressive disease that needs to be researched and treated before reforestation of chestnut trees can occur.

These data do not support the hypothesis that the distribution of *C. parasitica* occurs in a clumped pattern. Although there are several patches within plots where blight seems to be concentrated, I cannot conclusively identify a pattern of blight distribution other than a random distribution. In future plantings, I recommend collecting data on the size and condition of the native American chestnuts in or near the future plot. I also recommend monitoring the native *Castanea* trees throughout the experiment as separate data to compare for survivorship among pure American chestnut. There may be a correlation with native tree death in greatly impacted plots like the TRD 2009 site. Also, the sites could be searched for any *C. parasitica* colonies that already exist where the plot is to be established. I think data for the nearest specimen of *C. parasitica* versus the change in blight presence within a plot over-time could help us predict how quickly the blight will spread through a plot. This could eliminate some confounding variables when assessing the resistance of a family or a single tree. For example, every American chestnut tree in any given plot is should be very susceptible to chestnut blight, but every American tree is not infected. Knowing whether or not a tree has been exposed to chestnut blight would provide evidence for resistance.

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THE PRESENCE AND DISTRIBUTION OF RED IMPORTED FIRE ANTS (*SOLENOPSIS INVICTA*) ON THE HIGHLANDS-CASHIERS PLATEAU: CLIMATE CHANGE AND THE SPREAD OF AN INVASIVE SPECIES

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Abstract. The red imported fire ant (*Solenopsis invicta*) is an aggressive, social, invasive species native to Brazil, Uruguay, Paraguay and northern Argentina. The quick dispersal of this species from the port city of Mobile, Alabama at its introduction into the United States in the 1930s across the South, Southeast, West and northward is especially troublesome for native flora, fauna, and humans. Negative ecological effects directly associated with the establishment of the red imported fire ant (RIFA) are well-published and include decimating native ant populations, altering seed dispersal by myrmecochorous plants, altering species interactions, and affecting mammal distributions. The spread of RIFA is also an expensive problem, estimated by the FDA to cost the U.S. close to $5 billion in damages annually. This study examines sites surveyed for RIFA in Macon and Jackson counties, North Carolina, specifically across the Highlands-Cashiers Plateau in Western North Carolina. Mountain topography and a broad range of elevation on the current northern limit of RIFA are key factors leading this study to address a wider issue: the intersection of climate change, elevation, and soil composition and temperature with the spread of invasive species such as RIFA. Data confirms RIFA presence and its distribution across a wide elevation gradient up to 3870 feet. As ant's best friend, humans are literally paving the way for this invasive as it prefers highly disturbed and wide-open areas such as alongside roads, abandoned and cleared lots, and suburban backyards.

Key Words: Bruen; cleared area; climate change; dispersal; elevation; fire ants; forest fragmentation; GIS; habitat variables; invasive species; Jackson County; Macon County; niche differentiation; pest; plant ecology; predictive model; range limit; red imported fire ant; RIFA; roads; soil moisture; Solenopsis invicta*; southern Appalachians; spread; temperature; western North Carolina.*

INTRODUCTION

Solenopsis invicta, the red imported fire ant, is a major threat to humans, fauna, and flora across the southwestern, southern, and southeastern United States. Since its introduction into the United States, *S. invicta* has effectively outcompeted and displaced the previously predominate native fire ant species of the U.S., *Solenopsis xyloni* (Jacobson et al. 2006). *S. invicta* is also one of two species of fire ants introduced into the U.S. from South America, the other being *S. richteri*, the black imported fire ant (James et al. 2002). However, *S. invicta* has outcompeted *S. richteri* in almost all areas of dispersal (Tschinkel 2006).

S. invicta wreaks damage both as an agricultural pest as well as an ecological wonder, seemingly invincible in its territorial conquest in temperate and tropical areas across the world. Remarkably, in 1905, H. G. Wells accurately predicted RIFA global dispersal in *The Empire of the Ants*: "And why should they stop at tropical South America? … Suppose they go on spreading! Suppose they come down the river to the sea and send off an expedition in the hold of some eastward-travelling ship! What could one do?" Sure enough, the red imported fire ant (hereafter referred to by its acronym 'RIFA') landed in Mobile, Alabama in the 1930s, presumably from a cargo ship that was importing goods from South America (Fig. 1).
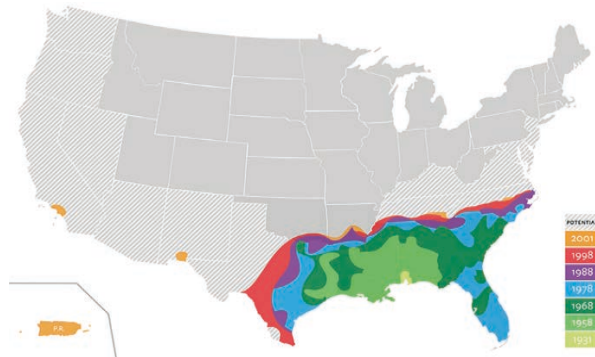
Fɪɢ. 1. Historical dispersal of RIFA in the United States – 1931 to 2001 (Hoey 2013).

The native range of RIFA is the central floodplains of South America, along the Paraguay River in Argentina and Brazil (Gotelli and Arnett 2000). Within this area, the ant is not nearly as dominant as in the U.S., as it competes with quite a variety of native ants and its population densities are also limited by biotic factors such as parasites, viruses, and bacteria (Taber 2000). Within the U.S., to date, there has only been one successful form of growth control, though limited in scope. It is a biological control agent in the form of host-specific phorid flies, *Pseudacteon tricuspis* and *Pseudacteon curvatus*, also known as zombie flies, which lay eggs inside RIFA, which then grow and decapitate RIFA, after which they mature, multiply, and continue to kill RIFA until reaching the queen(s). The program is currently under development by the USDA (Mathieu 2013).

RIFA dispersal is unique as the colony has adapted over evolutionary time to form a water-resistant mat in the face of otherwise catastrophic mortality when its habitat is flooded. As their native habitat is within floodplains, water is a means of colony dispersal along with wind and human transportation. In the floating mat, their mandibles act like hydraulic jacks and their legs lock in one of many demonstrated acts of swarm intelligence. Swarm intelligence is a function of their collective cooperation and superorganism status (Hölldobler and Wilson 1990). Pheromones are the primary means for the RIFA to communicate as one, whether they are stinging at the same time, defending their nest, or laying tracks for finding food that has been captured or killed far from the colony (Hölldobler and Wilson 1990). The mandibles and its notably sharp, needle-like stinger at the end of the gaster, make this ant a truly tough insect. The ant will lock its jaws onto an enemy and sting repeatedly in a circle-like motion (Taber 2000). In numbers, the formic acid and venom from their sting will kill most insects and even small mammals. Calves are known to freeze in the face of danger and this instinct plays to the ant's liking as they can swarm and kill the calf where they stand (Taber 2000).

The RIFA's impact on humans is varied, ranging from agricultural losses to anaphylactic shock and allergies for some. It has a sometimes fatal, insatiable hunger for electrical wires and equipment, shorting the circuits in many HVAC systems, stop lights, and other electrically-wired equipment. When it comes to aiding its spread, this ant's best friend is man. Cleared forest, agricultural land, and roads are perfect for the RIFA. It is also partly transported by humans within hay, fertilizer, soils, nursery stock, and pickup truck beds with standing water (Tschinkel 2006). Nationally, the bill approaches $5 billion dollars to cope with this invasive species (McDonald 2006). Although they are responsible for a host of damages and continue to disperse northward and westward, their special adaptations, such as the floating ant mat and their swarm intelligence, can be studied to discover advances in physiology and robotics (Watchulonis 2012).

Evolutionarily, they are a marvel and are one of the most studied superorganisms (or organisms) on the planet.

Along with in the U.S., RIFA has become a pest in Australia, the Caribbean, Taiwan, Hong Kong (Ascunce et al. 2011), and the southern Chinese provinces of Guangdong, Guangxi, Fujian, and Macau (Ling et al. 2005). RIFA spreads long distances through global trade, but locally the colonies grow and spread as a result of the queen or queens. There are two different types of colonies, monogyne and polygyne. Monogyne colonies have one egg-laying queen, while polygyne colonies have two or more (also known as pleometrosis). Polygyne colonies can undergo 'budding,' where one or more mated queens leave and set up new colonies relatively close by (Hölldobler and Wilson 1990). All colonies produce winged virgin queens, also called female alates, as part of the reproductive cycle of the colony. These virgin queens, at a designated time in a mature colony's development, will fly several hundred feet into the air and mate with male alates (Hölldobler and Wilson 1990). While the males die shortly afterward, the new queens fly (or are transported by water) up to several miles to find a suitable colony site and establish a new colony. The average longevity of RIFA queens is 5.8 to 6.8 years (Hölldobler and Wilson 1990).

Since the 1930s, the RIFA has spread consistently northward (Hoey 2013). Within North Carolina, much of the state is under federal quarantine in regard to RIFA and the inspection and transport of goods is regulated in an attempt to confine the spread of this invasive species (darkly-shaded counties in Fig. 2a and counties shaded red in Fig. 2b).
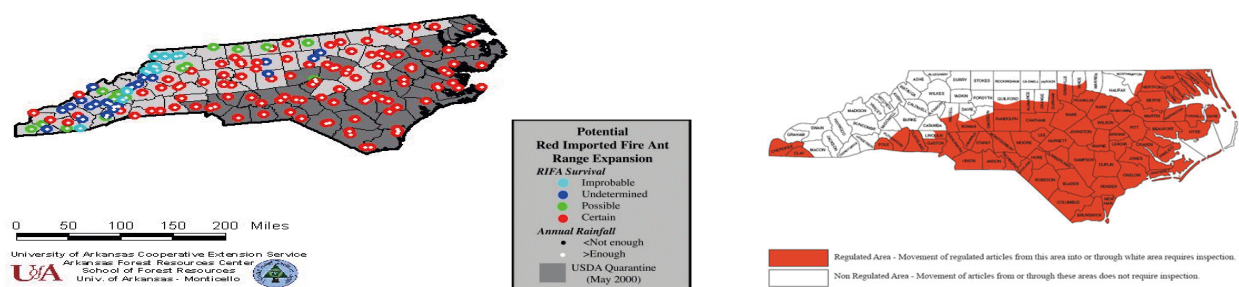
FIG. 2. (a) Left. Potential RIFA range expansion and USDA federal quarantine as of May 2000 within the state of North Carolina (USDA and University of Arkansas 2006). (b) Right. RIFA USDA federal quarantine as of 2010 within the state of North Carolina (Apperson and Waldvogel 2010).

The goal of this study was to document the presence of RIFA within Macon and Jackson counties, North Carolina. As of the time of this study, there was a lack of data regarding RIFA distribution in Macon and Jackson counties. The study was performed over the course of three months, September, October, and November, during the fall of 2014. These counties lay in the very southwest of the state of North Carolina and have a wide range of elevation along with highly variable microclimates, geologic and topographical formations (Costa, pers. comm.). With climate change currently altering average temperatures slightly and in the future potentially altering temperatures at a greater rate, the dispersal of RIFA has and will have a serious impact on many new counties and states. Although until recently it was thought that RIFA could not successfully colonize mountainous regions at higher elevations (Rosson 2004), these ants are adapting quickly. Therefore, this study surveyed for RIFA within a variety of elevations as a result of this importance and provided a further assessment of elevational distribution of RIFA in Macon and Jackson counties, North Carolina.

METHODS

69

Site Selection

A total of 211 sites were surveyed within Macon and Jackson counties for the presence and distribution of RIFA. Within these sites, 282 GPS waypoints were marked as there were some survey sites containing multiple ants and/or colonies. Sites were selected on the basis of easy accessibility to main roads, though considerable effort was taken to sample across a wide variety of areas, land covers, and elevations. Cleared areas, which are more suitable to RIFA (Taber 2000), were mainly surveyed. Site selection was therefore not done in a systematic way, rather was based upon methodological guidance from Dr. James Costa of Highlands Biological Station as to where RIFA should be pursued (Costa, pers. comm.). Areas around roads, parking lots, sidewalks, ball fields, community parks and buildings, and storage units were the most common sites surveyed. The rectangular survey area for each site averaged 600 feet by 200 feet or 2.75 acres. The area was surveyed by means of a visual scan while walking. Any dirt that looked like it harbored ant colonies was disturbed and examined in greater detail.

Specimen Collection

Specimens were preserved in glass vials half-filled with 80 to 95% ethyl alcohol. Aluminum forceps were used to capture approximately three RIFA individuals from each colony upon disruption of the mound. The ants were identified under a stereomicroscope with two auxiliary double goose-neck cold light sources at Highlands Biological Station in Highlands, North Carolina. RIFA identification was confirmed at the University of Georgia Fire Ant Lab by Dr. Ken Ross and doctoral student Pablo Chialvo (Ross and Chialvo, pers. comm.).

GPS

The GPS coordinates from each site were logged using a Garmin Oregon 450t portable personal GPS unit (Garmin 2010). When no ants were found at a location, a single GPS coordinate (waypoint) was obtained approximately at the center of the survey site. Single GPS coordinates were marked directly on top of mounds (while making sure to lean over mounds and not stand in the fire ant mound) as well as on top of independent foraging ants when found. Glass vials with ant specimens were labeled with the number of the corresponding GPS coordinate. This way, when RIFA was identified in the lab, the waypoint number was highlighted as RIFA within ArcMap for further analysis.

GIS

The GPS waypoints were loaded onto a computer via DNR GPS Freeware (Minnesota Department of Natural Resources 2014), a software program for loading Garmin GPS points and track logs. They were saved as a shapefile and loaded into ArcMap, within ArcGIS Desktop 10.2 (ESRI 2014). Various layers such as county elevation, hours of sunshine, and soils, were downloaded from governmental websites and loaded onto ArcMap. Macon and Jackson counties were used as a spatial mask to bound layers to the study area. Spatial analysis tools, including Extract by Mask, Extract by Points, Extract by Polygon, Join, and Create Layer from Selections, were used in geographical analysis operations to make output maps and charts related to the study. The add-in GIS application *Feature Analyst 5.0* (Overwatch Systems, Inc. 2014) was used

for automatic, supervised learning and subsequent feature selection. The program used the input of RIFA mound sites and found land cover areas (30-meter resolution pixel cells) within a raster orthoimagery layer that were most like the land cover area that held the RIFA mounds. The input representation selection of bull's eye was used to focus only on pixel cells in which mound points were found. Within the feature selector, narrow linear feature (< 10m) was selected and the ouput is a raster representation. Within post-processing options, aggregate small regions with a minimum of 100 square meters was selected. The orthoimagery was provided by NC OneMap (North Carolina Center for Geographic Information and Analysis 2014) and NASA *Landsat ETM+* (NASA Landsat Program 2003). Adobe PDF Reader XI is used to view the final outputs (Adobe Systems 2014). Microsoft Excel 2013 was used for data management (normalization) and creation of charts and graphs (Microsoft Corporation 2013).

RESULTS

Survey

The field survey included a total of 211 sites, 115 in Macon County and 96 in Jackson County. A total of 68 RIFA mounds were found within the 211 sites surveyed. Among the 211 sites surveyed, 47 (22.27%) contained some type of ant and 17 (8.06%) had RIFA colonies. While Macon County had 15 (13.04%) sites with at least one RIFA colony, Jackson County had only two sites (2.08%) with at least one RIFA colony. There were four sites with high densities of colonies present, three of them in Macon County. A notable high-density site in Macon County is featured in Figure 5a. The high-density site in Jackson County is shown in Figure 5b.
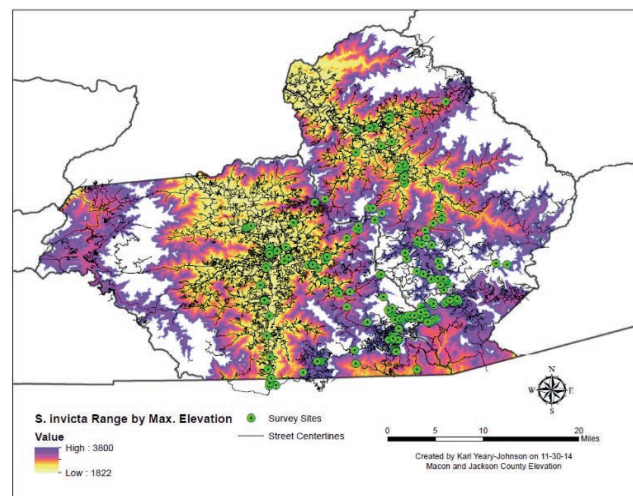
FIG. 3. Study survey sites overlaid on predicted maximum RIFA range by elevation – Macon and Jackson counties, North Carolina. Created in ArcGIS 10.2 (ESRI 2014).
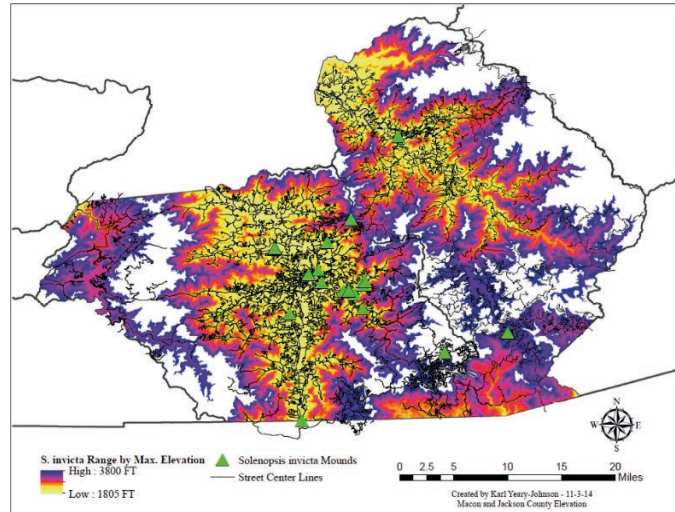
Fig. 4. RIFA mounds found during survey overlaid on predicted maximum RIFA range by elevation – Macon and Jackson counties, North Carolina. Created in ArcGIS 10.2 (ESRI 2014).
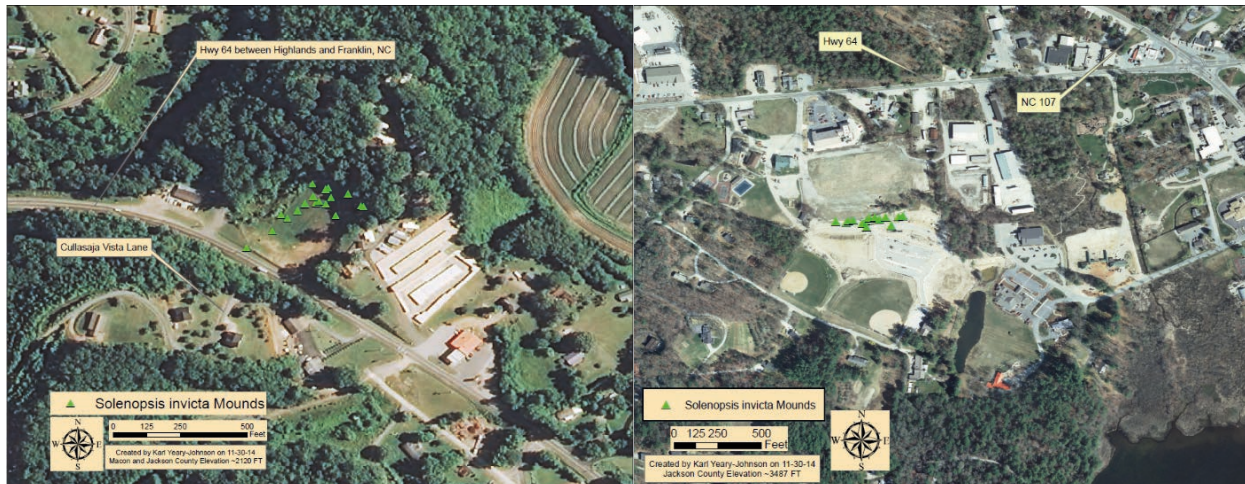
Fig. 5. (a) Left. High density of RIFA mounds between Highlands and Franklin, NC on US Highway 64. (b) Right. High density of RIFA mounds at Cashiers Community Park near US Highway 64 and NC 107. Created in ArcGIS 10.2 (ESRI 2014).

Elevational Diversity Gradient

The survey was conducted across the full elevation range of Macon and Jackson counties (Fig. 7a). The mean elevation for all sites surveyed was close to the median, helping the study maintain its objective of consistent and representative elevation gradient assessment. Among the RIFA mounds found, the mounds at highest elevation were 3872 feet in Macon County and 3489 feet in Jackson County. The average elevation among the 68 RIFA mounds found was 2499 feet. Approximately three-fourths of the mounds were found between elevations of 2000 feet and 2200 feet, however, the other fourth was found above 3000 feet, mostly at 3500 feet (Fig. 7b).
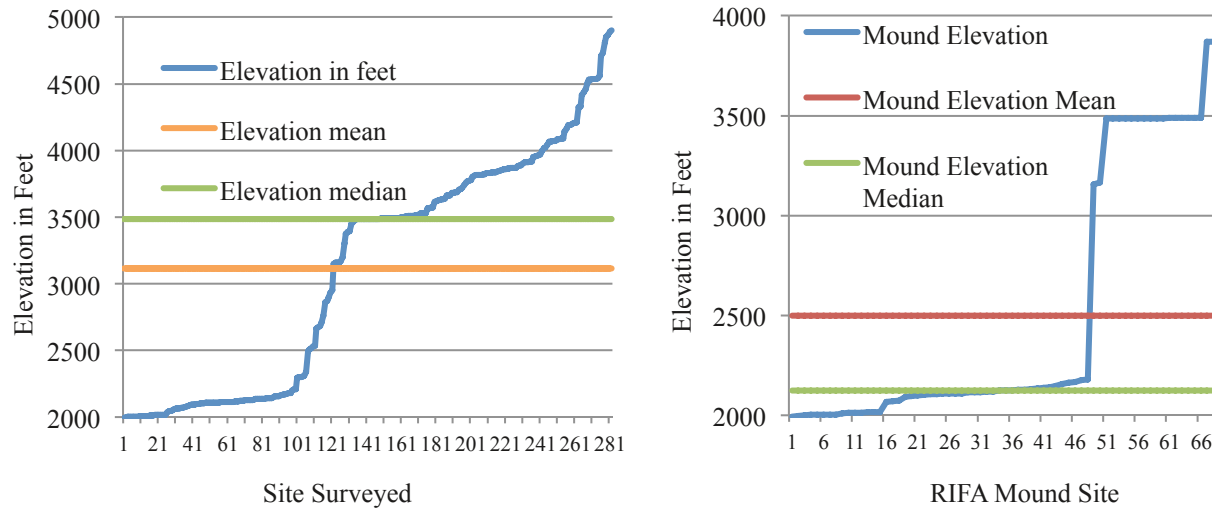
FIG. 7. (a) Left. Jackson and Macon County survey sites by elevation with median and mean elevation. (b) Right. Jackson and Macon County RIFA mound sites by elevation.

Predominant Habitat

RIFA colonies were found predominantly in grassy areas, usually alongside major roadways (Fig. 9a). Recently cleared land plots over five acres in size were the most typical habitat type. Most colonies were close by water (Fig. 9b). Some colony sizes were small (e.g. six inches by six inches wide and one inch tall) and some were large (e.g. four feet long by three feet wide and eight inches tall). The average was approximately one and a half feet by one and a half feet wide and four inches tall. The sites with high RIFA mound densities were found in recently disturbed areas that were cleared, in full sun, close to major state highways like US Highway 64, and were in grass approximately six to twelve inches tall. They did not generally successfully colonize in regularly mowed grass less than four inches tall.

FIG. 9. (a) Right. Typical RIFA mound found in occasionally mowed grass within 500 feet of a major state roadway. (b) Left. High-density site (colonies against inside of fence, not in recently mowed area) corresponding to FIG. 5b in Cashiers, NC. Note water catchment area with rocks to right side of photo. All photographs were taken by the author.

Soil Type

Soil is an important variable for RIFA colonization, colony growth, mortality, and dispersal. Soil temperature and moisture are main factors in success of the RIFA (Tschinkel 2006). I surveyed many soil types for RIFA (Fig. 10a) and found RIFA mounds in several different soil types (Fig. 10b). WtB is a type B hydric soil (soil which is permanently or seasonally saturated by water, resulting in anaerobic conditions). It refers to Woodtell loam, a well-drained, slowly permeable deep soil class with very fine sandy-loam and clays underneath the surface layer. EvE refers to Evard-Cowee complex. Although EvE is not classified as a hydric soil, it is also well-drained, slowly permeable deep soil mostly comprised of sandy loam with clays underneath the surface layer. Along with EvE soil, WtB soil comprised approximately half of the soil where RIFA mounds were found. Developed and highly developed land cover types and soils relate with successful RIFA colony distribution. These soils are associated with more developed areas. (Thomas 1996).
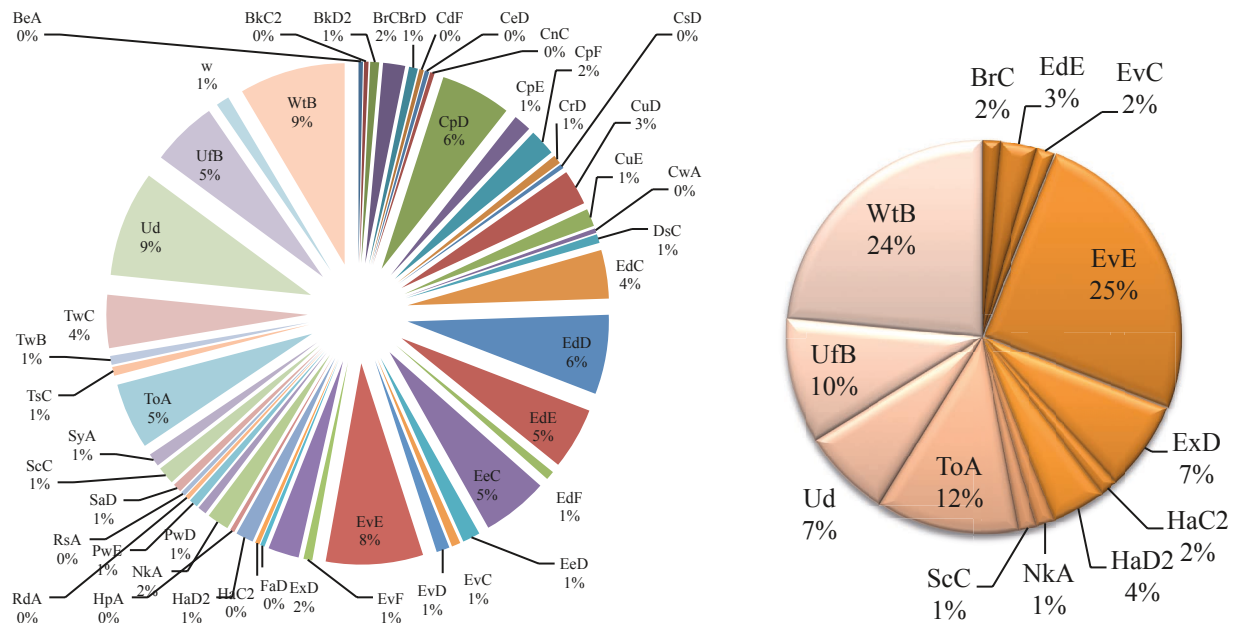
FIG. 10. (a) Left. Survey site by soil type (NRCS 1996, 1997). (b) Right. RIFA mound site by soil type (NRCS 1996, 1997).

DISCUSSION

Abiotic and Biotic Factors Affecting Colony Growth and Dispersal

At their range limits, which encompasses at least some of the areas surveyed, RIFA dispersal is affected by abiotic and biotic factors such as air and soil temperature, competition and predation, and suitable habitat (Rosson 2004). Although the RIFA is mainly in the landscape spread stage (Fig. 12), its successful dispersal is a function of a combination of factors and is in a constant flux among several of these stages in different areas within the study area. Within this model, climate change is a major variable that affects all stages of an invasive species' dispersal (Hellmann et al. 2008).
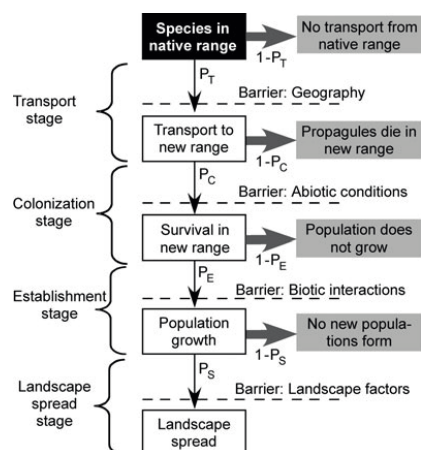
FIG. 12. Conceptual model of the process of species invasion. Transition probabilities between four distinct stages of invasion are marked Px. Arrows indicate key transitions that could be affected by climate change (Hellmann et al. 2008).

Among abiotic factors, temperature is a major limiting factor of RIFA dispersal. At air temperatures less than 17°C, eggs cease to develop. Colony growth occurs when temperatures are between 24 and 36°C, and experience optimal growth at 32°C (Korzukhin et al. 2001). Additionally, a hard winter can kill RIFA. Studies have shown that a minimum of two weeks of temperatures lower than -12.2°C would be needed to have a kill significant enough to affect numbers of RIFA colonies (Loftin and Hopkins 2009). Talking to a number of land owners on the Highlands-Cashiers Plateau, many exclaimed that they had RIFA mounds on their land before a notably hard winter during the 2013-2014 season. Therefore, the data I obtained may be a very conservative data set for the RIFA in this area. Further, I roughly estimate that I found 15% of the RIFA colonies in Macon and Jackson counties during the months this study was conducted as I was limited by time and resource constraints.

Humans play an important role in the dispersal of RIFA as they provide more and more ideal habitat for the ants by clearing forests and other natural landscapes to build new roads, homes, cropland, and other manmade features. However, RIFA can occur at relatively high densities in all grassland habitats regardless of disturbance history and thus does not require disturbance to invade natural grassland habitats (LeBrun et al. 2012), such as those found further south outside of the study area in close-by southeastern states like Alabama, Georgia, Florida, and South Carolina. Additionally, tunnels have been found to a depth of 10 feet in some soils. Soil type also has an impact on the depth of the tunnels. For example, clay soils will have deeper tunnels than sandy soils (Loftin and Hopkins 2009). Because the majority of soils in which RIFA were found were comprised of sandy soils with clay subsoil, the tunnels of RIFA colonies in the study area are estimated to be of medium to deep depth, perhaps three to seven feet deep.

The modification of surrounding lands by humans, whether it be a conduit for irrigation or urbanization, creates wetter and warmer conditions that increase the susceptibility of an area to invasion (Tschinkel 2006). Recent work on invasive ants suggests that invasion success in this group is in some situations governed more by abiotic suitability than by biotic resistance, particularly with respect to resistance from the native biota (Krushelnycky et al. 2005). RIFA is a synanthrope – it flourishes in the presence of human activity and needs alteration of the natural environment by humans to continue to disperse (Mathieu 2013). Preventing habitat disturbance is the most important way to reduce fire ant impacts. High mound densities are found in

landscape that has been fragmented and disturbed by humans (Fig. 5a and 5b). At both high mound density sites, the surrounding area was cleared and grasses were planted by humans.

RIFA: Characteristics of Success

RIFA is an aggressive, generalist species that dominates in disturbed habitats. The average life span of colonies in the United States is seven to eight years (Tschinkel 2006). Characteristics of RIFA that enable it to thrive in new environments include: aggressive colony defense, a painful sting, the relocation of the colony should conditions become unfavorable (or should better habitat be found elsewhere), dispersal by flight and budding (radial expansion of colonies), the presence of monogyne and polygyne colonies (as average worker size varies drastically between the two types), and dominance of food resources (Scanlan and Vanderwoude 2006).

The formation of ant rafts should flooding events occur is a unique adaptation of RIFA. They are also able to dig tunnels (called galleries) deep enough to reach the water table should drought conditions exist (Taber 2000). Although these evolutionary adaptations took a great deal of time to develop, it is quite possible RIFA is currently in the process of evolving to survive in colder climates and higher elevations as shown by this study. Most ants, including RIFA, are highly thermophilic (Hölldobler and Wilson 1990). Mating flights by RIFA virgin queens do not occur when surface soil temperatures are below 18°C and colony founding by newly mated queens only occurs if soil temperatures at five to seven centimeters below the surface is equal or greater than 24°C (Hölldobler and Wilson 1990). Additionally, cold temperatures are most likely a powerful selective pressure, selecting for hardier individuals, ameliorating behaviors, and/or genetic alterations (Costa, pers. comm.).

Other special characteristics of RIFA are unicoloniality, a social structure of ants in which the workers can move freely between different nests, and aphid farming (Mathieu 2013). As there are few arboreal ant species in the United States, RIFA take advantage of this niche opportunity and work with aphids. They consume the aphids' excess honeydew and in return protect them. This has led to proliferation of aphids on some crops (Mathieu 2013).

Future Prediction of Dispersal and the Role of Climate Change

According to a study incorporating a number of environmental and climate change variables, predictions indicate that the habitable area for RIFA may increase by approximately 5% over the next 40 to 50 years, a northward expansion of 33 to 35 kilometers. As the pace of global warming is expected to quicken in the latter half of the century, however, the habitable area for RIFA in 2100 is predicted to be greater than 21% of its current range, a northward expansion of 133 +/- 68 kilometers (Morrison et al. 2005).

The study also states that further expansion in the Western North Carolina region is not predicted by the model, probably due to the higher elevations and thus colder temperatures of the Appalachian Mountains. As historically there have been several predictive studies that grossly missed the mark (Vinson 1997), it is not surprising that this study is incorrect about the dispersal of RIFA in Western North Carolina's southern Appalachians. Predictive models are exactly that: models that hope to come as close to future dispersal as possible, though are inherently incomplete representations of the complexity of nature and the many variables that factor into the success of a species' growth. One of these factors is whether there are polygyne colonies in

addition to monogyne colonies, as the local colony density of polygyne colonies can be especially high and lead to faster dispersal through budding. Future studies may test allozymes or other genetic markers in order to determine what type of colonies are present within the study area (Hölldobler and Wilson 1990).

Climate change can play an important role in the dispersal of invasive species such as RIFA because of its potential pervasiveness and effect on fundamental biological processes. Climate change can interact with other existing stressors to affect the distribution, spread, abundance, and impact of invasive species (Hellman et al. 2008). This is especially distressing as the world's biota is experiencing an extinction crisis driven primarily by habitat loss and anthropogenic disturbance (Pimm and Raven 2000). Habitat disturbance often facilitates introduction and establishment of invasive species, which also cause loss of biological diversity (LeBrun et al. 2012).

Further GIS analysis as part of this study demonstrate that predictive factors and models can be used in conjunction with field surveys to better understand where RIFA dispersal occurs and may occur in the near future (Figs. 13 and 14). This in turn can help biologists, policy-makers, and government agencies, as well as the public, to implement actions and precautions to address potential future dispersal. As long as these groups keep in mind the complexity and inherent limitations of predictive models, analysis such as these maps can be useful tools in management of RIFA. Limitations include future adaptations of RIFA populations and unknown abiotic and biotic factor changes, such as climate change gradients. Examination of any recent adaptation(s) of RIFA since their introduction into the United States may serve as a useful baseline assessment to incorporate into future predictive models (Costa, pers. comm.).
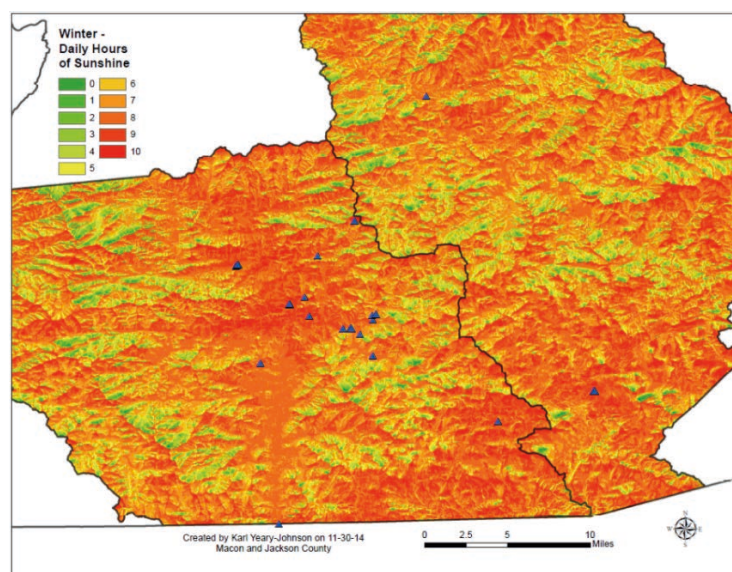
FIG. 13. RIFA mounds found during survey and daily hours of sunshine in winter, Macon and Jackson counties, North Carolina. Created in ArcGIS 10.2 (ESRI 2014).

RIFA mounds in the study area were mostly found in areas with eight or more hours of sunshine per day during the winter season (Fig. 13). This is an example of merely one of the many variables that can go into a predictive dispersal model, assigning various numerical weights to the variables within a formula to compile into a final output. I created a predictive model based on current RIFA colony establishment using the ArcMap add-in analysis tool *Feature Analyst 5.0* by Overwatch Systems, Inc. and NASA *Landsat ETM+* for orthoimagery layers (NASA

Landsat Program 2003) (Fig. 14). This is a good starting point for biologists and management groups.
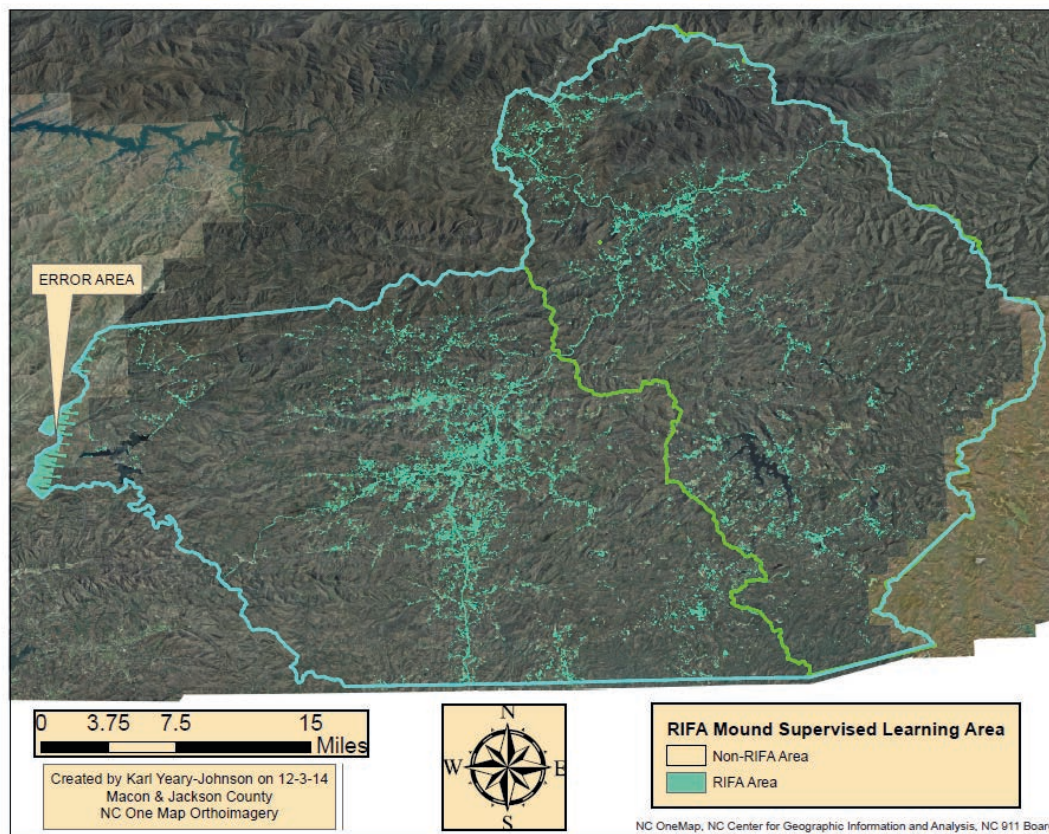
FIG. 14. Predicted RIFA distribution areas in Macon and Jackson counties, North Carolina, based on existing colony distribution using Feature Analyst tool in ArcMap. Created in ArcGIS 10.2 (ESRI 2014).

Ecological Impact of an Invasive Species

There are examples of invasive species altering the evolutionary pathway of native species by competitive exclusion, niche displacement, hybridization, introgression, predation, and ultimately extinction. Invaders themselves evolve in response to their interactions with natives, as well as in response to the new abiotic environment. Flexibility in behavior, and mutualistic interactions, can aid in the success of invaders in their new environment (Mooney and Cleland 2001). Impacts may occur through simple trophic interactions such as herbivory, predation or parasitism. Alien species may also affect native species and communities through more complex mechanisms such as competition for resources, disease transmission, apparent competition, or pollination disruption, among others. Finally, some invasive insects, particularly forest herbivores and ants, are known to affect ecosystem processes through cascading effects (Kenis et al. 2009).

In the presence of RIFA, community structure of an ecosystem converges to a random pattern. Not only does the presence of RIFA reduce species density at local scale, it alters the co-occurrence patterns of surviving species at a biogeographic scale (Gotelli and Arnett 2000). The implications of these negative impacts on local ecosystems is troublesome, especially in the southern Appalachians, where native ant species' diversity (as well as biodiversity in general) is high relative to the rest of the United States (Gotelli and Arnett 2000). According to comparative

studies, native ant species biodiversity displays a significant mid-latitude peak in Southern Virginia (Gotelli and Arnett 2000). Almost all of the areas where RIFA has established significant numbers of colonies in the United States, native ant species living in the same habitat area has been wiped out (Tschinkel 2006).

By reducing native ant populations in the southern Appalachians, RIFA could negatively impact the native ant-seed mutualism in that fewer seeds are dispersed. Moreover, RIFA may function as seed predators of at least some ant-dispersed plants (Zettler et al. 2001). These disruptions might ultimately affect forest structure and community composition. Although the mountainous terrain and forest matrix of the southern Appalachians is not ideal for establishment of RIFA colonies, humans and their wide-spread fragmentation of the mountain landscape with roads, homes, and other uses is leading to the increased presence of RIFA. In turn, because RIFA is known to forage up to 100 yards (Tschinkel 2006) into forests, if colonies are located at the edge of forests, edge effects affecting wildflowers, insects, and trees are a likely consequence of increased RIFA presence. It is important to limit anthropogenic disturbance of the natural habitat in order to limit RIFA dispersal in the study area, as well as farther north.

Studies have found that both RIFA and native ant nests have greater fungal abundance but lower species richness and diversity than non-nest soil. Given the high densities of RIFA and their frequent mound movements, changes in soil fungal communities might have lasting impacts on soil conditions and seedling survival (Zettler et al. 2002). The abundant biodiversity of plant species in the southern Appalachians could become threatened by RIFA through this behavioral mechanism.

Additionally, forest clear-cutting could jeopardize mutualistic relationships between native ant species and ant-dispersed plants (called mymecochores). Some mymecochores in southeastern forests include violets (*Viola spp.*), wild ginger (*Hexastylis spp.*), trilliums (*Trillium spp.*), and bloodroot (*Sanguinaria canadensis*). Within these forests, many native species of ants are attracted to the elaisome on seeds, a lipid-rich attachment (Zettler et al. 2004). This same attraction is found in RIFA. A study found that RIFA remove seeds, consume elaiosomes, and scarify or destroy seeds. (Zettler et al. 2001). This could develop into a serious threat to ephemerals and other species should RIFA distribution continue to increase.

Fig. 15. Dr. James Costa (pictured right) and author upon completion of project.

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Comparison of Stream Temperature Dynamics Between North-facing And South-facing Watersheds in The Southern Appalachians

Zhexi Zeng

Abstract. Stream temperature is an essential factor in the study of water quality and aquatic biological habitats. A study was designed to investigate the influence of watershed aspect on water temperature in the Coweeta Basin in the southern Appalachians. South-facing watersheds are warmer than north-facing watersheds at the same elevation on average but are cooler in summer. To explain this pattern, we designed a longitudinal stream temperature study to observe temperature changes along streams from the origin to the outlet on a pair of first order watersheds of different facing aspect. The data collected from the watersheds in Coweeta Basin showed that the bottoms of south-facing watersheds appear to be warmer than the bottoms of north-facing watersheds at similar elevation during most times of the year except summer. North-facing watersheds experience a larger increase in temperature during the late summer time, and we speculated that the denser canopy cover on south-facing side of the basin in the summer and the removal of canopy cover during the autumn caused the difference in temperature changing dynamics on watersheds of both aspects.

Keywords: Coweeta Basin; seasonal changes; southern Appalachians; stream temperature; watershed aspects.

Introduction

Stream temperature is a significant characteristic to be considered in assessing stream properties. Not only does temperature help determine stream water quality, it also helps to control the properties of aquatic biological habitats (Cassie et al. 2006). For instance, the brook trout, a species of fish known to be a cold-water dweller, is seldom found in streams warmer than about 22 $^\circ$C (Siegler 1948). As air temperatures have increased over the last several decades due to carbon dioxide emissions from human industrial development, the impact of climate change becomes an overwhelming concern (Allen et al 2009). As predicted by General Circulation Models, the global temperature will continue to increase over the 21st century. Increasing air temperature may significantly impact stream temperatures. For example, a recent study by Caldwell et al. (2014) predicted that stream temperatures in the southern Appalachians, the natural biodiversity hotspot and home to several cold water fish species, will be especially vulnerable. If global warming occurs as predicted in this century, stream temperature studies will help in finding possible refuges for the cold water dwelling species as temperature increases.

There are many factors that influence stream temperature and the sensitivity of stream temperature to changing air temperatures. These factors fall into four main categories: atmospheric conditions, topography, stream discharge and streambed (Cassie et al. 2006). Atmospheric temperature and radiation control the general direction of stream temperature changes due to heat exchanges, and are affected by the change of vegetation and canopy cover conditions throughout the year. Topographic factors such as elevation, stream slope, and watershed aspect need to be considered because they can affect atmospheric conditions around the streams. Stream discharge influences the heat capacity of stream reaches, and streambed structure controls the mixing of underground water from different sources (Cassie et al. 2006). Watershed aspect can also affect stream temperature (Ebel 2012). North-facing slopes in the northern hemisphere receive less solar radiation than south-facing slopes. The difference in solar radiation on both aspects might cause a variance of stream temperature between streams located on north-facing watersheds and south-facing watersheds. Thus, it is possible that streams in

north-facing watersheds have lower stream temperatures than those in south-facing watersheds, potentially providing refuge for cold water fish species under climate change (McMahon 2007).

For this study, my mentor and I designed an experiment to test the hypothesis that south-facing streams are warmer than north-facing streams due to greater exposure to solar radiation in small, forested, first-order streams in the southern Appalachian mountains. We analyzed temperature data previously collected on weirs at the bottoms of each watershed and developed a new longitudinal stream temperature test to study the changing dynamics along the streams. Specifically we sought to 1) determine whether there are differences in stream temperature in north-facing vs. south-facing watersheds at low and high elevation, and 2) examine longitudinal changes in stream temperature in north- vs. south-facing watersheds.

METHODS

Study Site
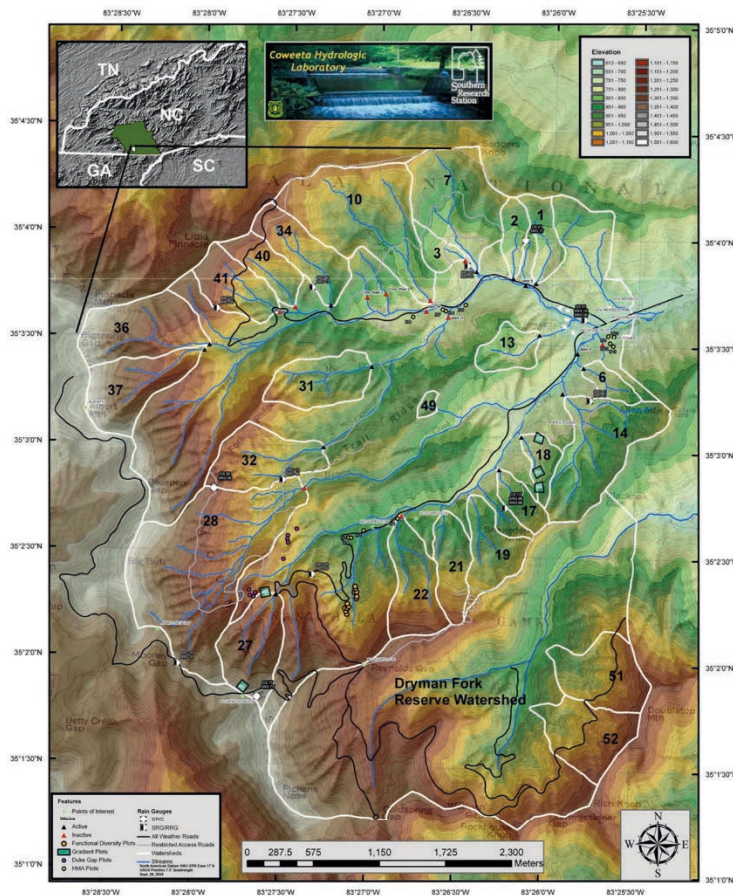
FIG. 1. Map of Coweeta Basin and numbered watersheds

The experiment was conducted at the Coweeta Hydrological Lab, located in Otto, NC (Fig. 1). The Coweeta Basin is a 2,185 hectare basin that has been used as experiment site for forest climate and watershed monitoring studies for almost 80 years. A comprehensive monitoring network for scientific research has been in place since establishment of the site (Laseter et al. 2012). The Coweeta basin is located in the temperate rainforest climate zone in the Appalachians

and is considered among the places with highest annual precipitation in eastern US, averaging 1794 mm (Laseter et al. 2012). Within the basin, the elevation ranges from 675m to 1592m. The Coweeta basin faces east, and is divided into smaller sub-watersheds generally oriented on the north and south sides of the basin.

Table 1. Conditions of watersheds in the Coweeta Basin. The data for precipitation and runoff are based on the measurements from November 1993 to October 2012.

	WS02	WS07	WS14	WS18	WS27	WS36
Elevation category for this study	Low	Low	Low	Low	High	High
Aspect category for this study	South	South	North	North	North	South
Area (ha)	12.3	59.0	61.0	12.5	39.1	48.6
Min elev (m)	709	722	707	726	1061	1021
Max Elev (m)	1004	1077	992	993	1455	1542
Slope (%)	60	57	49	52	55	65
Aspect	SSE	S	NW	NW	NNE	ESE
Mean Annual Precipitation (mm)	1791	1834	1817	1995	2271	1999
Mean Annual Runoff (mm)	788	924	939	964	1657	1665
Runoff Ratio	0.44	0.50	0.52	0.48	0.73	0.83

Study Design

In this study, we examined stream temperatures at the outlets of five north- and south-facing watershed pairs at similar elevation. We also examined stream temperatures along the length of the stream reach for a single north- and south-facing watershed pair at a similar elevation. We chose several pairs of watersheds that met these criteria (Table 1). Watersheds 02 and 07 are representative of south-facing watersheds at low elevation and watersheds 04 and 18 are north-facing watersheds at low elevation. Similarly, Watershed 27 (generally north- facing) and Watershed 36 (generally south- facing) were paired to examine differences in stream temperature at high elevation. All of the watersheds are reference watersheds that have not been harvested since the 1920s except Watershed 07, which was clear-cut in 1976, but has naturally regenerated since then.

Stream Temperature Measurements

Stream temperatures were measured using Onset HOBO Water Temperature Pro v2 Data Logger sensors and a HOBO Waterproof Shuttle was used to transfer from the site. The published uncertainty of the temperature sensor in water is ±0.2 ℃ (Onset Company), and the sensors were set up to record the temperature every hour.

Water temperature was measured at the outlet of each sub-watershed from December 2009 to June 2014. In addition to these more recent data, water temperature at the outlet of low elevation watersheds WS07 (south facing) and WS14 (north facing) was recorded from March 1986 to January 1997.

We selected watershed 2 and Watershed 18 as the two sites to implement our longitudinal stream temperature studies. The streams in both watersheds are approximately 350 meters in length, and we put 8 sensors in each stream every 50 meters. Sensors were attached to trees with wires and placed under rocks to stabilize them. In addition, we installed two sensors in specially made caps to measure the air temperature of the watersheds at site WS02_14 and WS18_14 midway along the stream reach. To study the influence of possible inflow of water from

intermittent tributaries, we installed sensors WS02_9 and WS18_6 in branches from the main streams. The data collection started at 4pm on September 10th in Watershed 18 and at 4pm on September 12th in Watershed 2. Data were collected until November 6th, 2014 in both watersheds. Over 20,000 data points were recorded from the 20 temperature sensors during this period.
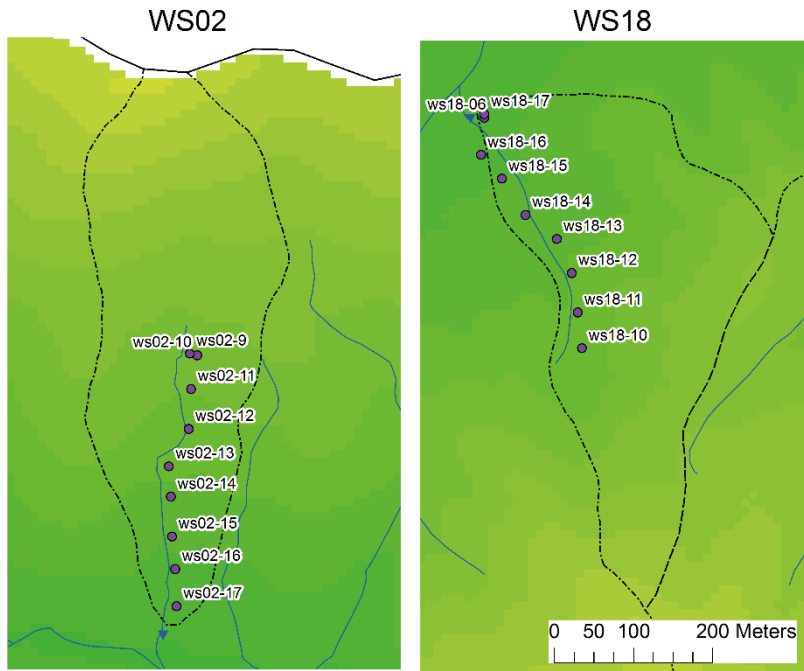

FIG. 2 Map showing sensor locations on Watershed 02 and Watershed 18. Each sensor is placed 50 meters away from each other along the streams.

RESULTS

North- and South-facing Watersheds Along an Elevation Gradient

Based on the stream temperature data collected at the outlet of each watershed, we developed pairs of comparison to see the difference of change in stream temperature on watersheds with different facing aspects. Both north and south facing watersheds exhibit a seasonal pattern of stream temperature with minimum temperatures occurring during the winter months and maximum temperatures occurring during the summer months. Average stream temperatures from 2009-2014 across all low elevation watersheds (WS02, 07, 14, and 18) were 11.6 ˚C while average temperatures across high elevation watersheds (WS27 and 36) were 10.4 ˚C (1.2 ˚C less than at low elevation), illustrating the influence of elevation on stream temperatures. Average stream temperatures were lower in north-facing watersheds than south-facing watersheds for a given elevation. The average stream temperature from 2009-2014 for low elevation south-facing watersheds 02 and 07 was 11.9 ˚C while the average stream temperature for low elevation north facing watersheds 14 and 18 was 11.4 ˚C (0.5 ˚C less than south-facing watersheds). Similarly, the average stream temperature for high elevation south-facing watershed 36 was 10.8 ˚C and that of high elevation north-facing watershed 27 was 10.1 ˚C (0.7 ˚C less than south facing). However, south-facing watersheds tend to be warmer than north-facing

watersheds in most time in the year but not summer when the air temperature is highest. For example, stream temperature in north facing WS14 was 0.1°C lower than in south facing WS07, but was 1.1 ˚C higher in August. Stream temperatures in winter months (e.g. February) were consistently lower in the north-facing watersheds than in the south-facing watersheds. August stream temperatures in north-facing low elevation watersheds 14 and 18 were 0.8 ˚C higher than south-facing watersheds 02 and 07 while February stream temperatures were 1.7 ˚C lower (FIG. 4). Similarly, average August stream temperature in high elevation north-facing watershed 27 was 0.7 ˚C higher that south-facing watershed 36 while the average February stream temperature was 1.8 ˚C lower.
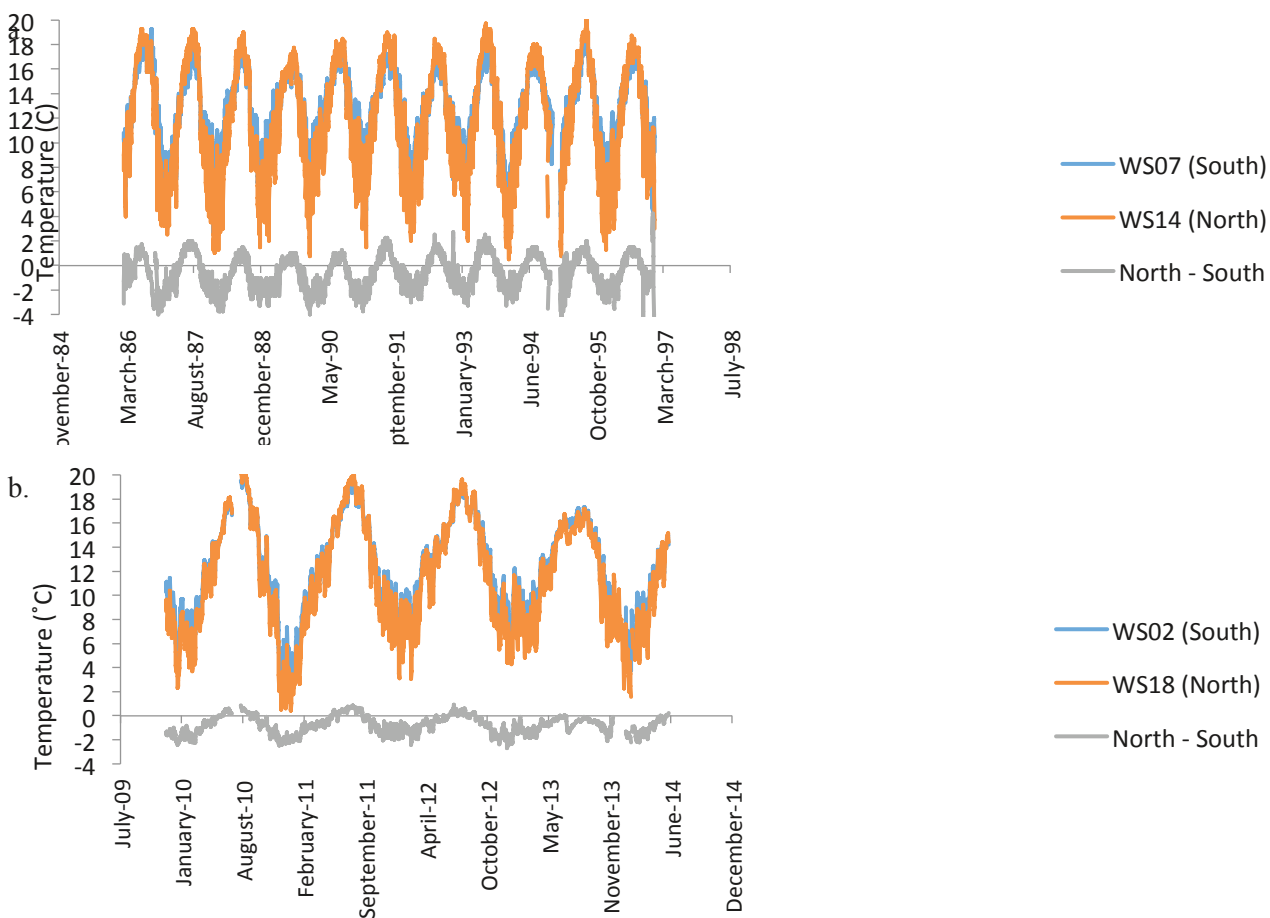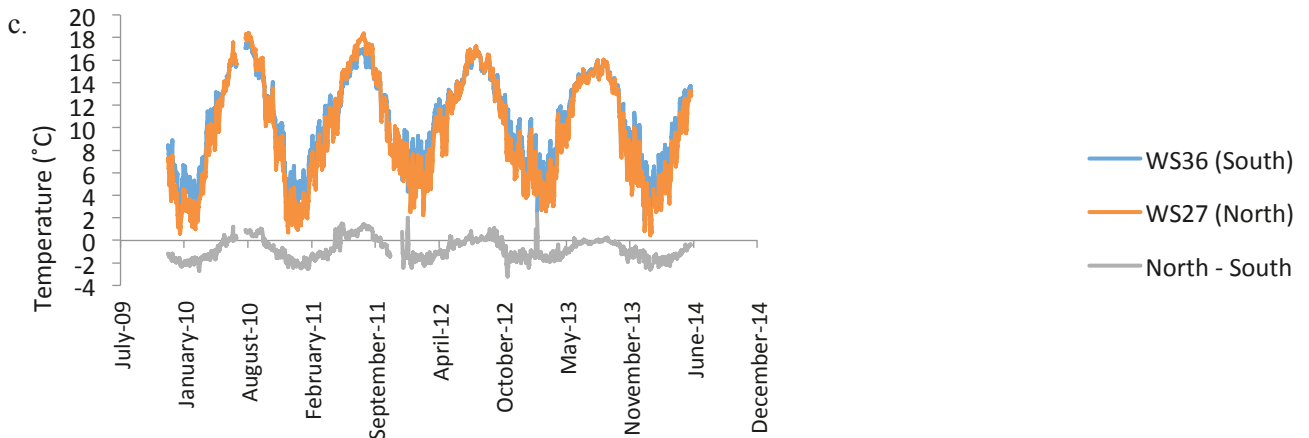
FIG. 3. The stream temperature comparisons on three pairs of watersheds. Chart a. shows the temperature data and differences collected on the bottoms of watershed 07 (south-facing) and watershed 14 (north-facing) from 1986 to 1997. Chart b. shows the temperature data and differences collected on the bottoms of watershed 02 (south-facing) and watershed 18 (north-facing) from 2009 to 2014. Chart c. shows the temperature data and differences collected on the bottoms of watershed 36 (south-facing) and watershed 27 (north-facing) from 2009 to 2014.
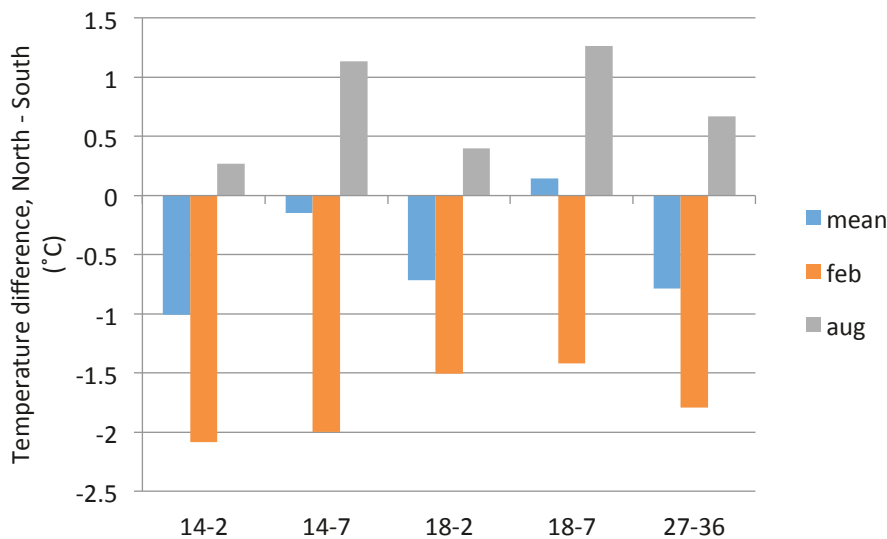
FIG. 4. Mean temperatures of north-facing watersheds minus south-facing watersheds in February, August and all months of each pair of watersheds.

Longitudinal Stream Temperature in Low Elevation North and South-facing Watersheds

The 20 temperature sensors installed in watershed 02 (south-facing) and Watershed 18 (north-facing) recorded data on longitudinal water temperature from the origins of both streams (Fig. 5). There was an apparent reversing in upstream to downstream temperature on both watersheds between the late summer time to the early fall. (Fig. 5). During summer days when air temperature is higher and stays over 20 °C, stream temperature increases in the downstream direction. In contrast, in late October and early November, when the air temperature dropped significantly in the area, stream temperature decreased in the downstream direction. There is

some variability in the general pattern, however, due to water coming from other streams or underground water infiltration.

Comparison at Fixed Distances Downstream of the Origin

The stream temperature at the top of Watershed 02, the south-facing watershed, is always warmer than at the top of Watershed 18, with an average difference of 1.64 ℃. At low temperature, the difference between the two sites is smaller than at high temperature. On sites at 200m downstream from the headwater, the stream at south-facing watershed still has a warmer temperature for most of the time, except the time when temperature becomes low. The difference of the temperature turns to be 0.66℃, which is smaller than the mean difference at the top of the streams. At the sites on the bottoms, the temperatures of the north-facing watershed appears to be warmer than that of south-facing watershed in early September. However, in late October and November the stream temperature in south-facing Watershed 02 is higher than in north-facing Watershed 18. The mean difference is 0.54 ℃ throughout the whole sampling period. For most time periods we sampled, the air temperature was higher (0.93 ℃, on average) in the south-facing Watershed 02 than in the north-facing watershed (Fig. 7.). During the hottest time in the day around 15:00, especially in the early winter, the difference in air temperature between the two watersheds is highest. The abrupt change could be due to exposure of the sensor at Watershed 02 to the sun at 15:00 every day.
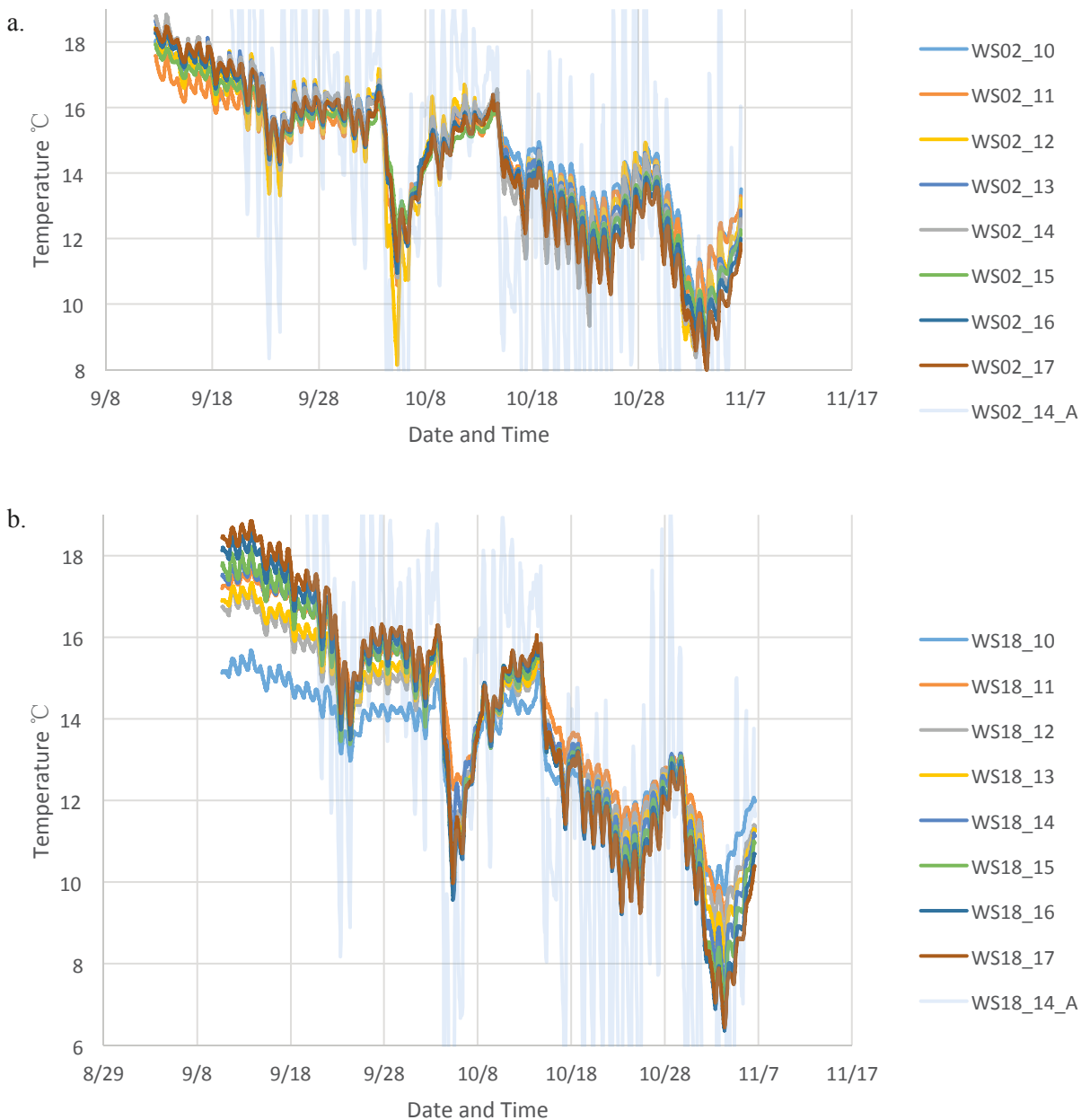
FIG. 5. Overall data collected by the 8 sensors on watershed 02 (Chart a.) and watershed 18 (Chart b.) from Sept. 10, 2014 to Nov. 6, 2014. Sensors were numbered from the headwater to the weir from 10 to 17, with a distance of 50 meters between each sensor. The air temperatures collected (WS02_14_A and WS18_14_A) is displayed as a reference in the background.
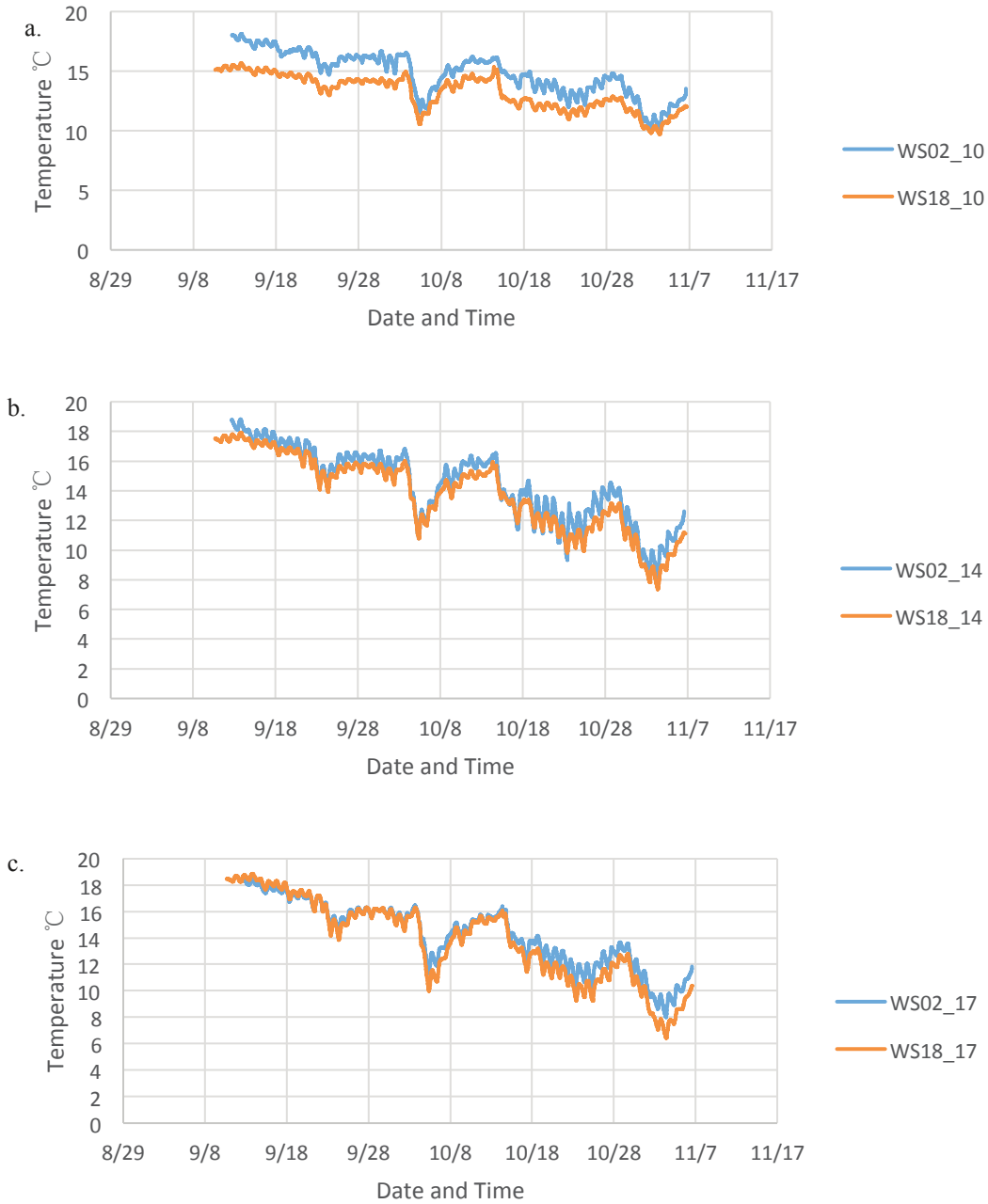
FIG. 6. Temperature curves at three fixed points on both streams collected from Sep.10 to Nov.6, 2014. Chart a. shows the recorded temperatures at the headwater. Chart b. shows the recorded temperatures 200m downstream from the headwater. Chart c. shows the recorded temperatures at the bottoms of the watersheds 350m downstream from the headwater.
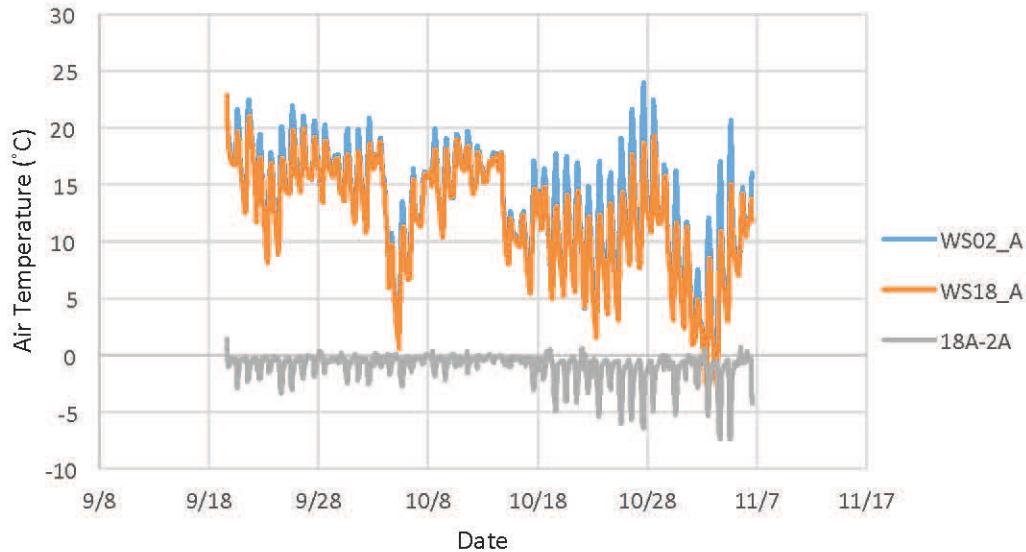
FIG. 7. Air temperature (°C) curves at both watersheds. The gray line represents the difference in air temperature (Watershed 18 minus Watershed 02).

Mean temperature longitudinal changing trend

To analyze how the temperature changes along streams over the transition from summer to fall, we calculated the mean temperature collected from each sensor for every two weeks (FIG. 8). There was a higher slope of temperature versus distance downstream in watershed 18 of 0.0055 °C/meter downstream than in watershed 02, which has a slope of 0.0011 °C/meter, such that the temperature at the outlet of watershed 18 was higher than watershed 02.

In the trend observed in the 3rd and 4th week, both of the slopes of stream temperature versus distance downstream decreased as the air temperature decreased. The slope in WS02 was approximately zero indicating that the temperature remained relatively constant over the length of the stream, while the slope in WS18 was 0.0017 °C/meter, lower than that of the first two weeks but remained positive indicating an increase in temperature in the downstream direction. Further, the stream temperature in north-facing watershed 18 remained cooler than watershed 02 at the outlet of the streams.

As the air temperature continued to decrease, the slopes of the mean stream temperature vs. distance from the 5th to the 6th week further decreased and became negative, -0.0019 °C/meter for the south-facing WS02 and -0.0008 °C/meter for the north-facing WS18, indicating a decrease in stream temperature in the downstream direction. The south-facing watershed WS02 had warmer temperatures everywhere in the stream, but had a larger slope of temperature dropping rate. In the 7th to the 8th weeks from Oct. 24th to Nov. 6th the two streams tended to both experience a larger temperature drop, and the slopes were calculated to be-0.004 °C/meter for north-facing WS18 and -0.005 °C/meter for south-facing WS02.

a.

$y = 0.0011x + 16.508$
$R^2 = 0.1968$

$y = 0.0055x + 15.308$
$R^2 = 0.69422$

WS02_1
WS18_1
Linear (WS02_1)
Linear (WS18_1)

Temperature ℃
Distance from Origin (m)

b.

$y = -5E-05x + 14.908$
$R^2 = 0.00132$

$y = 0.0017x + 13.933$
$R^2 = 0.34525$

WS02_2
WS18_2
Linear (WS02_2)
Linear (WS18_2)

Temperature ℃
Distance from Origins (m)

c.

$y = -0.0019x + 14.283$
$R^2 = 0.7619$

$y = -0.0008x + 13.233$
$R^2 = 0.16931$

WS02_3
WS18_3
Linear (WS02_3)
Linear (WS18_3)

Temperature ℃
Distance from Origins (m)

FIG. 8. Longitudinal change in mean temperature collected by each sensor on both watersheds in different time period. Chart a. displays mean temperatures of the 1st and 2nd weeks from Sept. 10 to Sept. 26. Chart b. displays mean temperatures of the 3rd and 4th weeks from Sept. 26 to Oct. 10. Chart c. displays mean temperatures of the 5th and 6th weeks from Oct. 10 to Oct. 24. Chart d. displays mean temperatures of the 7th and 8th weeks from Oct. 24 to Nov. 6.

DISCUSSION

We hypothesized that stream temperatures are warmer in south-facing watersheds than in north-facing watersheds. The stream in the south-facing Watershed 02 had higher temperatures than the stream in the north-facing Watershed 18 during the time period we sampled. However, the stream temperatures at the outlets of south-facing watersheds are warmer than those of north-facing watersheds at similar elevation most times of the year, north-facing watersheds tended to have higher outlet temperatures than south-facing watersheds in summer when stream temperatures were highest. The north-facing Watershed 18 experienced a larger increase in temperature along the length of the stream during the first two weeks we sampled in September so that temperature surpassed Watershed 02 at the outlet. However, as the time moved from late summer towards early fall the slope of temperature change over the length of the stream decreased over time faster in the north-facing watershed than in the south-facing watershed. This resulted in higher temperatures at the bottom on watershed 02 than the bottom on watershed 18. These findings suggest that stream temperature in north-facing WS18 is more sensitive to external environmental factors (e.g. solar radiation) than south-facing WS02.

We found that the riparian zone in Watershed 02 tended to have dense rhododendron forest, while canopy in Watershed 18 had several open areas along the stream, due in part to the loss of eastern hemlock (Ford et al. 2011). The open canopy in north-facing WS18 could explain the higher sensitivity to external environmental factors. Sun et al. (2004) conducted a comprehensive review of the impact of riparian forest management on stream temperatures and found studies demonstrating that forest harvesting along streams usually results in increased stream temperatures (Swift 1973, Wooldridge and Stern 1979). Watersheds that were completely harvested exhibited increases in summer stream temperatures of 3-4 °C compared to water temperatures in streams with intact riparian vegetation (Swift and Messer 1971, Swift 1982). Stream temperature regimes in the treated watersheds were found to return to pretreatment levels as the vegetation regenerated. While the south-facing watersheds have more exposure to solar radiation throughout the year, the difference in canopy cover over the stream could outweigh differences in solar radiation. In summer when all deciduous trees have leafed out, the south-

facing watershed may be less influenced by the sunlight. The south-facing watershed thus experienced a lower rate of increase in stream temperature along the river than the north-facing watersheds, resulting in a higher stream temperature at the north-facing watershed outlet. During the fall season, when leaves start to drop, the canopy cover is reduced so the slope of temperature change along south-facing watershed became larger than that of the north-facing watershed. During the last week of the time period we sampled in late October and early November, most leaves had fallen and both watersheds experienced a temperature drop, with the south-facing watershed dropping slower due to higher reception of solar radiation when no canopy is present. As the origin of south-facing watershed is always warmer than the north-facing one probably because of long-term exposure to the sun for years, the changes along the streams explains the fluctuation of temperature difference at the bottoms of each pair of watersheds. The presence of canopy cover may decide the rate of change in both watersheds. This study suggests that greater solar radiation on the south-facing watershed does not necessarily always increase temperature everywhere since there are other factors that affect stream temperature like underground water infiltration and canopy cover.

Further work still needs to be done to fully understand the relationship between stream temperature and watershed aspect. While the data we collected helped identify key differences between north- and south-facing watersheds, a minimum of one full year of data should be collected to include the warmest times of the year when north-facing watersheds have higher stream temperatures than south-facing watersheds. To explore the role of differences in canopy cover over the stream, point measurements of leaf area index and perhaps sunlight or solar radiation could be taken. Moreover, to confirm whether the results of the longitudinal study in watershed 18 and watershed 02 are representative of all watersheds in the Coweeta Basin, another longitudinal temperature measurement should be done in a different pair of watersheds at high elevation. After confirming the changing dynamics of stream temperature in watersheds in the Coweeta Basin, further research can be done to compare the result from Coweeta Basin with other stream systems outside of the basin to determine the general dynamics of stream temperature, in order to find possible refuges for cold-water dwelling species.

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A COMPARATIVE ASSESSMENT OF THE STREAM HEALTH OF CANEY FORK, A TRIBUTARY TO THE TUCKASEGEE RIVER

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Abstract. Caney Fork is located within the Little Tennessee watershed in the southern Appalachian Mountains in a region of Jackson County, North Carolina that has been subjected to recent human development. However, the headwaters of the stream are less accessible to humans and have larger riparian zones, more canopy cover, and less exposure to the adverse effects of human development than the downstream areas of the stream. In each of our assessments we observed physical parameters that affect the ecology of the stream including reduced riparian zones, increased erosion, sedimentation and channelization, and reduced habitat diversity for aquatic life. To conduct these assessments we followed the protocols "NC Division of Water Quality: Benthic Macroinvertebrate Procedures", a habitat assessment from the NCDENR Department of Water Quality, the Wolman Pebble Count procedure, and the standard procedures of Rosgen's Bank Erosion Hazard Index (BEHI). We also compared our data to the findings of Allan et al. (2013) and found that the only change from last year was an increase in BEHI, while all other assessments did not indicate an increasing impact from human activity from the headwaters to the mouth.

Key words: BEHI; EPT; FBI; macroinvertebrates; NCBI; pebble count; southern Appalachians; stream health; watershed; western North Carolina.

INTRODUCTION

In the Southern Appalachians, the biotic health and stability of mountain streams reflect the human activity and land use in the surrounding area. Past studies have found that increasing amounts of human activity and land use near streams have negatively altered the biotic health and stability of those streams. Increased human activity can cause disruptions to the ecology of the stream, such as reduced riparian zones, increased erosion and sedimentation, channelization and reduced habitat diversity for aquatic life (Moore and Palmer 2005, Violin et al. 2011).

Caney Fork is a tributary of the Tuckasegee River, located in Jackson County, North Carolina. It is part of the Tuckaseegee sub-watershed within the Little Tennessee watershed. At Caney Fork, the headwaters of the stream are less accessible to humans and have wider riparian zones, more canopy cover, and less exposure to the adverse effects of human development. Transitioning downstream along Caney Fork, human development and activity increases. The number of gaps in the riparian zones increase, while canopy cover and distance from impervious surfaces decreases (Allan et al. 2013). Observed land use immediately surrounding Caney Fork includes pasture, forest, residential housing, and commercial development.

From upstream to downstream, the amount of riparian zone decreases because of increased human activity along Caney Fork. Riparian zones are important to the stream ecosystem because deep root systems reduce the potential for erosion and root mats provide habitat for aquatic life. Intact riparian zones stabilize the stream, filter runoff from impervious surfaces and help contain sediments and sediment-bound pollutants (Naiman and Decamps 1997). Canopy cover provides important shade that cools the stream and increases the amount of dissolved oxygen that the water can hold, while riparian vegetation is an important source of organic matter. According to the River Continuum Concept, there are trends in stream dynamics that occur naturally between the headwaters and the mouth of the stream (Vannote et al. 1980).

These include an increase in stream capacity and a decrease in stream competence, velocity, stream gradient, intermediate axis length of rocky substrate, and canopy cover (Vannote et al. 1980, Brown 2007). It is important to note that these natural trends can be altered by human activity.

The purpose of our study was to assess the stability and biotic health of Caney Fork as well as the spatial and temporal effects of human activity on the stream. In order to evaluate spatial effects, we assessed the biotic health and stability of Caney Fork from September to October 2014 at eight sites from the headwaters at Mull Creek to the mouth of Caney Fork. We also compared our data to the findings of last year's capstone study (Allan et al. 2013), which analyzed Caney Fork at similar reaches during Fall 2013. Based on the results of the 2013 study, we hypothesize that there will be increasing negative impacts to stream health and stability from the headwaters to the mouth of Caney Fork due to increasing amounts of human activity. We also hypothesize that stream health will have declined within a one-year period because of continuing development surrounding Caney Fork.

METHODS

Site Selection

In order to do a comparative study of Caney Fork, we sampled the same sites chosen by Allan et al. (2013), and we added two additional sites (Fig. 1). The previous study selected sites based on ease of accessibility from a road. We sampled two sites along Mull Creek, a third order tributary to Caney Fork, and six sites along Caney Fork, covering a total of 13.8 river miles. Caney Fork is a third order stream when it meets Mull Creek and becomes a fourth order stream at its confluence with the Tuckasegee River. The NC Department of Natural Resources (NCDENR 1992) classifies Caney Fork as a WS-III stream, which means that the water is used as a supply for drinking, culinary, or food processing purposes where a more protective WS-I or II classification is not possible. This classification also indicates that the watershed is typically low or moderately developed (NCDENR 1992). The previous study examined only six sites that extended from the headwaters of Caney Fork to its confluence with the Tuckasegee River. We numbered our sites S1-S8, with S1 located in Mull Creek, extending in numerical order to S8 where Caney Fork flows into the Tuckasegee. We chose a 100 m reach at each site to obtain a representative sample of the riffles, runs, and pools present.
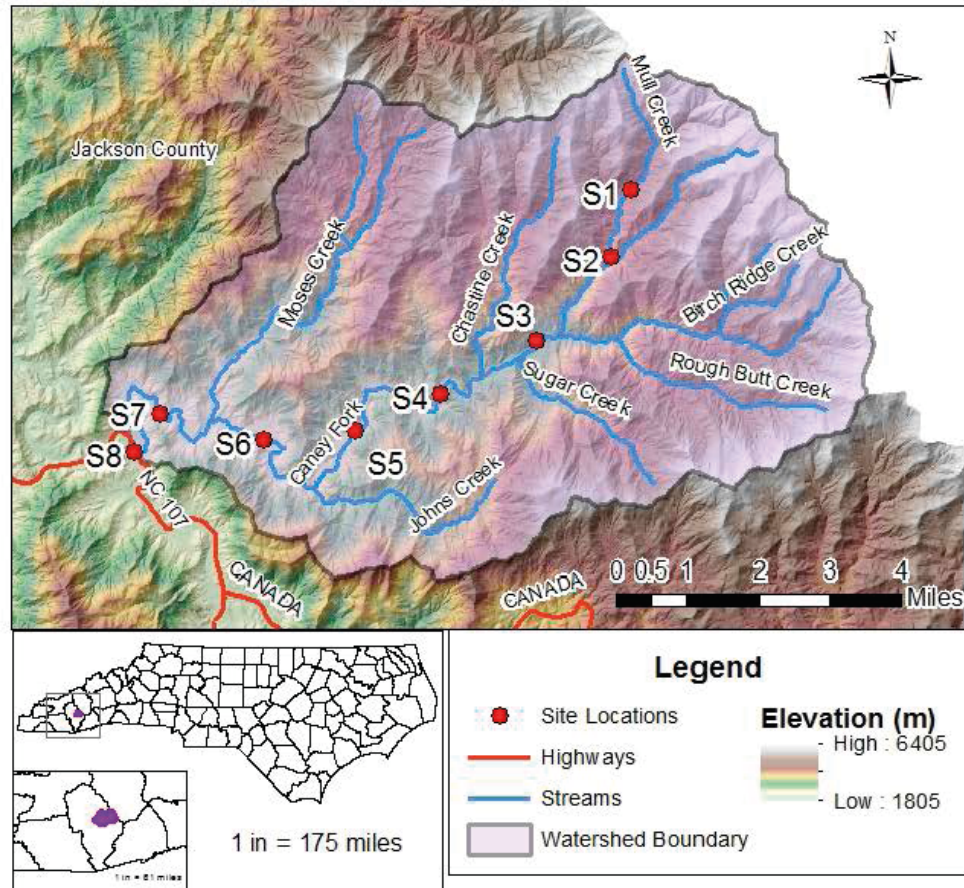
FIG. 1. Study site locations, streams, tributaries, and elevation within the Caney Fork watershed. The map in the left corner displays the location of the watershed within North Carolina and Jackson County (ESRI 2012).

Habitat Assessment

At each site we used the same standardized Habitat Assessment Field Data Sheets developed by the NCDENR Department of Water Quality (NCDENR 2012) that were used by Allan et al. (2013). This protocol was developed by the state of North Carolina and uses a visual assessment of the stream habitat quality for each reach. In the visual assessment of each reach, the basic stream characteristics (bank angle, bank height, stream width and depth, the presence of man-made structures, modifications, flow conditions, turbidity, weather conditions) and stream surroundings (land usage) were described. Habitat assessment scores were based on eight variables: instream habitat, bottom substrate, pool variety, light availability, riffle habitat, riparian zone width, bank stability and vegetation, and channel modifications. The highest possible overall score was 100.

Bank Erosion Hazard Index

To assess the stability of the stream's banks, the Bank Erosion Hazard Index (BEHI) takes into account several indices including bank height ratio, root depth ratio, root density, bank angle, and surface protection, to create a Total Field Index (TFI). Then, the final BEHI score is adjusted based on the physical composition of the bank (Rathburn 2008). We calculated the bank

height ratio by measuring the bank height and comparing this to the measurement of bankfull height which is the height of water during a high discharge event. We determined the root depth ratio by visually observing and taking measurements of the average depth of roots in the stream bank. We then calculated the ratio of depth of roots in the bank to total depth of the bank. We determined root density by analyzing the proportion of woody vegetation roots embedded in the stream bank. Bank angle is the visual estimation of the angle of the bank from the surface of the water to the top of the bank with respect to the perpendicular cross-section of the stream. Surface protection was visually estimated as a percentage of the bank covered with woody vegetation roots, logs, embedded woody debris, and large rocks.

Each of the calculated ratios, proportions, and percentages corresponded to a BEHI index value. For example, a "very low" hazard index is given to bank with a low bank height ratio, a high root depth ratio, a high root density, a bank angle near 0°, and is indicative of a bank that is very stable. Higher hazard index scores indicate that the stream bank is less stable, and the most unstable hazard category is "extreme" (BEHI value greater than 45). The scores were adjusted based on the composition of sediment in the banks (bedrock, boulders, cobble, gravel or sand). Banks composed of bedrock received a "very low" score, and those composed of boulders received a "low" index. The scores of banks composed of gravel can be increased five to ten points depending on the amount of sand present. Because different layers of stratification do not erode uniformly, BEHI scores were adjusted upwards by five to ten points, depending on the position of sedimentation layers with respect to bankfull height.

Pebble Count

A pebble count procedure was performed to determine the median particle size to help determine the stream's ability to move particles and to assess its ability to hold sediment and particles (Wolman 1954). The reach was first examined to determine the proportion of riffles, runs, and pools. We collected 100 samples, ten pebbles at each of ten sites along the reach. Each sample site was chosen to keep the percentage of cross sections in each feature (riffle, run, or pool) proportional with the stream composition. After we measured the intermediate axis of the pebble, the number was recorded on the data sheet under its specified column (riffle, run, pool, or cascade) and by its size class. Each pebble size falls under a category, such as silt, sand, gravel, cobble, boulders, and bedrock, and subcategory, such as "large" boulders, etc.

Macroinvertebrates

At each site we used the NC Division of Water Quality's Benthic Macroinvertebrate Procedures to assess the macroinvertebrate life in the stream, focusing on Plecoptera, Trichoptera, and Ephemeroptera (NCDENR:DWQ 2012). At each site, we used several techniques for capturing aquatic insects including kick-nets, D-nets, leaf packs, and visual collections. A total of four 60-second kick-net samples, four one-minute sweep net samples, four leaf pack samples, and eight 5-minute visual inspections were completed at each site. Employing multiple collection methods was important because each yields different kinds of insects. For example, we find more Trichopteran families through visual surveys because many Trichoptera build casings and anchor themselves to rocks. Plecoptera are more commonly found in leaf pack collections, and Ephemeroptera are more likely found in sweeps and kicks. Kick-nets were used to collect within the benthic environment, dislodging and washing macroinvertebrates from the

stream bed into the net. Sweep nets or D-nets were used to sweep bank-edges next to logs and large rocks, or other macroinvertebrate habitat. Leaf pack samples were conducted by collecting large accumulations of fallen leaves or debris in the stream into a bucket or pan. We performed visual collections by turning larger stones and debris in the stream to find insects. After each collection, we placed insects into a vial with alcohol to be preserved, and identified them using an identification key by Merritt et al. (2008) at a later date. Each family of orders Trichoptera, Ephemeroptera, and Plecoptera (EPT) is each assigned a tolerance value. We multiplied the tolerance value by the number of individuals within each order to assess stream health using its Family Biotic Index (FBI) value according to Merritt et al. (2008). The FBI corresponds to the tolerance to pollutants and oxygen of that family, serving as an indicator of stream health.

RESULTS

Habitat

A total score for each site was based on the habitat assessment (Table 1). In order to compare the scores calculated, a column including the maximum possible scores was added. Riffle habitat, as well as light penetration, decreased on average as the sites got further away from the headwaters. In general, the total habitat assessment scores decreased with the increase in river miles (Fig. 3). When comparing habitat assessment scores from 2013 to 2014, we noticed inconsistencies with last year's data. We noticed that the 2013 sites were mis-numbered based on their habitat assessment scores (Fig. 2). Since last year, site 5 and site 7 have changed significantly.

TABLE 1. Habitat assessment scores.

| | Assessment Site | | | | | | | | |
	S1	S2	S3	S4	S5	S6	S7	S8	Ideal
Channel Modification	4	5	4	5	3	4	4	3	5
Instream Habitat	17	20	20	20	16	20	20	15	20
Bottom Substrate	12	3	12	12	11	12	15	11	15
Pool Variety	4	10	8	6	4	6	6	6	10
Riffle Habitat	14	14	16	16	16	7	7	7	16
Bank Stability and Vegetation	14	6	13	11	8	14	13	6	14
Light Penetration	10	10	10	6	2	7	7	2	10
Riparian Vegetative Zone Width	7	10	8	8	2	7	8	1	10
Total	82	78	91	84	62	77	80	51	100
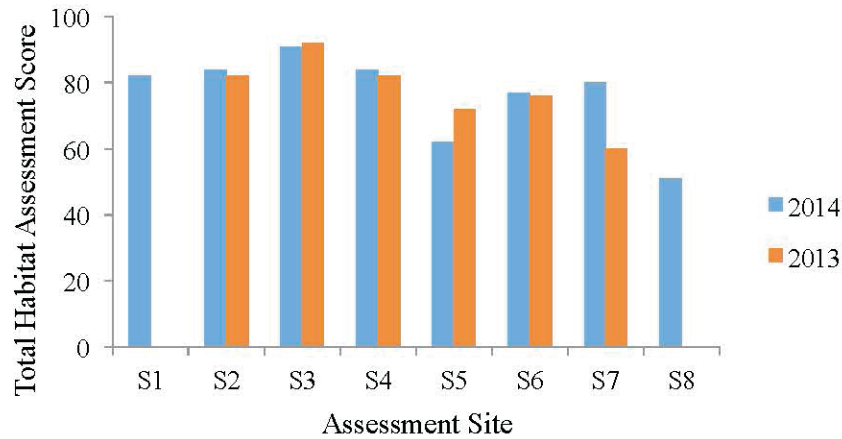
Fɪɢ. 2. Total habitat assessment scores for the sites sampled in 2013 and 2014.

$$y = -1.7561x + 87.667$$
$$R^2 = 0.44882$$
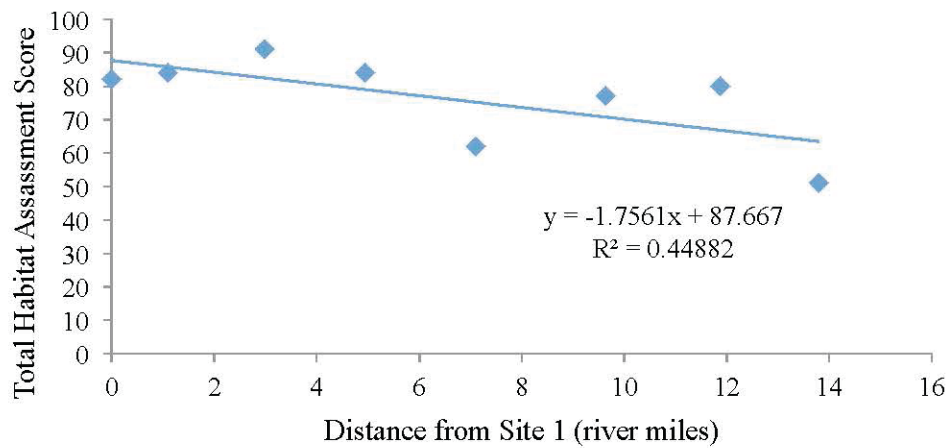
Fɪɢ. 3. Total habitat assessment scores compared to the distance in river miles from S1.

Tᴀʙʟᴇ 2. Bank aspect measurement indices, Total Field Index (TFI), adjusted Bank Erosion Hazard Index (BEHI), and erosion potential at each site.

Site	Bank Height Ratio (ft/ft)	Root Depth Ratio (%)	Root Density (%)	Bank Angle (degrees)	Surface Protection (%)	TFI	Adjusted BEHI	Erosion Potential
S1	2	90	40	70	85	21.5	11.5	Low
S2	1.66	70	55	30	95	17.5	5	Very Low
S3	4.1	100	70	70	90	21	21	Moderate
S4	2.4	40	50	50	75	24	24	Moderate
S5	2.25	50	90	45	70	20.5	20.5	Moderate
S6	2.5	30	30	120	60	27.5	27.5	Very High
S7	2	100	70	70	40	22	22	Moderate
S8	2	100	100	80	90	18	18	Low

101

Our BEHI scores generally increased with increasing distance from the headwaters. The BEHI scores also represented an increase over the scores from 2013. Despite the general trend of the increasing BEHI with increasing distance downstream (Fig. 2), not all indices showed a similar trend. Neither bank height ratio nor root depth showed any spatial trend. Root density and bank angle both increased and surface protection decreased with increasing distance downstream (Table 2).

Our adjusted BEHI scores increase with increasing distance downstream, but the TFI does not show any such relationship (Fig. 4). Bank height ratio contributed most to the TFI, while surface protection contributed the least.
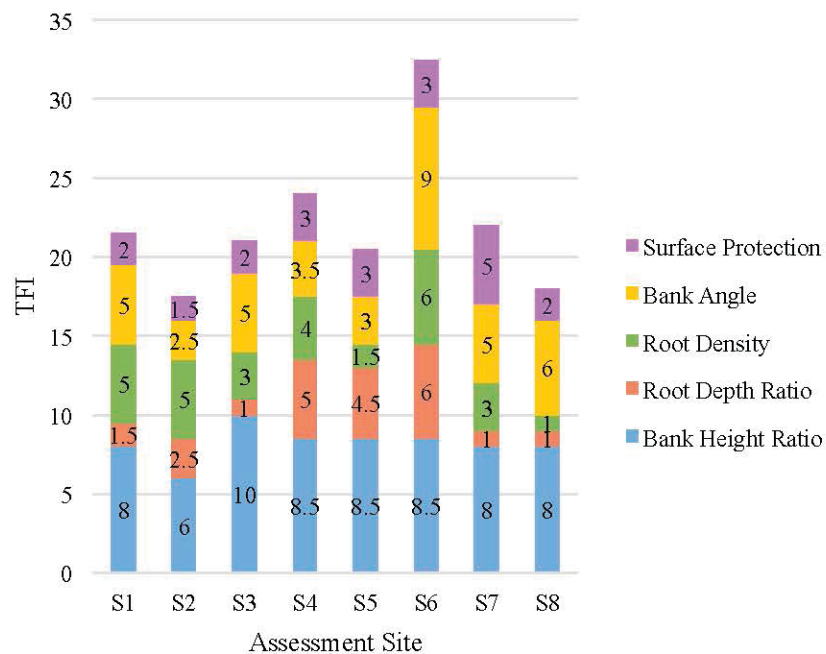
FIG 4. Overall Total Field Index (TFI) and bank aspect measurement indices at each site.

As distance downstream from the headwaters increased, the adjusted BEHI scores generally increased, though this relationship is not significant with an R^2 value of 0.2547 (Fig. 5). A similar trend of increasing BEHI from upstream to downstream sites was observed in 2013. BEHI scores at each site did not change drastically since 2013 with the exception of S6. Only two sites, S4 and S6, had changes in BEHI scores greater than 7.5. Due to the large difference between the BEHI at S6 in 2013 and 2014 and video evidence we reviewed after the BEHI score was calculated, we suspect that the 2014 BEHI score at S6 was inflated due to human error. The BEHI score increased at every site from 2013 and 2014, with the exception of S2. The average score in 2014 was 20.56, while the average score in 2013 was 12.42, a difference of 8.14. The range in scores in 2014 was 37.5, while the range in scores in 2013 was 12.5, a difference of 25.0.
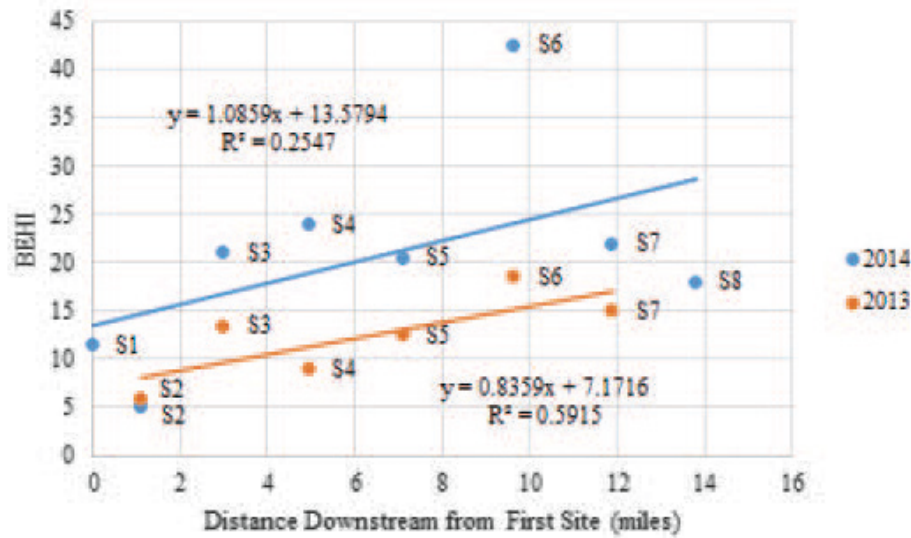
FIG 5. Adjusted Bank Erosion Hazard Index (BEHI) at each site in 2013 and 2014 with increasing distance downstream in Caney Fork.

Adjusted BEHI generally decreased with increasing forest cover on land surrounding each site (Fig. 6). This relationship is not significant with an R^2 value of 0.1060.
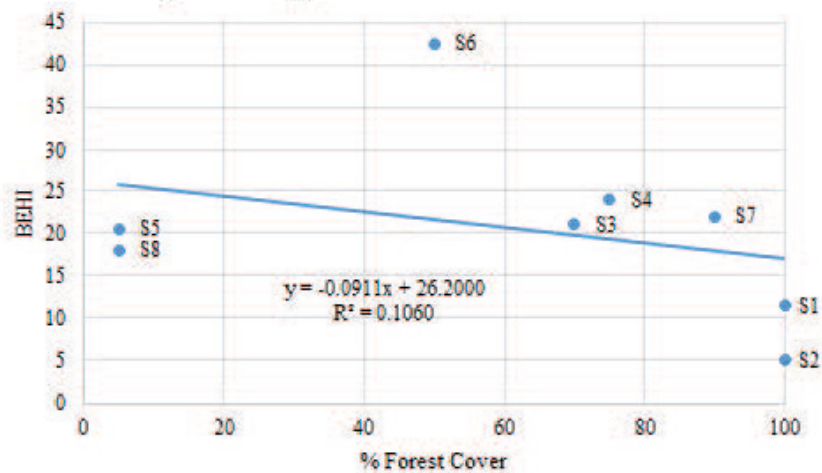
FIG 6. Adjusted Bank Erosion Hazard Index (BEHI) compared to percent forest land cover surrounding each site.

Pebble Count

Since the median particle sizes were recorded within ranges and were not exact numbers, we evenly divided the ranges into the same number of segments with number of pebbles in the ranges to estimate the numbers for the median sizes. The regression slope of median particle size compared to distance from S1 is -5.0922, which means that we expect to see an approximately 5 mm decrease in median particle sizes for each river mile downstream from the headwaters.
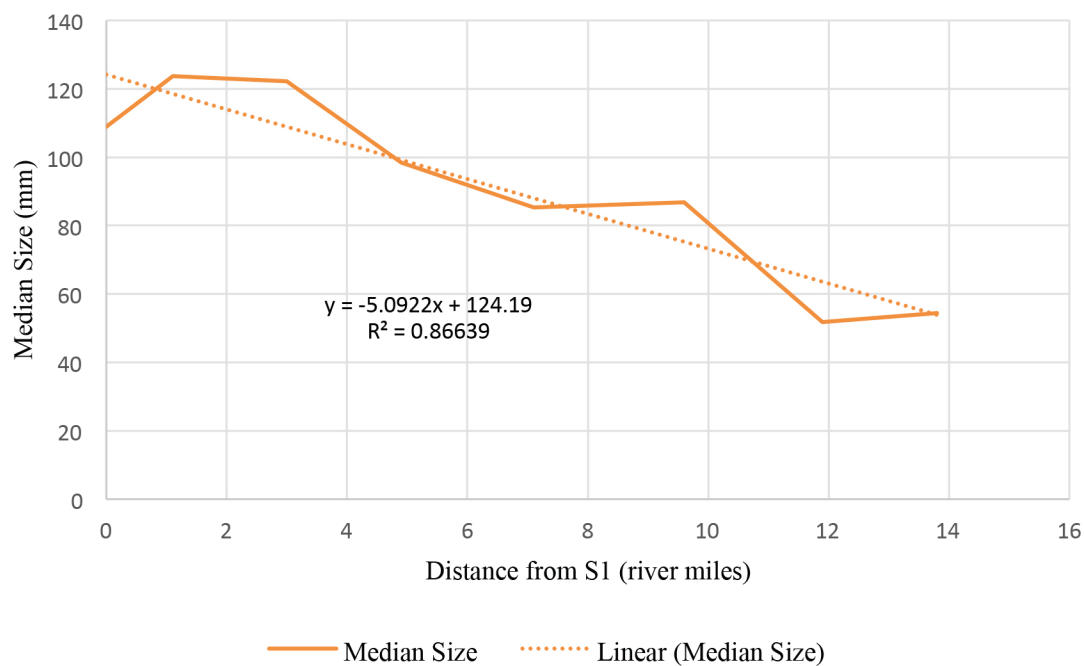
FIG. 7. Median particle size at each assessment site compared to distance from S1.
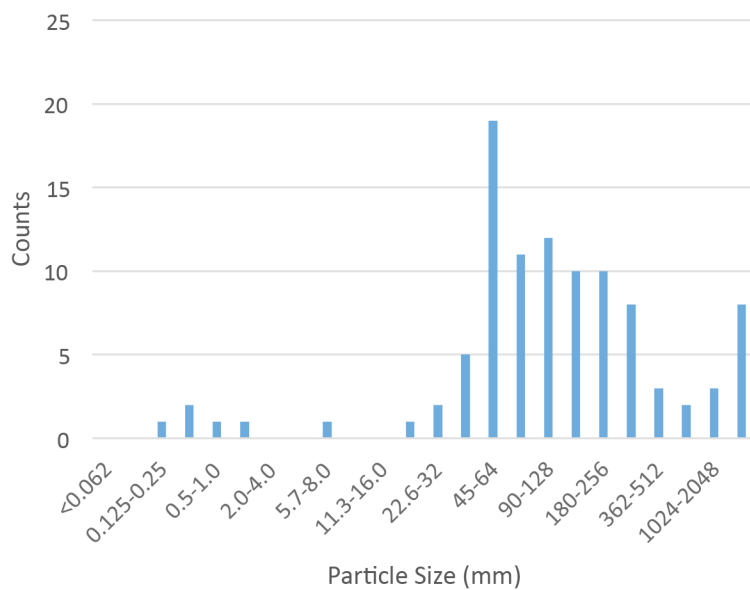
FIG. 8. The particle size distribution at S1.
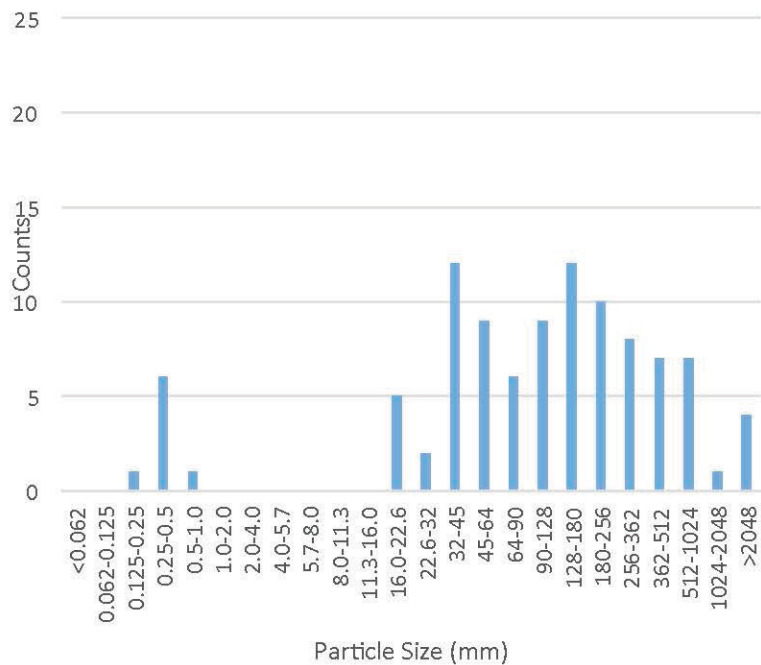
FIG. 9. The particle size distribution at S2.
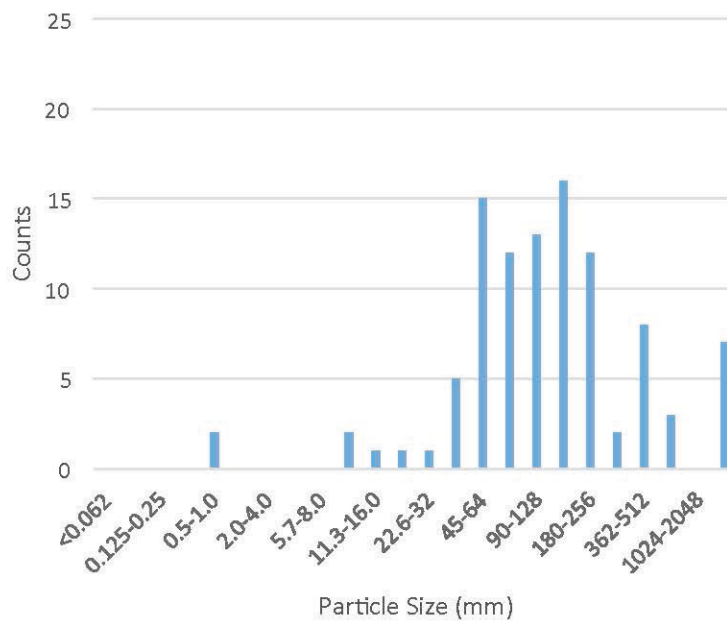
FIG. 10. The particle size distribution at S3.
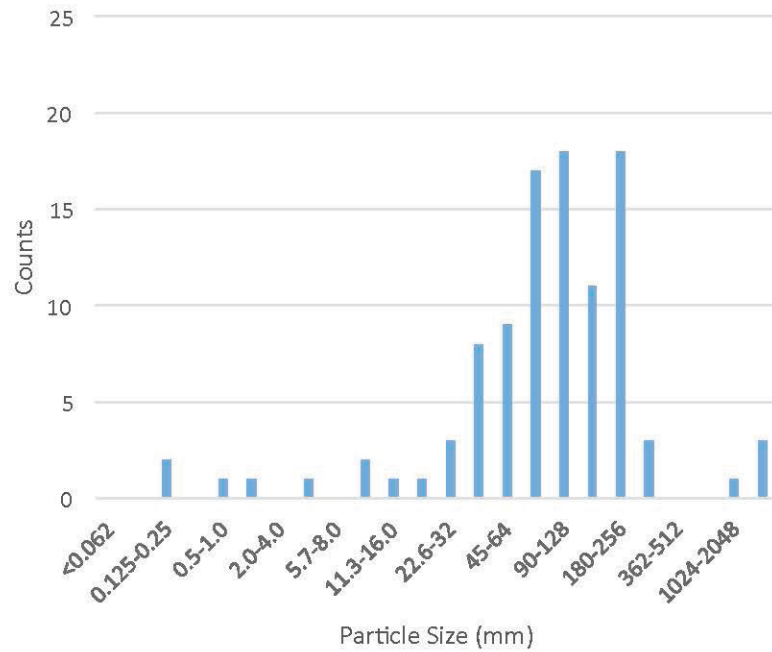
FIG. 11. The particle size distribution at S4.
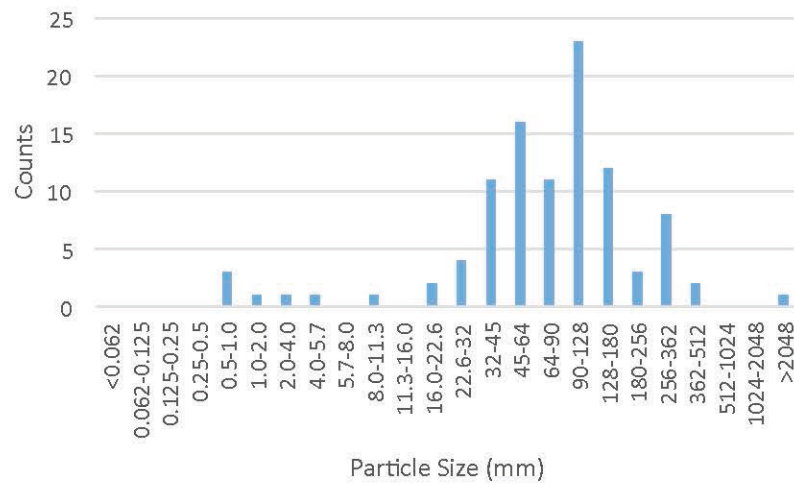
FIG. 12. The particle size distribution at S5.
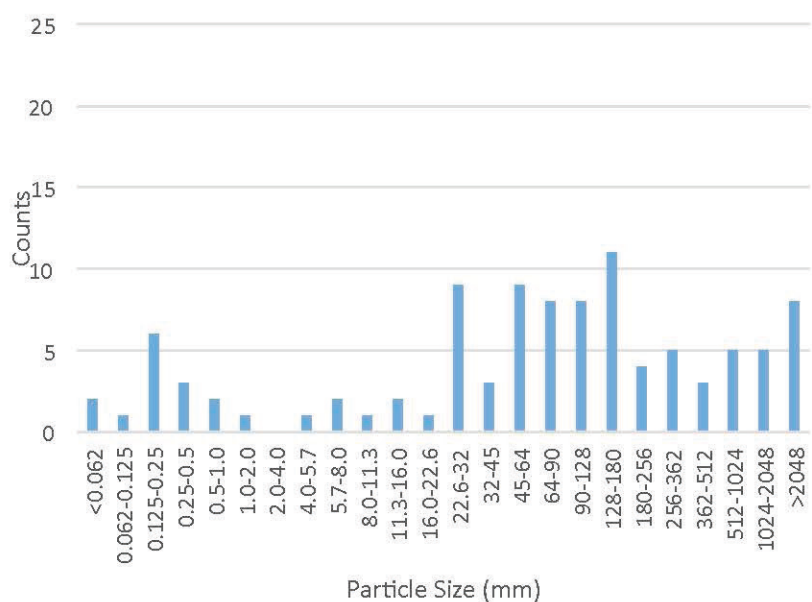
FIG. 13. The particle size distribution at S6.
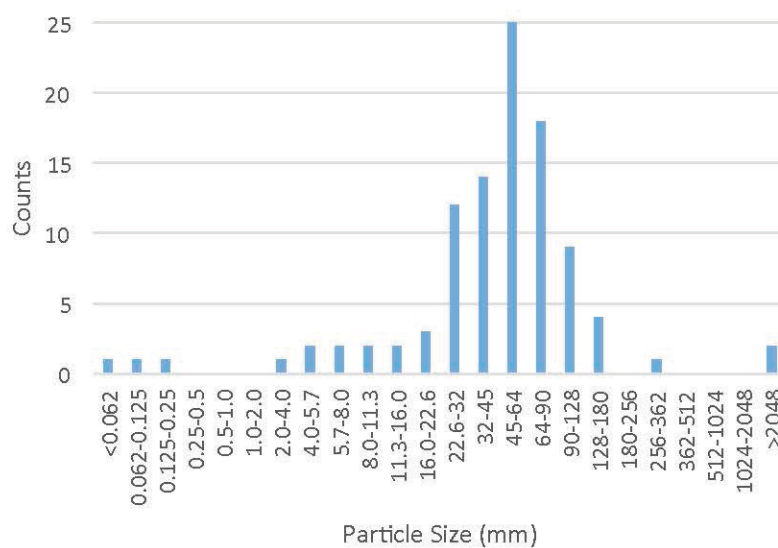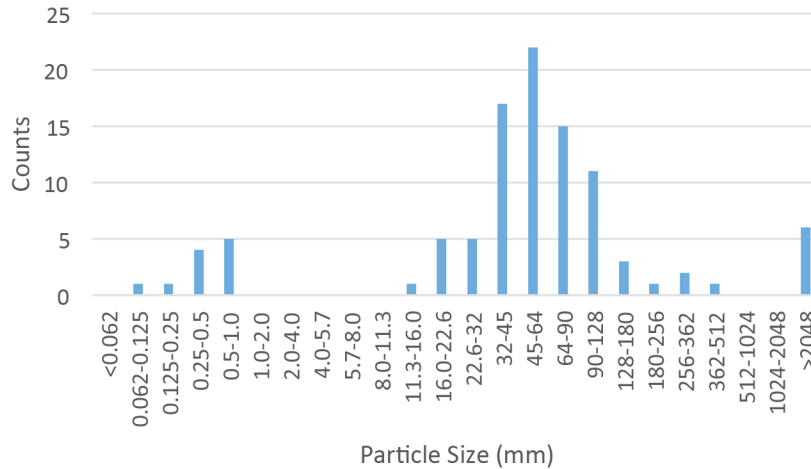

FIG. 14. The particle size distribution at S7

FIG. 15. The particle size distribution at S8.

Upstream sites tend to have wider distributions of different particle sizes (Fig. 8-15). There are significant decreases in the presence of large cobbles and boulders in downstream sites from S4 to S8. Therefore, upstream sites tend to have more evenly distributed patterns of sizes from coarse gravels to bedrocks but downstream sites have a more concentrated size distribution at very coarse gravels. Among the downstream sites, S6 is an outlier that has an even distribution of particle sizes and a higher median particle size than most other downstream sites (Fig. 13).

We plotted cumulative percentage charges and compared them to last year's data to examine changes in particle size (Fig. 16-23). In the charts below, reference percent represents the cumulative percentage we sampled from the single site and study percent represents the cumulative percentage we took from all eight sites. Sands with smaller diameter than 2mm and bedrock larger than 2048 mm in size were not recorded in the cumulative percent charts, so they are responsible for the blank space at the bottom and top of the diagrams (Potyondy and Bunte 2007).
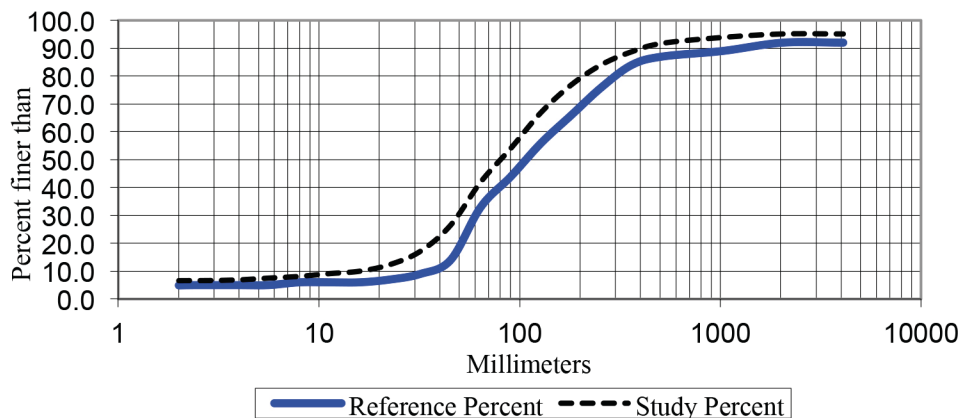

FIG. 16. Cumulative percentage chart for S1 in 2014. S1 was not sampled in 2013, so no comparison can be made. The reference percentage lies under the study percent because we picked up more bedrocks than average at the top site.
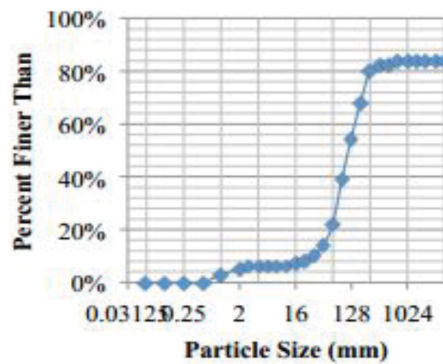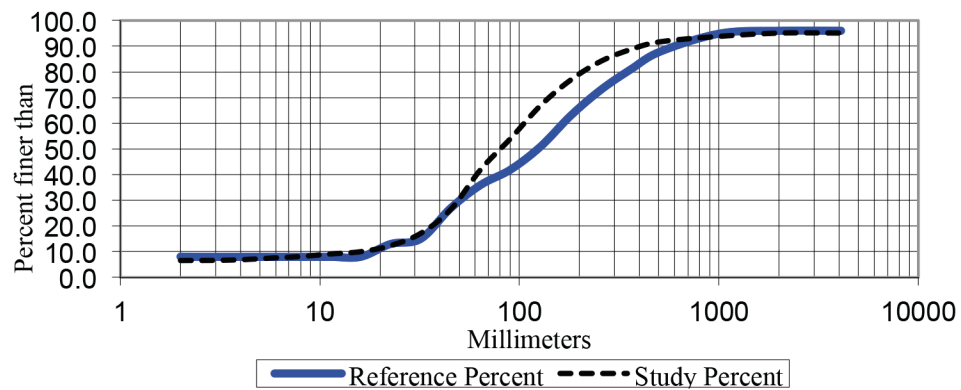
108

FIG. 17. Cumulative percentage chart for S2 in 2014 and 2013. More particles were found within the size range of large cobbles and small boulders. The pattern in 2014 on this site does not vary too much 2013 as the most particles also appear in the size range between around 90 mm and 256 mm. The diagram stops at just over 80% probably because they touched more bedrocks when they were sampling the data last year.
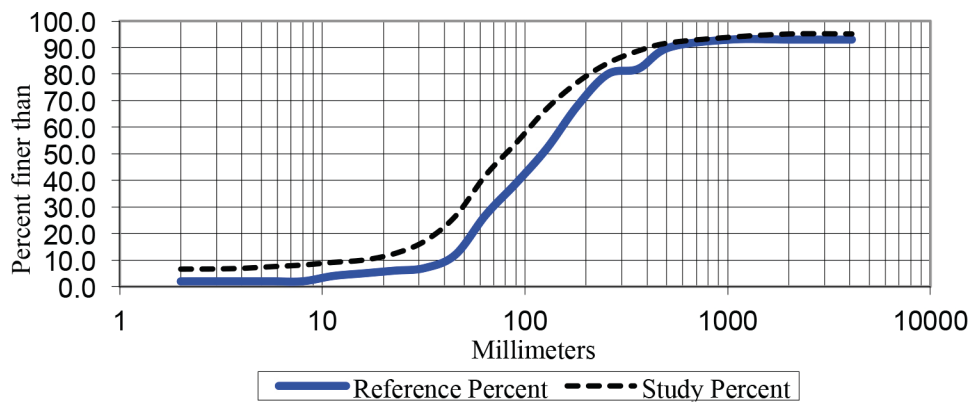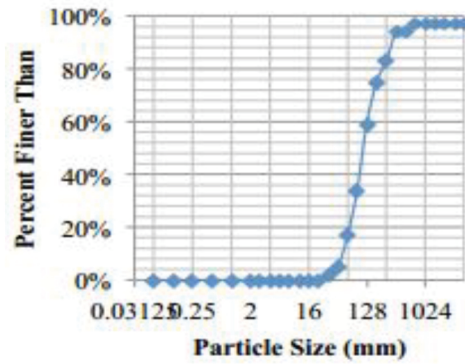
FIG. 18. Cumulative percentage chart for S3 in 2014 and 2013. The chart still shows the characteristic of upstream sites, where reference percentage line lies under the study percentage because more bedrocks were picked. Comparatively both charts start their slopes at around 50 mm and end at around 250 mm. No bedrocks were picked up last year, so the diagram reached 100%.
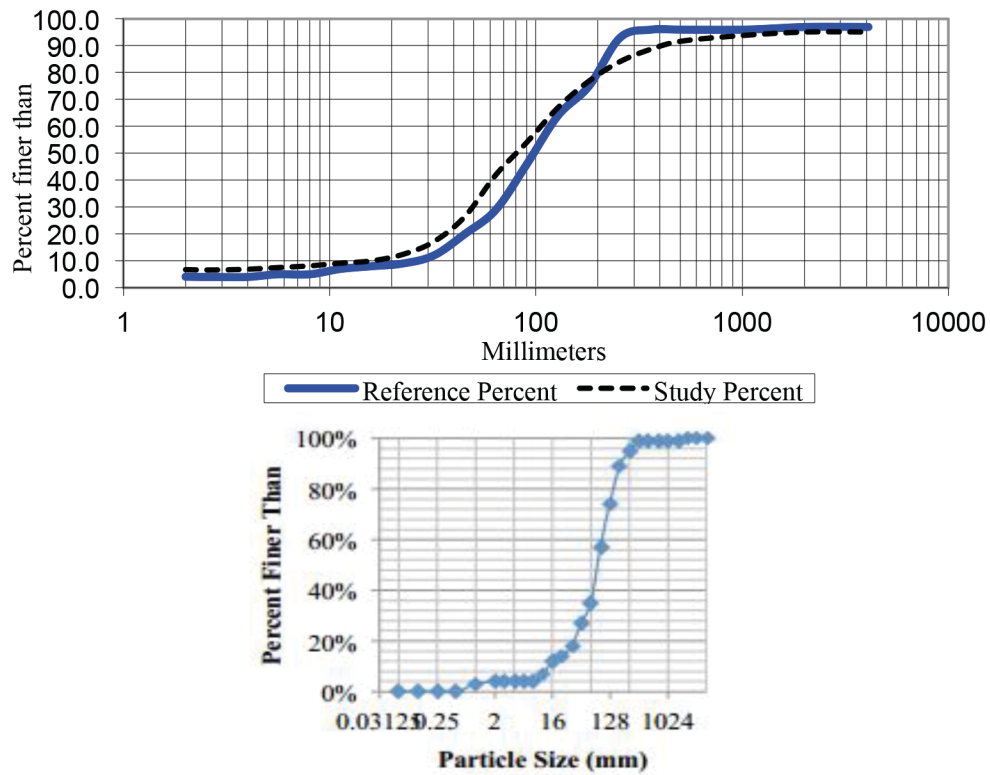

FIG. 19. Cumulative percentage chart for S4 in 2014 and 2013. The reference percentage line intersects study percent line at around 200 mm, indicating more large cobbles and less boulders and bedrocks were found. We can see similar changing patterns on the chart from last year.
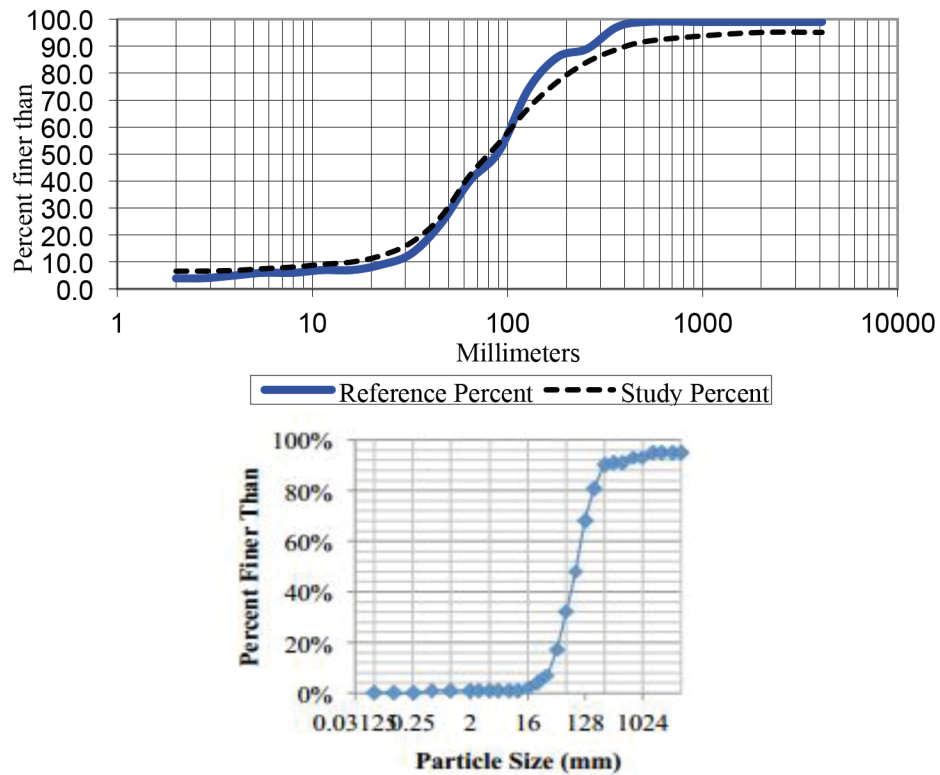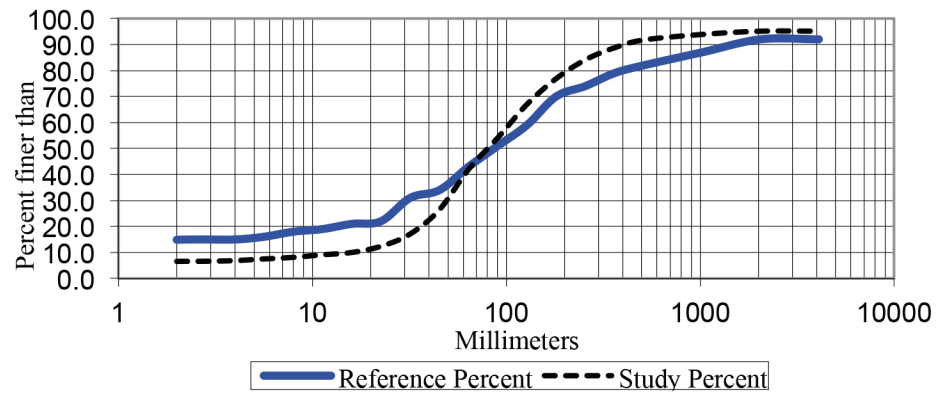
110

FIG. 20. Cumulative percentage chart for S5 in 2014 and 2013. The intersection moves backwards to 100 mm, indicating less larger particles were recorded than upstream sites. Differences were seen from last year that much less bedrocks were picked at Site 5.
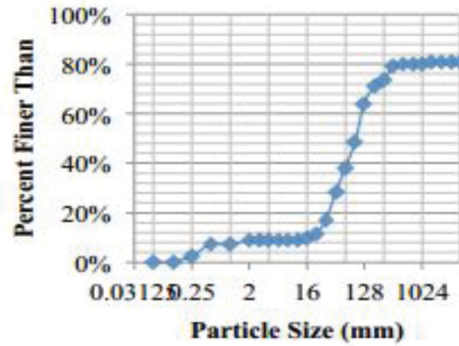
FIG. 21. Cumulative percentage chart for S6 in 2014 and 2013. Site 6 is an outlier from the trend we got from all other sites, where the pebble sizes were most evenly distributed. We can see it from the flat slope we see on the reference percentage line. We do not see the flattened slope on the diagram from last year. Reasons for the huge change in Site 6 remain to be discussed.
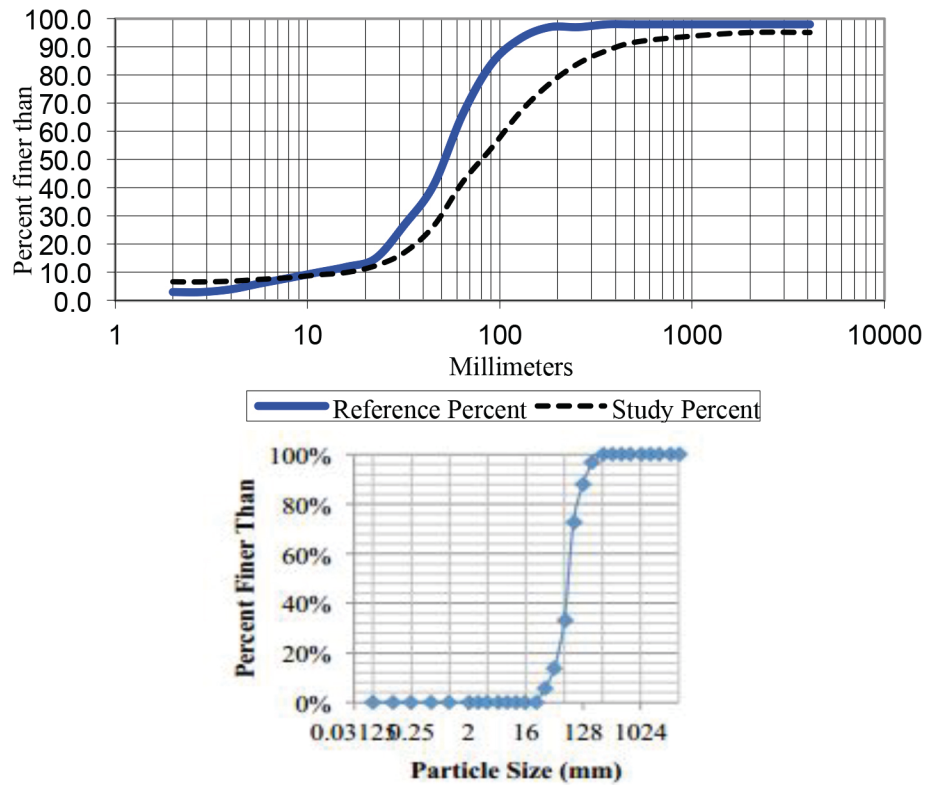

FIG. 22. Cumulative percentage chart for S7 in 2014 and 2013. Significantly more gravels and small cobbles were picked up at Site 7 as the slope starts and ends at smaller sizes than the study percentage of the 8 sites. The data from last year generally displays the same pattern of most particles less than 200 mm wide.
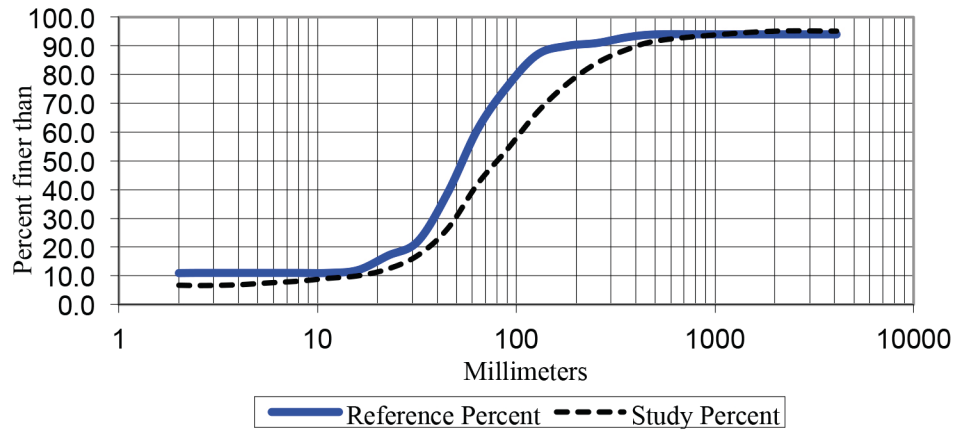
FIG. 23. Cumulative percentage chart for S8 in 2014. S8 was not sampled in 2013, so no comparison can be made. Site 8 shows the same properties of a downstream site as S7 does, with most of the particle sizes lie in 40-100 mm.

Overall, from the data we collected from the eight sites in two years we saw most patterns and distributions remain mostly unchanged throughout the whole year (Fig. 16-23). Conspicuously we observed a decreasing trend in pebble sizes as we go downstream.

We took data from different zones on the river as reach-wide pebble count analysis: riffles, pools, runs and cascades (Fig. 24). The runs had the smallest median sizes overall
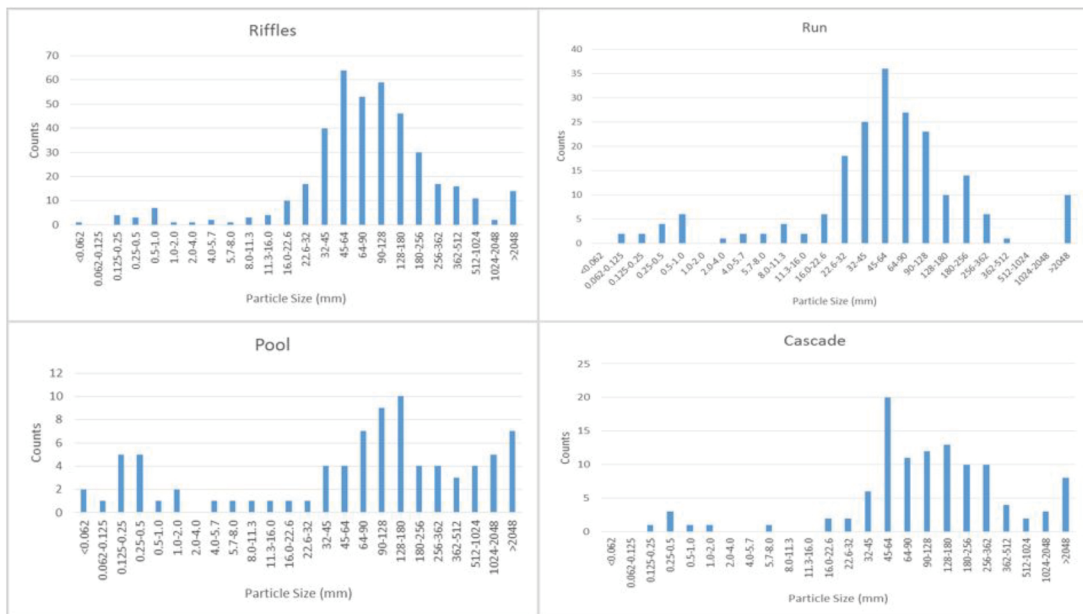

FIG. 24. The distribution of pebble sizes in each type of the stream zone.

(Fig. 25). Cascades have larger median particle size because they were mostly located in upstream sites. The size distribution in the pools appear to be most evenly distributed. Median particle size was also compared to slope degree and distance from the headwaters (Fig. 26).
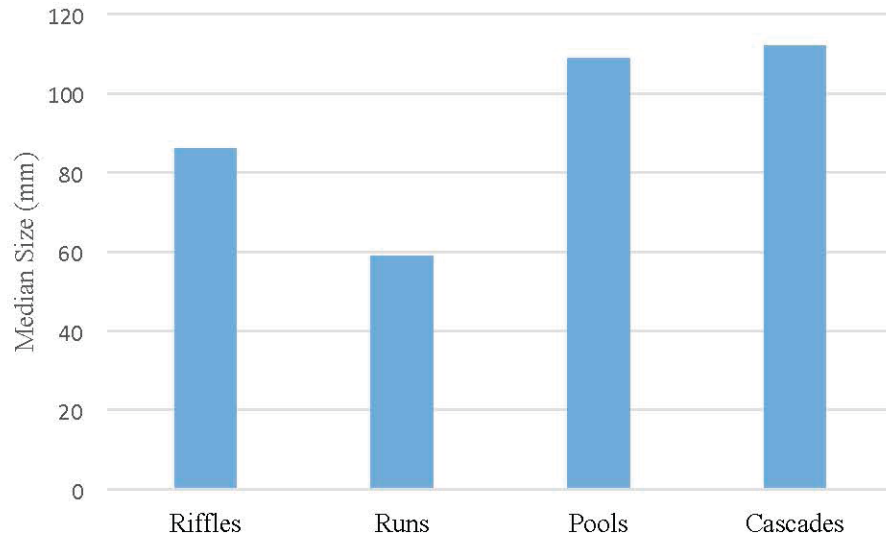
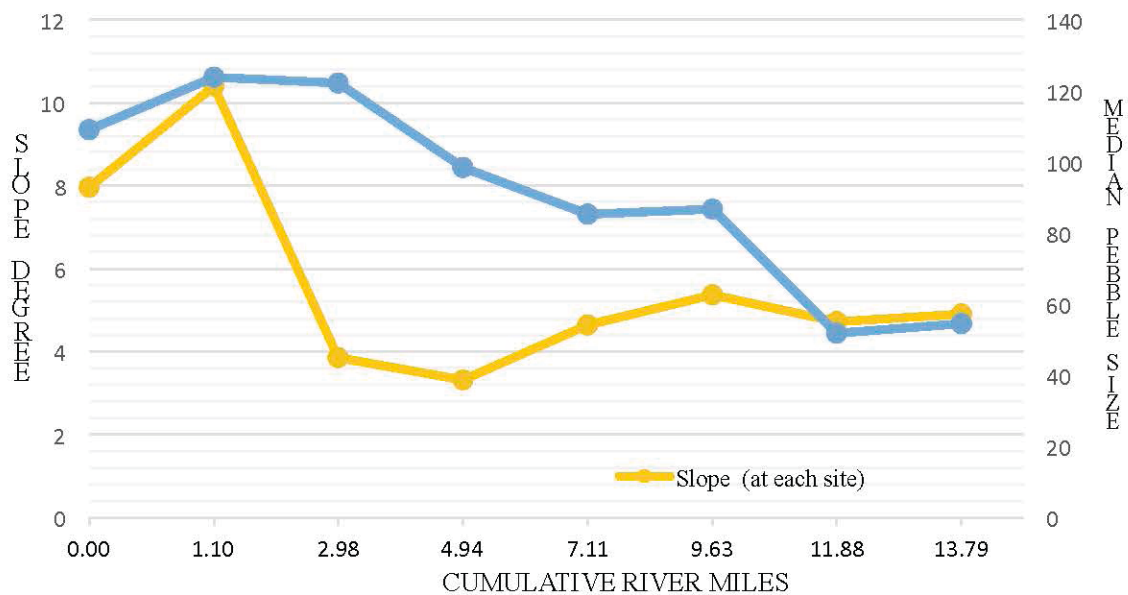FIG. 25. Median size of particles in each type of the stream zone.

FIG. 26. Caney Fork slope degree and median particle size at each assessment site compared to distance along the stream from S1.

Macroinvertebrates

We identified the Family Biotic Index (FBI) for each family of Ephemeroptera, Plecoptera, and Trichoptera (EPT) (Merritt 2008) (Fig. 27). The FBI value directly correlates with a family's ability to tolerate pollution levels within a body of water. Low values indicate a low level of tolerance, and higher FBI values indicate a higher level of pollution tolerance. We compared our FBI values for each site to those found in the 2013 study (Fig. 28). We also examined the correlation between the most prominent feeding types of each family and the surrounding habitat of the respective sites at which they were found (Fig. 29). The highest number of macroinvertebrates was observed at S3 (Fig. 27). At S2, S3, and

114

S4 the most abundant family was Peltoperlidae (roach-like stonefly) of the order Plecoptera. Peltoperlidae are shredder-type feeders and have a biotic index value of 2. At S5 and S8 the Trichoptera Hydropsychidae (common netspinner caddisfly) was the most abundant family. Hydropsychidae are filter-feeders and have a biotic index of 4. At S6 and S7, the most abundant species was the Ephemeroptera Heptageniidae (flat-headed mayfly), scrapers with a biotic index of 4.
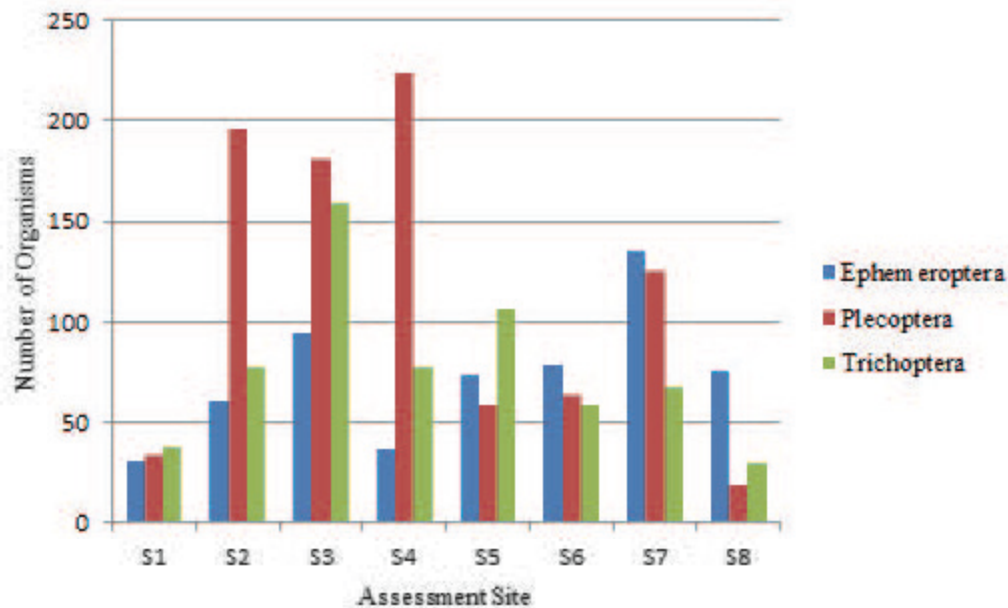
FIG. 27. Number of individuals by order from each site from 2014.

The biotic indices of the identified macroinvertebrates ranged from 0 to 4. The overall FBI for Caney Fork was 2.460, which is a 0.108 decrease from the 2013 data (Fig. 28). A lower biotic index indicates that a species or group of species are less tolerant to pollution. This relatively low score and 2013's score rank the Caney Fork as a healthy stream.

FIG. 28 Weighted Average FBI for each site in 2014 compared to 2013 data.

Scrapers were most abundant at S5, S6, S7, and S8 with 30.6%, 32.8%, 37.8%, and 50% respectively. Shredders were most abundant at S1, S2, S3, and S4 with 31.0%, 48.4%, 33.4%, and 59.4% respectively (Fig. 23). Filterers, gatherers, predators, and collectors made up the rest of the population.

FIG. 29. Percentage of feeding groups from each site. Scrapers were most abundant at S5, S6, S7, and S8. Shredders were most abundant at S1, S2, S3, and S4.
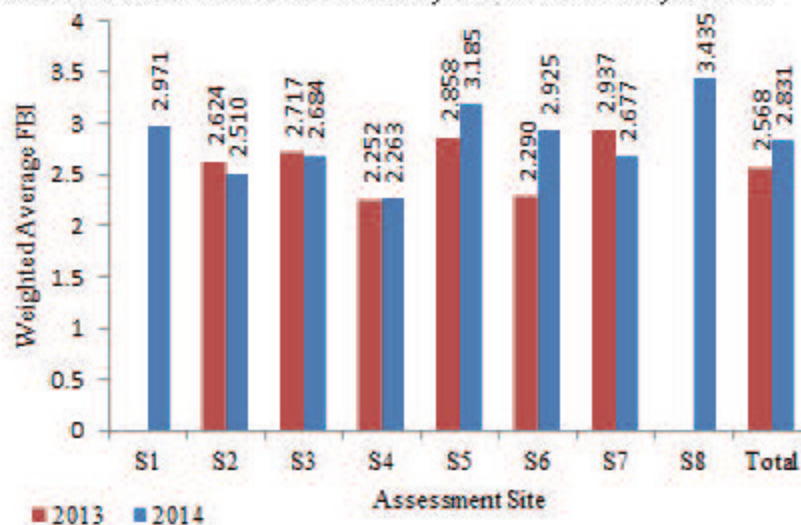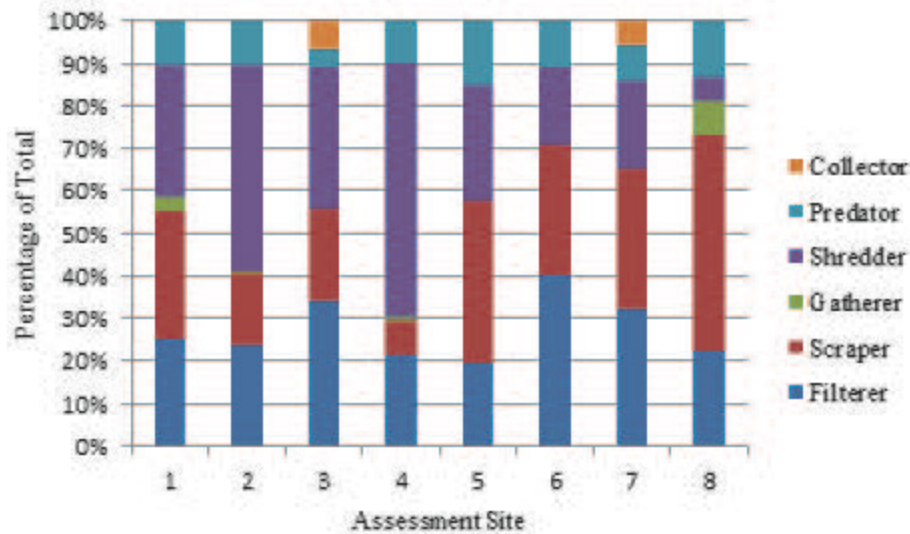
DISCUSSION

Habitat

The overall habitat assessment scores decreased with increasing distance from the headwaters (Fig. 3). This trend indicates that the habitat is more stable near the headwaters and less stable near the confluence of the Caney Fork River and the Tuckasegee River. We define river stability by using the eight categories listed in Table 1. Each category represents a factor of the stream habitat that contributes to the stability of the river. Less stable habitat downstream is, in part, due to the presence of human development, which can degrade the natural stream channel. Channel modification impacts the quantity and timing of water movements, which is important for any organism that depends on the stream as a source of water or habitat (Maddock 1999). From upstream to downstream, there was an increase in impervious surfaces, riprap, bridges, and roadways adjacent to the stream.

The total habitat assessment scores decreased from the headwater site to the mouth both in 2013 and in 2014, but in 2014 the decrease in scores was not as steep. Sites 5 and 7 were significantly different, with respect to the habitat assessment scores, this year compared to last year. Site 5 showed a decrease in total habitat assessment from 2013 to 2014. This could be due to a number of reasons including increased development, a difference in surveying techniques, or a slightly different reach studied. Site 7 showed a considerable increase in the total habitat assessment score from 2013 to 2014. In 2013 pool variety at site 7 was scored as 0 out of 10 while in 2014 pool variety at site 7 was scored at 6 out of 10. In 2013, sampling may have

occurred when the water level was relatively low, resulting in no obvious pools or a different reach may have been sampled. To corroborate this, in 2013, the bottom substrate consisted of gravel and cobble, whereas in 2014 the bottom substrate consisted of boulders, gravel, and cobble. These discrepancies suggest that different reaches at site 7 were surveyed in the two years. For all sites in both years, the left bank of the stream had the same or greater amounts of vegetation than the right bank. The road is primarily on the right side of most of the eight reaches, and likely accounts for this observation.

When deciding if stream restoration is a priority, only surveying habitat in a few reaches may be misleading (Habberfield 2014). We assessed just eight reaches in a 13.8 mile long river, and if we had surveyed more reaches, our results would be different. Another source of error in our sampling methods is the number of people who conducted the habitat assessment. The habitat assessment was obtained by 12 different individuals with different interpretations of habitat stability, which leads to fluctuation in the results.

Bank Erosion Hazard Index

As we sampled sites farther from the headwaters, we generally saw higher BEHI scores (Fig. 5). There were notable differences in strata, surface protection, and land use among the sites. At S1 and S2, the TFI was adjusted downwards due to the substrate type; the primary stratum was cobble at S1 and bedrock at S2. The transition from bedrock near the headwaters of a stream to smaller sediments farther down is explained by the river continuum concept. The higher reaches of a stream have a greater slope, leading to increased velocity and competence. Smaller sediments are washed downstream, leaving bedrock and boulders uncovered. Farther downstream, the stream is flatter, so it has lower competence. Small sediments will drop out and accumulate in slow-moving sections (Foster, pers. comm.). Landowners along slow-moving, sandy sections of the stream far from the headwaters should be more concerned about erosion.

Surface protection at S1 and S2 was mostly bedrock and vegetation. This high degree of surface protection reduces the amount of sediment threatened by stream erosion, leading to a lower BEHI. Below these two upstream sites, the stream has been affected by human development, resulting in less surface protection in general (Fig. 5). Of the surface protection that was present, less was vegetative or bedrock and more was riprap. These findings indicate that human development negatively affects the presence of surface protection and erosion.

Vegetation in riparian zones provides high geotechnical reinforcement in stream banks and protects against bank scour, so it plays an important role in stream bank stabilization (Wynn et al. 2014). Non-vegetated stream bends are almost five times more likely to undergo detectable erosion than vegetated bends (Beeson and Doyle 2007). Because bank vegetation is generally considered to be effective in reducing erosion, we expected the BEHI to decrease as the percent forest land cover surrounding the stream increased. Though weaker than other studies (Beeson and Doyle 2007, Wynn et al. 2014), this relationship was present in Caney Fork (Fig. 6). The sites with the lowest BEHI, S1 and S2, were the only sites with 100% forest cover. At sites farther downstream, more area was residential or alongside a road and less was forested. These results suggest that if landowners of these farther downstream sites plan to reduce erosion, they should increase the amount of riparian vegetation.

The sites with the highest BEHI, S4 and S6, were dominated by residential areas and fallow fields, land uses characterized by less vegetation and shorter root depth. Grassland stream sections such as S5, S6, and S8 are known to have higher rates of erosion (Laubel et al. 1999).

Vegetation clearance or land use changes can cause increases in sediment yield of an order of magnitude or greater (Walling 1999), especially when cattle are present. Cattle grazing is an important factor in bank breakdown, resulting in six times as much gross bank erosion as in protected streams (Trimble 2006). This may explain the high BEHI at S6 because it had 45% active pasture land use. However, it is important to note that some studies have found no significant relationship between stream bank erosion and land use (Laubel et al. 1999), and some have even shown that forested stream banks are more prone to destabilizing stream channels (Trimble 1997). The lack of an established relationship across all land uses and stream bank erosion potential may help explain the low relationship between BEHI and forest cover in Caney Fork.

BEHI scores between 2013 and 2014 assessments increased in both value and range. We saw no obvious changes to the landscape at each site; no recent construction, deforestation, vegetative changes, or natural disturbances such as flood, fire, or storms were visible. Therefore, we suspect that changes in BEHI scores were likely due to the subjective nature of assessment by individuals. Different individuals calculated BEHI each year and at each site. Additionally, some assessments may have been based on a specific part of the reach, and individuals may have inspected different sections than those used in 2013. We suggest that future research evaluates BEHI every several years for at least ten years to better characterize changes in erosion with respect to human disturbance. Cataloging land use along Caney Fork over time would also aid this analysis.

In summary, BEHI increased with increasing distance from the stream headwaters due to changes in sediment composition, surface protection, and land use. BEHI decreased with increasing forest cover. Changes in the BEHI scores since 2013 are likely not due to human disturbance.

Pebble Count

Pebble counts are instrumental in determining the competency and capacity of streams as well as suitability of habitat for many stream organismal groups such as macroinvertebrates and fish. Distribution of vertebrate and invertebrate populations on the micro and macro scale within stream ecosystems are greatly influenced by substrate composition and distribution (Komar and Carling 1991).

The increase of fine sediment within streams is a common problem that contributes to habitat degradation as it often changes the composition of the substratum (i.e. sedimentation). A reduction in substratum heterogeneity is detrimental to aquatic life and diversity as it is responsible for limiting the availability of interstitial spaces used by benthic organisms in foraging as well as cover and nesting habitats (Keitzer and Goforth 2012). Specifically, benthic fish as well as Ephemeroptera, Plecoptera, and Trichoptera diversity and numbers are reduced from increased sedimentation as they prefer a heterogeneous mixture of gravel and cobble. In fact, other species previously unknown have recently been found to be impacted by increased sediment input. Keitzer and Goforth (2012) determined that the number of larvae from two salamander species (*D. quadramaculatus* and *E. wilderae*) decreased with greater stream sediment when compared to moderate stream sediment.

In our study, median particle size decreased substantially from a study-wide high of 123.8 mm at S2 down to a low of 51.8 mm at S7, likely affected by decreasing stream gradient (slope degree; Fig. 20) and somewhat by anthropogenic channelization. Thus, geologic variables

such as slope and elevation variability within our study contribute to changes in stream morphology and substrate heterogeneity.

Elevation gradient was essentially twice as steep at S1 and S2 as at the other six sites. These sites were markedly cascade-prominent environments high in the watershed with lowered impacts from human land-use while the others were comprised of a relatively proportionate combination of riffles, runs, and pools, all with increased land-use development.

S6 stood out as an outlier (and was not last year) likely due to the area being a grassland stream section with a higher rate of erosion (Laubel et al. 1999). As noted earlier, the reach area at S6 had 45% active pasture land use, presumably leading to erosion of a variety of pebble sizes and sediments into the stream area. This explains the more evenly distributed particle sizes.

Shurgar (2007) notes that landscape and catchment scale processes are more likely to influence the hydrologic regime and sediment transport capacity, while riparian canopy and cover are known to affect the delivery of organics, amount of light penetration, and influence local geomorphic conditions. Furthermore, it is believed that this inconsistency in the data between the two years is due to significant alteration of the form of landscape and catchment scale process by increased erosion and runoff from active pasture land use.

We found several notable differences between last year's pebble count and this year's. Comparing median substrate particle size to last year, the results were very similar except for S7. Last year it was 75.05 mm compared to 51.8 mm this year. This may be due to increased alteration of stream banks, riparian zones, and/or increased flood events thus affecting competency and capacity. Reach-wide pebble count sites were kept constant, thus other factors are responsible for differences in findings between the two years. Although pebble count procedure was kept uniform, differentiation in data theoretically could be from particular pebble measurement site differences as the total number of comparable measurements for the entire study was only 600, which is few. Small differences in cross-sectional placement between groups performing the pebble count or individual measurement technique could also result in human error and slight variance in technique is possible and assumed to a certain degree.

Significant habitat degradation along Caney Fork due to anthropogenic land use and riparian zone alteration may contribute to less healthy riffles. Riffles are especially important to examine because they are critical locations to identify changes in particle size distribution due to their sensitivity to increased sediment supply as discharge rates increase here (Parker and Klingeman 1982). We found that particle size homogeneity within riffles increased slightly at S7 and S8, however, among the other sites, riffle particle size heterogeneity was high. Higher particle size homogeneity at S7 and S8 may be due to the sites being relatively straight and having fewer boulders (causing cascade effects) and riffles.

Riffles are especially important to stream ecosystem health. Riffle habitats are particularly sensitive to added sediments from anthropogenic sources as these habitats are characterized by shallow zones with more turbulent discharge, high oxygen levels, and substrates comprised of unconsolidated particles. Riffle substrates tend to be more heterogeneous allowing a wide variety of niches for macroinvertebrates to inhabit. Additionally, the greatest levels of aquatic diversity tend to occur in riffles. Therefore, the loss of heterogeneity in riffle substrates due to increased sedimentation or channel armoring (due to saltation of a wide size range of particles) can result in more homogeneous substrates. The loss of variability in habitats in these streams may lead to declines in macroinvertebrate species richness, abundances, and distributions (Landrum and Tolley-Jordan 2013).

Overall, the distribution of particle size within the riffles we sampled was among the most homogeneous compared with the other stream sections (runs, pools, cascades) we sampled. This may become more pronounced as anthropogenic land-use is increased in the future. Further studies may help delineate the weight of importance of riffles and other stream sections and their corresponding particle size matrix by noting in which section macroinvertebrates were found and comparing against the particle size distribution.

Macroinvertebrates

Macroinvertebrates are key organisms that reflect the quality of their environment and respond to human disturbances in fairly predictable ways (EPA 2006). We consistently found an abundance of organisms within the orders of Ephemeroptera, Plecoptera, and Trichoptera throughout our stream sites, indicating that these sites are all healthy enough to support a number of pollution-intolerant invertebrate species. Of these three orders, Plecoptera was most abundant at S2, S3, and S4, Ephemeroptera was the most abundant at S6, S7, and S8, and Trichoptera was the most abundant at S1 and S5 (Fig. 21).

Macroinvertebrates have specialized feeding groups based on their aquatic environment, and as the stream changes, the distributions of feeding groups were found to change as well (EPA 2006). We observed a distinct gradient in abundance of different feeding types as the stream progressed from S1 to S8. The upstream sites, S1 through S4, were primarily inhabited by shredders, while the invertebrates at the downstream sites, S5 through S8, were primarily scrapers (Fig. 23). With increased human alterations to the stream, there tends to be an increased abundance of scrapers and a decreased abundance of shredders, due to the differences in food sources present (Karr 1991). Our upstream sites generally have more leaf litter and larger suspended particulate organic matter due to the greater presence of canopy cover (Foster, pers. comm.).

FBI value indicates how tolerant a respective family is to pollution, and if this level is exceeded, the organism cannot survive in the polluted area (EPA 2006). Families with a low FBI value can only be found in streams with good water quality. We expected to see some form of correlation between stream location, pollution levels, and FBI of the invertebrates assessed, however in reality, little association was actually found. In addition, when assessing the FBI of the river sites in 2013, Odonata, Coleoptera, Diptera, and Megaloptera were considered along with the previously mentioned EPT. The non-Ephemeroptera, Plecoptera, and Trichoptera orders assessed in 2013 include many that have generally higher FBI values than the EPT assessed in 2014. Because of this discrepancy between the families assessed in 2013 and 2014, it is difficult to extrapolate a change in stream health of our sites across time based on the FBI alone. Even though we only looked at three orders of macroinvertebrates, healthy streams have a high level of diversity within these orders.

As previously stated, the FBI values of macroinvertebrates are assigned based on their tolerance to pollution within a body of water (William 2002). The most abundant families at the sites closest to the headwaters (S2, S3, and S4) were found to have slightly lower FBI values, while the most abundant families at the sites further downstream had slightly higher FBI values. This indicates that the sites upstream may be healthier, due to their lower FBI. Factors such as light availability, groundwater inflow, and leaf litter availability all affect stream health in various ways. These can impact available macroinvertebrate habitat, the presence of dissolved nutrients, and organic matter abundance (Sweeney 1993). Overhead canopy cover shades the

stream and provides woody debris to act as habitat for macroinvertebrates (Sweeney 1993). These factors all influence the survivorship and abundance of aquatic macroinvertebrates. Increased anthropogenic land use on the Caney Fork as the stream progresses from its headwaters may explain the observed trend in increasing FBI.

CONCLUSION

Based on each assessment we performed at the eight sites along Caney Fork, our results were mostly consistent with what we expect according to the River Continuum Concept. For each assessment we used procedures as similar as possible to those of Allan et al. (2013) and to observe any trends in the data that could indicate a change over a one year period. In Caney Fork the only significant changes from 2013 to 2014 we observed were in BEHI assessment. The results of habitat assessment and pebble count are consistent with the findings of Allan et al. (2013) and described similar conditions at each of the common sites. There did not seem to be any increase in development that drastically changed the results from 2013 to 2014. However, there were trends in increasing land use and gaps in the observed woody vegetation as distance from the headwaters increased. When comparing our assessment of Caney Fork to that performed by the NC Division of Natural Resources Water Quality Division in 1992, our results tend to indicate that Caney Fork is a stable and healthy tributary to the Little Tennessee River based on an overall assessment of habitat and biotic health.

ACKNOWLEDGEMENTS

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Average density, basal area (BA), foliage biomass, and total aboveground biomass (leaf + wood) of tree and shrub species (≥2.5 cm dbh) in Watershed 10. Values are followed by their standard error in parentheses. Species nomenclature follows Kirkman et al. (2007).

Year	Species	Density (stems ha^{-1})	BA (m^2 ha^{-1})	Foliage Biomass (kg ha^{-1})	Total Biomass (kg ha^{-1})
1934	*Castanea dentata*	832.95 (92.89)	13.1861 (1.7021)	1590.36 (207.43)	124832.41 (21401.45)
	Quercus montana	285.8 (37.31)	3.5534 (0.6148)	512.97 (99.39)	33041.96 (9065.90)
	Carya spp.	155.11 (22.65)	1.6668 (0.3722)	226.07 (57.67)	15492.44 (4385.93)
	Acer rubrum	282.67 (39.25)	1.6107 (0.2732)	279.41 (42.87)	10820.52 (2448.51)
	Quercus rubra	52.56 (8.10)	1.26 (0.44)	186.36 (72.69)	10797.06 (4646.59)
	Quercus velutina	61.93 (10.0)	1.2127 (0.2745)	159.26 (40.83)	8182.91 (2345.81)
	Quercus coccinea	76.7 (14.4)	1.0923 (0.2656)	278.09 (59.45)	6488.46 (1640.50)
	Nyssa sylvatica	125.57 (17.65)	0.9318 (0.1892)	67.92 (9.97)	5479.72 (1555.30)
	Pinus rigida	31.53 (13.70)	1.0591 (0.4988)	76.93 (36.56)	4659.58 (2195.21)
	Kalmia latifolia	256.07 (67.16)	0.8263 (0.2222)	406.89 (109.20)	2810.23 (760.00)
	Rhododendron maximum	248.71 (79.60)	0.8163 (0.2689)	401.43 (131.96)	2787.66 (924.61)
	Quercus alba	19.89 (5.46)	0.3559 (0.2277)	48.24 (26.86)	2442.44 (1824.69)
	Oxydendron arboreum	66.19 (9.10)	0.4898 (0.0730)	42.49 (5.94)	1867.99 (308.96)
	Robinia pseudoacacia	42.33 (6.67)	0.2932 (0.0922)	34.64 (11.06)	1650.33 (670.26)
	Aesculus flava	1.14 (0.89)	0.1475 (0.1308)	17.9 (15.9)	1584.32 (1469.65)
	Quercus marilandica	39.49 (22.53)	0.2545 (0.1558)	24.39 (15.19)	906.44 (574.04)
	Hamamelis virginiana	140.63 (40.61)	0.2845 (0.0822)	32.89 (9.50)	619.71 (178.95)
	Tsuga canadensis	14.2 (5.3)	0.1034 (0.0597)	29.49 (14.19)	483.22 (294.52)
	Castanea pumila	88.64 (29.27)	0.193 (0.065)	22.29 (7.50)	445.62 (152.96)
	Liriodendron tulipifera	32.95 (12.71)	0.1045 (0.0458)	8.83 (3.86)	303.75 (149.33)
	Cornus florida	46.88 (12.40)	0.12 (0.03)	13.28 (3.47)	300.6 (94.8)
	Sassafras albidum	44.6 (8.9)	0.1 (0.0)	11.35 (2.16)	229.18 (44.24)
	Betula lenta	5.4 (2.0)	0.035 (0.012)	2.07 (1.39)	184.03 (126.11)
	Fraxinus americana	12.78 (4.77)	0.0468 (0.0239)	5.41 (2.85)	179.16 (128.31)
	Amelanchier arborea	18.75 (3.69)	0.0455 (0.0090)	5.15 (1.04)	108.87 (24.41)
	Magnolia fraseri	1.99 (0.99)	0.0155 (0.0117)	1.79 (1.40)	75.76 (68.21)
	Symplocos tinctoria	8.24 (4.89)	0.0166 (0.0098)	1.93 (1.14)	36.31 (21.56)
	Prunus serotina	3.41 (1.64)	0.0091 (0.0048)	1 (0)	21.64 (12.63)
	Diospyros virginiana	0.85 (0.63)	0.0018 (0.0013)	0.2 (0.1)	3.76 (2.77)
	Acer pensylvanicum	0.28 (0.28)	0.0007 (0.0007)	0.13 (0.13)	1.67 (1.67)
	Acer saccharum	0.28 (0.28)	0.0007 (0.0007)	0.13 (0.13)	1.67 (1.67)
	Quercus falcata	0.28 (0.28)	0.0007 (0.0007)	0.54 (0.54)	1.56 (1.56)
	Salix nigra	0.28 (0.28)	0.0007 (0.0007)	0.07 (0.07)	1.25 (1.25)
	Lindera benzoin	11.08 (7.31)	0.0223 (0.0147)	0.06 (0.04)	0.06 (0.04)
1952	*Quercus montana*	332.81 (43.56)	5.2348 (0.6901)	739.47 (103.70)	42612.45 (8082.39)
	Carya spp.	187.81 (24.78)	2.2485 (0.4171)	300.88 (63.61)	20415.22 (4920.58)
	Quercus velutina	56.88 (10.75)	1.5555 (0.4168)	221.54 (69.78)	12491.48 (4688.53)
	Acer rubrum	301.25 (33.45)	1.9048 (0.2481)	335.66 (39.41)	11705.15 (2059.31)
	Quercus rubra	83.13 (10.54)	1.2713 (0.2642)	164.22 (38.16)	8337.55 (2142.56)
	Oxydendron arboreum	90.94 (11.05)	1.089 (0.197)	80.9 (11.6)	5390.98 (1378.17)
	Nyssa sylvatica	79.06 (12.43)	0.8518 (0.2610)	58.8 (12.5)	4987.77 (2176.80)
	Liriodendron tulipifera	48.75 (12.05)	0.7063 (0.2199)	58.61 (18.15)	4533.7 (1809.6)
	Rhododendron maximum	1077.81 (232.54)	2.1893 (0.4973)	178.19 (39.29)	4280.57 (947.25)

Quercus alba	40 (7)	0.5705 (0.2826)	81.19 (32.61)	3648.13 (2386.40)
Kalmia latifolia	810.94 (189.03)	1.114 (0.282)	113.31 (27.31)	2655.9 (643.9)
Magnolia fraseri	1.25 (0.60)	0.1808 (0.1196)	21.95 (14.62)	1874.35 (1426.62)
Tsuga canadensis	35 (10)	0.346 (0.135)	92.95 (29.75)	1644.16 (706.02)
Robinia pseudoacacia	33.75 (5.36)	0.2923 (0.0673)	34.37 (8.05)	1434.46 (439.80)
Betula lenta	27.19 (6.22)	0.1628 (0.0475)	10.48 (4.06)	949.17 (382.19)
Castanea dentata	194.06 (30.30)	0.4078 (0.0668)	47.05 (7.73)	915.45 (158.00)
Cornus florida	112.81 (24.57)	0.3138 (0.0664)	34.94 (7.42)	781.37 (169.52)
Pinus rigida	11.88 (6.09)	0.1738 (0.1115)	9.48 (6.82)	651.24 (452.53)
Hamamelis virginiana	103.13 (27.64)	0.2185 (0.0593)	25.24 (6.87)	490.87 (135.13)
Fraxinum americana	27.19 (9.29)	0.1088 (0.0370)	12.67 (4.34)	403.82 (166.25)
Sassafras albidum	46.88 (8.72)	0.1218 (0.0215)	14.05 (2.50)	314.49 (61.75)
Amelanchier arborea	34.69 (7.44)	0.093 (0.025)	10.69 (2.87)	247.89 (84.55)
Asimina triloba	11.56 (6.67)	0.0258 (0.0139)	2.93 (1.59)	58.24 (30.66)
Acer pensylvanicum	1.88 (1.15)	0.0113 (0.0087)	2.03 (1.51)	54.02 (46.84)
Diospyros virginiana	1.88 (1.59)	0.0128 (0.0095)	1.5 (1.1)	44.41 (32.38)
Aesculus flava	0.63 (0.63)	0.0083 (0.0083)	0.97 (0.97)	34.13 (34.13)
Castanea pumila	3.44 (2.57)	0.007 (0.005)	0.8 (0.6)	15.15 (11.33)
Cornus alternifolia	2.5 (2.2)	0.0053 (0.0045)	0.56 (0.49)	9.8 (8.6)
Fagus grandifolia	1.88 (1.05)	0.0043 (0.0023)	0.44 (0.25)	8.26 (4.65)
Quercus coccinea	1.25 (0.98)	0.0028 (0.0021)	2.38 (1.87)	6.85 (5.37)
Ostrya virginiana	0.31 (0.31)	0.0008 (0.0008)	0.07 (0.07)	1.38 (1.38)
2014 Quercus montana	113.95 (13.82)	8.2305 (0.9132)	1241.05 (141.96)	79533.93 (10224.81)
Acer rubrum	239.83 (24.63)	5.0744 (0.4357)	804.87 (66.32)	37076.71 (3643.49)
Liriodendron tulipifera	108.14 (41.30)	3.3712 (0.8253)	278.83 (68.10)	24306.84 (6515.50)
Quercus rubra	11.63 (2.51)	1.6058 (0.4874)	244.72 (78.21)	14144.51 (4765.73)
Oxydendron arboreum	77.33 (9.39)	2.2872 (0.2923)	149.99 (18.12)	12170.47 (1789.94)
Carya spp.	61.63 (11.16)	1.4721 (0.2951)	187.33 (40.35)	11361.56 (2661.31)
Quercus alba	9.3 (2.9)	1.2837 (0.4143)	152.05 (48.78)	9987.35 (3273.47)
Quercus velutina	13.08 (5.44)	1.1267 (0.3014)	168.24 (45.98)	9509.83 (2655.23)
Quercus coccinea	19.77 (5.71)	0.9316 (0.2585)	137.92 (36.42)	6995.59 (2148.81)
Betula lenta	35.17 (6.10)	0.8763 (0.2405)	72.62 (22.01)	6758.44 (2076.77)
Kalmia latifolia	1261.63 (301.78)	2.6947 (0.5616)	218.08 (48.87)	5269 (1172)
Rhododendron maximum	968.31 (178.50)	3.2633 (0.6269)	202.43 (37.38)	4997.49 (924.49)
Nyssa sylvatica	42.73 (8.81)	0.8956 (0.1880)	59.5 (11.7)	4942.3 (1178.5)
Robinia pseudoacacia	29.36 (9.00)	0.6395 (0.1842)	76.72 (22.34)	4849.66 (1889.7)
Prunus serotina	127.33 (79.77)	0.5502 (0.3071)	65.5 (36.7)	4235.87 (3099.82)
Pinus rigida	7.56 (3.56)	0.5809 (0.2766)	45.97 (22.82)	2733.39 (1330.32)
Amelanchier arborea	48.55 (13.49)	0.4826 (0.1277)	57.11 (15.10)	2254.91 (608.74)
Magnolia fraseri	6.1 (2.0)	0.1558 (0.0566)	18.61 (6.77)	982.56 (390.98)
Acer pensylvanicum	43.6 (12.5)	0.1326 (0.0471)	25.51 (8.48)	652.38 (289.79)
Tsuga canadensis	0.29 (0.29)	0.0637 (0.0637)	8.37 (8.38)	369.53 (369.53)
Acer saccharum	5.52 (1.96)	0.0516 (0.0230)	8.94 (3.76)	305.5 (152.3)
Hamamelis virginiana	29.07 (10.34)	0.0835 (0.0322)	9.7 (3.8)	224.98 (90.55)
Fraxinus americana	9.88 (5.54)	0.043 (0.022)	5.09 (2.61)	214.55 (139.98)
Sassafras albidum	15.12 (7.04)	0.0481 (0.0176)	5.61 (2.06)	159.84 (70.65)
Cornus florida	8.43 (2.62)	0.03 (0.01)	3.37 (1.07)	86 (30)
Castanea dentata	25 (7.0)	0.0328 (0.0090)	3.69 (1.02)	68.45 (19.63)
Symplocos tinctoria	5.52 (2.57)	0.0142 (0.0060)	1.61 (0.69)	38.46 (16.98)

Fagus grandifolia	0.29 (0.29)	0.0063 (0.0063)	0.75 (0.75)	30.24 (30.24)
Cornus alternifolia	0.87 (0.64)	0.0044 (0.0041)	0.5 (0.5)	14.67 (14.44)
Castanea pumila	0.87 (0.64)	0.0023 (0.0021)	0.29 (0.25)	7.02 (6.43)
Ostrya virginiana	0.29 (0.29)	0.0002 (0.0002)	0.03 (0.03)	0.35 (0.35)
Aesculus flava	0.29 (0.29)	0.0002 (0.0002)	0.02 (0.02)	0.27 (0.27)
Vitis spp.	29.94 (6.93)	0.0595 (0.0151)	0.15 (0.04)	0.16 (0.04)
Rhododendron calendulaceum	9.3 (4.2)	0.0079 (0.0035)	0.02 (0.01)	0.02 (0.01)
Pyrularia pubera	3.49 (1.40)	0.0035 (0.0014)	0.01 (0.00)	0.01 (0.00)
Clethra acuminata	2.62 (1.84)	0.003 (0.002)	0.01 (0.01)	0.01 (0.01)
Aristolochia macrophylla	2.33 (1.33)	0.0019 (0.0011)	0 (0)	0 (0)
Calycanthus floridus	2.03 (1.50)	0.0014 (0.0010)	0 (0)	0 (0)
Lindera benzoin	0.58 (0.58)	0.0005 (0.0005)	0 (0)	0 (0)
Parthenocissus quinquefolia	0.58 (0.41)	0.0005 (0.0003)	0 (0)	0 (0)
Vaccinium corymbosum	0.58 (0.41)	0.0005 (0.0003)	0 (0)	0 (0)
Vaccinium stamineum	0.58 (0.41)	0.0005 (0.0003)	0 (0)	0 (0)
Toxicodendron radicans	0.29 (0.29)	0.0002 (0.0002)	0 (0)	0 (0)

Ella Weckliff

Jake WARD

Katorına Bussell

Jordan Hessler

Andrea Stewart

Zach Martin

Marah Goodman

Whitlee Angel

Kane Blaze

POST CARD

Jan Sitterson

Zhexi Zeng

Lindsay Clough