EFFECTS OF PRESCRIBED FIRE, INVASIVE SPECIES, AND GEOGRAPHIC DISTRIBUTION PATTERNS ON GRANITE ROCK OUTCROP PLANT COMMUNITIES IN THE GEORGIA PIEDMONT

by

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(Under the Direction of JAMES AFFOLTER)

ABSTRACT

The granite rock outcrops of the southeastern United States are a unique assemblage of island-like habitats that support many endemic and rare species in their unique plant communities. The flora associated with the ecotone for many of these outcrops has shifted over time. Fire suppression, exotic plant invasion, and anthropogenic impacts are creating a dense understory environment of high competition and low light levels that discourage establishment and persistence of native outcrop flora. This research project includes two small-scale field studies and a broader landscape-level assessment of disturbance effects on species diversity and composition in the ecotone of rock outcrop environments.

Prescribed burns were applied to the ecotone plant community of two rock outcrops in the Piedmont of Georgia. Two other outcrop sites were chosen to assess community response to removal of invasives in the ecotone. Occurrence of vegetation classes and species richness were measured along random stratified belt transects. Hemispherical photographs, soil nutrients, and soil depth were used to monitor changes initiated by the prescribed burn. Neither fire, nor removal of invasives had a significant effect on vegetation class occurrence, including invasive and rare plant occurrence, or species richness. A site-specific evaluation, including the consideration for disturbance history and potential rare species' response, would be necessary to determine whether prescribed fire should be an appropriate management tool on rock outcrop ecotones.

Data for tree canopy, impervious surface, and landcover produced by the University of Georgia Natural Resources Spatial Analysis Laboratory were used to observe a 33-year snapshot of landcover change surrounding Georgia outcrops. Predictive models of species geographic distributions were used to determine the key environmental variables that determine local and regional species patterns. Landcover trends reveal increasing urban development and deforestation in all areas that were analyzed. Impervious surface surrounding rock outcrops significantly increased from 1991 to 2005. Rare plant occurrence increased with latitude and outcrop spatial complexity and did not appear to be tied to outcrop size or isolation. Geology, population density, and outcrop shape complexity were predictive of quarrying. These models may be applied to specific habitats to assist in identifying threats and informing conservation decisions.

INDEX WORDS: granite rock outcrop, ecotone, endemic species, Piedmont, invasion, prescribed fire, species diversity, rare plant, spatial distribution, plant community

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BS, Lehigh University, 2000

BA, Lehigh University, 2000

A Dissertation Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

DOCTOR OF PHILOSOPHY

ATHENS, GEORGIA

2011

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August 2011

DEDICATION

To my family who have inspired me to grow.

ACKNOWLEDGEMENTS

I would like to acknowledge my major professor, Dr. James Affolter, for his steadfast support and patience. I would also like to acknowledge the members of my committee, Dr. Ron Carroll, Dr. Dov Sax, Dr. Rebecca Sharitz and Dr. Robert Wyatt, for sharing their time and the wealth of their experience. I am grateful to Dr. Donald Shure of Emory University for his guidance and time on my advising committee. I am indebted to my colleagues in the Research and Conservation Lab, Heather Alley, Jennifer Ceska, and Linda Chafin, for their companionship and infectious love of plants. I am so grateful to the entire staff of the State Botanical Garden for making me feel like a part of their family. I would like to thank the State Botanical Garden Research and Conservation Volunteers for their weekly assistance, privet clearing, and years of friendship. I would like to express my gratitude to Kim Love-Myers, with the Statistical Consulting Center in the Department of Statistics at the University of Georgia, for her continual willingness to field my questions. I would like to thank James Rickard and Pete Pattavina at the U.S. Fish and Wildlife Service Georgia Ecological Services Field Office for their humor and commitment to support my love affair with rock outcrop plant communities. I would like to thank Malcolm Hodges and Erick Brown with The Nature Conservancy for their expertise in lighting fires and willingness to humor me. I would like to acknowledge Shan Cammack, Chris Canalos, Lisa Kruse, Dr. Mincy Moffet, and Tom Patrick with the Department of Natural Resources, Nongame Conservation Section, for their guidance in setting up this research. I want to thank James Allison for his advice in developing the project. I am grateful to Hugh and Carol Nourse for their willingness to teach me about outcrop flora and share their beautiful photographs. I am indebted to the Garden Club of America for the Catherine H. Beattie Fellowship, the Native Plant Society for the Marie Mellinger Grant and the Georgia Botanical Society for a Field Botany Grant. I want to thank Jennifer Bartmess for countless hours of fieldwork and companionship. Finally, I would like to express my most sincere appreciation to husband Mark, my brother Scott, and my sister Alye for taking care of Ethan and Sam so I could do field work.

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INTRODUCTION AND LITERATURE REVIEW

Introduction

Granite rock outcrops of the southeastern United States are an assemblage of bare rock exposures that support a highly specialized flora with a relatively large number of endemic species. These outcrops range from gently rolling flatrocks to inselbergs more than 200 meters in height and contrast starkly against a backdrop of oak-hickory forest (Shure 1999). High solar radiation and shallow to non-existent mineral soils combine to create high temperatures, low nutrient availability, and drought-like conditions which result in a harsh environment for plant survival. Islands of granite and granite-gneiss stretch throughout the Piedmont physiographic province, from Virginia to eastern Alabama. These exposures host unique species that have evolved a number of different physiological, morphological, and life history traits to withstand the extreme environmental temperature and moisture regimes present on the flatrock surface (Wyatt and Fowler, 1977; Wyatt, 1983; Quarterman, Burbanck, and Shure, 1993; Shure, 1999).

The center of the distribution for these granite rock outcrop exposures is located in the Piedmont of Georgia where the greatest concentration of endemic vascular flora also occurs (Murdy, 1968). The majority of granite outcrops lie within 100 km of Atlanta, a population center of approximately 5.5 million people and encompassing a metropolitan area of 35,000 hectares (U.S.C.B., 2009). The loss of populations, species, and entire ecological communities from human population centers and their surrounding landscapes is well documented and immense (Murphy, 1988). As a consequence of their isolated and restricted distributions, endemic species of this habitat type may be particularly vulnerable to extirpation and, ultimately, extinction. The integrity of many granite rock outcrops has been endangered by the quarrying of rock exposures for granite stone. Outcrops have also served as dumping grounds for trash, and playgrounds for humans and recreational traffic such as all-terrain vehicles and humans (James Allison, personal communication). With increasing pressure from urban influence, there is an immediate need to define and understand the threats to this unusual floristic community.

The present condition of granite rock outcrops in the southeastern Piedmont is not well understood or documented. Many of these rock exposures have not been mapped or surveyed in recent years. The growing urban center of Atlanta continues to impose environmental stresses on rock outcrop habitat through the physical impact of human traffic and increased erosion associated with urban development. As the demand for roads increases, quarrying of rock outcrop exposures is likely to increase. In the past, outcrops were avoided as development sites. As urban centers expand, some outcrops have been covered by fill dirt and used for residential or commercial development. Granite rock outcrops have significant moral, aesthetic, and economic value. For those who are interested in biological conservation, scientific research, or recreation and tourism, the irreversibility of species extinction demands that we assume responsibility for protecting these rare environments.

Rock Exposures in North America

Barrens, glades, and outcrops throughout North America, while all hosting their own specific flora, bear a striking resemblance to granite outcrops of the southeastern

United States because of similar abiotic characteristics. Congruent environments to outcrop habitats, with their bare rock and shallow soils, include alvars, granite barrens, serpentine barrens, sandstone outcrops, shale barrens, limestone or cedar glades, dolomitic glades, chert glades, and igneous glades (Table 1.1). These systems are well documented refuges for endemic species adapted to particular soil conditions and are often associated with relatively harsh abiotic conditions (Anderson, Fralish, and Baskin, 1999). Much of the early work on North American outcrop habitat is descriptive. Many early publications provide a species account for the subject plant community and assess how many of these species are unique to that particular environment. Past research has addressed fundamental ecological questions concerning successional processes, evolutionary trajectories toward edaphic or narrow endemism, and the role that spatial population dynamics play in driving species diversity.

Inselbergs, or isolated rocky outcroppings, are insular in nature and contrast markedly with the surrounding landscape. They appear to serve as refugia for many of their component species (Main, 1997). In North America, granite outcrops occur in three concentrated locations: western states (with an emphasis on California), the south-central states of Texas, Oklahoma, and Arkansas, and the southeastern states of Georgia, Alabama, North Carolina, South Carolina and Virginia (Wyatt and Allison, 2000). Previous research in plant community ecology has focused on primary succession on southeastern rock outcrops (Burbanck and Platt, 1964; Shure and Ragsdale, 1977; Burbanck and Phillips, 1983), but few studies have looked at threats to these exposures and little has been done in the last 25 years to assess the status of the remaining southeastern Piedmont exposures in a comprehensive way.

Rock Outcrops of the Georgia Piedmont

Granite rock outcrops occur in the southeastern Piedmont from east-central Alabama through Georgia and the Carolinas into south-central Virginia, covering an expanse 1,125 km long and 195 km wide (Emory, 1987; Godfrey, 1997). The Piedmont Plateau is composed of a shield of granite schists and gneisses where the soil layer is comprised of more readily weathered materials but more resistant granite rock lies exposed at the surface (Shure, 1999). McVaugh (1943) estimated that granite outcrops occupied approximately 3,235 hectares with 2,425 of those hectares occurring in the state of Georgia. Figure 1.1 shows the location of granite rock outcrops in the Georgia Piedmont using modern maping techniques. Nelson (1957) estimated that these "granitic lands" covered approximately 14% of the Piedmont physiographic province. These exposures may exist as flatrocks with little topographic relief or massive inselbergs, such as the famous monadnock of Stone Mountain. Individual outcrops may be isolated or have an almost continuous stepping stone-like spatial distribution. The thin soils that surround many of these rock exposures are typically dominated by loblolly pine and red cedar, often forming sinuous flows of glade-like forest and bare rock. The granite and gneiss rocks that characterize rock outcrops of this region are high in silica and aluminum and low in magnesium and iron, although individual rock units are typically heterogeneous in mineral composition and may be geologically dissimilar from one another (Watson, 1910; Wharton, 1978).

Formed by molten material that extruded into existing continental rock, the granite rock cooled below the surface. Through time, the softer surface rock eroded away leaving peaks of more resistant granite. It is difficult to estimate the amount of time a particular outcrop may have been exposed to the surface due to the cyclical nature of erosion. Piedmont granitic rocks are estimated to be 300-350 million years old (Watson, 1910; Whitney, Jones, and Walker, 1976; Wyatt and Allison, 2000). Wyatt (1983) estimated that some rock outcrops have been uncovered at least 150 million years before present. These exposed rocks are harsh environments that have been compared to desert islands in a sea of mesic oak-hickory forest. Temperatures at ground level can reach up to 50 - 55°C in the summer due to high incident radiation, heat absorption by the rock, and low evapotranspiration (Wyatt, 1997; Shure, 1999). Annual rainfall in the Georgia Piedmont averages 100 cm but 90-95 % of this water quickly drains off the rock surface where there is little soil or plant cover to hold it (Wiggs and Platt, 1962). Water loss due to runoff in other areas of the Piedmont averages a much smaller amount, approximately 10 % of total rainfall (Nabholz, 1978).

Individual plant species within the plant community of granite rock outcrops have evolved physiological mechanisms to deal with habitat severity. Adaptations to xeric environments include succulence and the presence of spines or silky hairs (Wyatt and Allison, 2000). Some plants associated with this environment are poikilohydric, meaning they have no biological mechanism to prevent drying out, but can survive cycles of dehydration and rehydration without losing viability (Porembski, Seine, and Barthlott, 1997). It has been suggested that many outcrop species, like *Portulaca smallii* and *Isoetes tegetiformans,* which employ C_4 and CAM photosynthetic pathways, respectively, use these physiological adaptations to survive the high temperatures and limited water availability of granite outcrop habitat (Emory, 1987; Wyatt, 1997). Nevertheless, Baskin and Baskin (1988) point out that most granite outcrop plants use the traditional C_3 pathway for carbon fixation.

Vascular flora of the granite outcrop systems present a different life form spectrum than that of the surrounding Piedmont flora (Wyatt and Allison, 2000). The harsh, strongly-seasonal environments of the outcrops favor a winter annual life cycle. Outcrop flora is dominated by therophytes (Phillips, 1982). Where drought and high summer temperatures would kill many plants, spring-flowering annuals germinate in the fall and overwinter as tiny rosettes on the soil surface. This life cycle typically produces small, wind dispersed seed which translates to good dispersal ability and high propagule numbers, but also results in high propagule mortality and greater turnover of local populations (Walters and Wyatt, 1982). Such rapid turnover could be a key to the evolution of floristic traits that enable survival of environmental extremes.

Plant community succession, especially in depression pools of insular outcrop habitat, has been one of the primary areas of focus for granite outcrop research in the Southeast. These island communities were first differentiated by Burbanck and Platt (1964) into *Diamorpha* (soil depth of 2-9 cm), Lichen-annual herb (soil depth of 7-15 cm), Annual-perennial herb (soil depth of 14-41 cm), and Herb-shrub (soil depth of 40-50 cm). Rogers (1971) later used the term shrub-tree community to characterize soil islands dominated by trees with a shrub understory. Whereas these shallow depressions are an integral part of the rock outcrop community, outcrop endemics may also become established within crevices and pools, along the margin, and to a lesser extent, on the bare rock itself.

Granite rock outcrops have long been considered ideal communities for studying primary plant succession. The classification system created by Burbank and Platt (1964) has been used to define seral stages, in which each stage, with its characteristic plant community and soil depth, represents a successional progression toward climax (Shure and Ragsdale, 1977). Constructive soil building effects of growth and succession in soil depressions are tempered by wind, rain, and fire. If successional change is occurring on rock outcrops, it is occurring at a very slow rate and demonstration of this successional change is difficult to prove on the time scales of human studies (Wyatt and Allison, 2000).

The earliest publications on rock outcrop habitats focused on describing outcrop flora. McVaugh (1943) has given the most thorough descriptive account of the granite rock outcrops throughout their range. Some of the plant communities and species of interest at Stone Mountain were described by Campbell (1921) and Allison (1994). A plant inventory of the floristic community of Echols Mill outcrop was done by Givens (1971). Panola Mountain was originally surveyed by Mathews (1941), with a more comprehensive list of vascular flora by Bostick (1971). Houlé (1987) performed a vascular flora inventory at Arabia Mountain.

A wealth of local-scale, field-based research was done on Georgia outcrops in the 1960s, 70s and 80s but little work of that type has taken place in recent years. These studies attempted to characterize a particular community or explore the ecological processes of plant succession. Several rock outcrops in the Atlanta area, including Panola Mountain, Arabia Mountain, Rock Chapel Mountain, and Mount Rollaway, have been the subject of previous research (Burbanck and Platt, 1964; Rogers, 1971; Shure and

Ragsdale, 1977; Phillips, 1981; Burbanck and Phillips, 1983). Due to the isolation of some of the rock exposures, the outcrops offered an opportunity for the study of plant communities in geographically separated but physically similar habitats (Burbanck and Platt, 1964).

Granite outcrop plant communities of the Southeast have been used to study a wide range of ecological processes. Researchers have used rock outcrop flora to study reproductive biology and germination ecology (Baskin and Baskin, 1972, 1977; Wyatt, 1981, 1983, 1984a, 1984b, 1984c, 1984d; Wyatt, 1986; Baskin, Baskin, and McCormick, 1987; Wyatt, 1990; Wyatt, Evans, and Sorenson, 1992; Tate, 2005). Studies have been performed to better understand species competition and nitrogen fixation (Herford, 1969; Sharitz and McCormick, 1973; Snyder and Wullstein, 1973; Ware, 1991). Radiation treatments have been applied to some outcrop species to ascertain their general effects on vascular plants (McCormick and Platt, 1962; Garrett, 1963; Helseth, 1965; Plummer and Helseth, 1965; Pittillo, 1966). The insular nature of these exposures makes them ideal for observing genetic diversity and the implications of metapopulation dynamics on population structure (Goldner, 1984; Godt and Hamrick, 1993). Several rock outcrop endemic species have been used to research allopolyploid and allopatric modes of speciation (Murdy, 1966, 1968; Matthews and Murdy, 1969; Chapman and Jones, 1971; Black and Murdy, 1972; Murdy and Carter, 1985; Levy, 1991; Van De Genachte, 1996b). Endemism as an ecological phenomenon has also been explored using the granite outcrop plant community (Cotter and Platt, 1959; Wyatt, 1977; Garris, 1980; Wyatt, 1981; Baskin and Baskin, 1988; Wilbur, 1988; Ware, 1990; Ware and Pinion, 1990; Ware, 1991). Understanding the mechanisms that encourage the evolution of restricted endemic species has been an area of interest for conservation ecologists attempting to protect and restore these populations (Loehle 2006).

Granite Outcrops as Centers of Endemism

Granite outcrops serve as reservoirs for biological diversity, where a significant component of the plant community is comprised of endemic flora. Endemic plants are defined as those with a small and specialized geographic range (Brown and Lomolino, 1998; Loehle, 2006). Wharton (1978) noticed that many endemic species that occur in the Piedmont province are encountered on granite outcrop habitat. According to Kruckeberg and Rabinowitz (1985), granite outcrop endemics are classified as narrow endemics which tend to belong to distinct communities of singular habitat. Wyatt and Allison (2000), identified eight strict endemics, 13 near-endemics (75% of reported occurrences in granite outcrops of SE), 13 half-endemics (the majority of reported occurrences in granite outcrops of SE) and 54 provincial endemics (95% of the known occurrences in the Piedmont are in outcrop habitat, but the majority of known occurrences are from outside of rock outcrop habitat) as part of the rock outcrop flora. The climate of these exposures is discontinuous compared to the surrounding region, stimulating endemism at the local site (Wyatt and Allison, 2000). The Piedmont also represents the juncture between the Coastal Plain, Ridge and Valley and Cumberland Plateau, and Blue Ridge physiographic provinces and therefore serves as a natural geographical distribution limit for many Coastal Plain and Appalachian Plateau plant species (Shure, 1999). Some of these species may find the habitat anomaly of the rock outcrops a welcome extension of their natural range.

Worldwide, inselbergs have been known to act as refugia for endemic species (Burke, 1998; Withers, 2000; Burke, 2002). Granite rock outcrops in the southeastern United States have been exposed since the Tertiary uplift and have served as a refugium for some species for a very long time (Oosting and Anderson, 1939; McVaugh, 1943). Observations about the narrow rock outcrop endemic *Amphianthus pusillus* led McVaugh (1943) to write, "the extreme restriction of this species to a single habitat, added to its high degree of vegetative and reproductive specialization, indicates that it has flourished in its present habitat for a very long time…with little outside change." Shure (1999) also posited that the large number of endemics affiliated with rock outcrop habitat indicated that these exposures had been inhabited by plants for a long time.

Raven (1964) theorized that a connection existed between edaphic endemism and catastrophic selection. According to his hypothesis, a species growing at the margins of its geographical range may grow in edaphic situations that are unusual for the species. It does not appear, however, that the endemic plant community of the granite outcrops of the southeastern United States is restricted by edaphic conditions. Baskin and Baskin (1988) observed that these rock outcrop endemics grow just as well in non-granitic soils, so it is unlikely that they are constrained by a chemical, physical, or biological element found there and nowhere else. Instead, the endemism of these communities appears to be tied to the lack of ability to compete in deeper soils and a requirement for high photon flux density (Griggs, 1940; Baskin and Baskin, 1988). Endemic and characteristic outcrop species have adapted to combinations of high light intensity, rapid and extreme fluctuations of moisture and temperature, low pH, and shallow sandy soils (McCormick and Platt, 1962; Lugo, 1969). Whether endemism on rock outcrops can be attributed to edaphic or environmental conditions, or competition, it is likely that these plant communities have experienced significant selective pressure over their long association with the rock outcrop habitat.

The island pattern of spatial distribution may foster plant speciation (Carlquist, 1974) and could be responsible for the high level of endemic species found in Southeastern rock outcrops. Isolation of some of the outcrops offers an opportunity for species to become locally adapted and undergo allopatric speciation (Murdy, 1966; Wyatt and Allison, 2000) or saltational speciation via allopolyploidy or other mechanisms (Murdy 1968; Wyatt 1997). Environmental patterns strongly influence ecological processes of population dynamics and community structure (Turner, 1993).

Many studies affirm that endemism, especially of a narrow or insular type, often leads to extinction (Wilson, 1961; Kruckeberg and Rabinowitz, 1985). Terborgh (1974) cites continental endemics and insular oceanic endemics as two categories of species that have featured prominently in the history of extinction. Regions rich in endemics dominate the global patterns of extinction (Pimm et al., 1995). Murdy (1968) notes "the disadvantage of specialization may be the probability that once a species has started down a path of restricted variability it will be unable to regain competitive ability under heterogeneous habitat conditions." Selfing can allow a population to fix local adaptations and persist in harsh conditions, but may also limit the genetic variation that would enable adjustment to drastic change or colonization of other habitats. Stebbins (1942) postulated that adaptation to narrow ecological conditions translates to lowered heterozygosity and therefore a lack of diversity capable of managing drastic change. However, while within-population variation is likely low, a high metapopulation number and degree of isolation can preserve diversity among populations (Drury, 1974). Several studies on *Arenaria uniflora*, *Isoetes tegetiformans*, and *I. melanospora* revealed the predicted trend of low within-population variation but very strong genetic differentiation between sites (Wyatt, Evans, and Sorenson, 1992; Van De Genachte, 1996). These patterns in population-level diversity demonstrate the need to consider spatial distribution patterns in managing threats to outcrop habitat. It is likely that the narrow endemics so inextricably bound to the rock outcrop habitat will not respond quickly to changes in their environment.

Threats to Rock Outcrops of the Georgia Piedmont

Although Georgia rock outcrops have received ample attention in the past as sites for the study of plant succession, little research has been done recently on these systems. No studies have specifically focused on the ecotones of these habitats or experimentally examined threats to species of these systems. In a survey of 52 outcrops from Alabama through Georgia and into South Carolina, Caspary and Rickard (2007) observed that 26 of the sites visited could be classified as disturbed or mostly disturbed, and 41 had some form of disturbance. These figures contrast with a survey of 28 rock outcrop sites performed by Magee (1987), who reported that 7 of the sites visited were disturbed or mostly disturbed and 18 exhibited some level of disturbance. Additional work is required to determine how the habitat that is left can be properly managed to ensure that the remaining diversity is left intact. According to Wilcove et al. (1998), there are four general threats that lead to species extinction: habitat destruction, over-exploitation, invasive species, and climate change. Quarrying has been the main cause of habitat destruction for outcrops worldwide (Twidale, 2000), although increasing population pressure appears to be driving destruction of habitat through development. Patrick

(1997) notes quarrying and off-road vehicles as the most important threats to rock outcrop habitats in the Piedmont of Georgia.

These systems are dominated by a number of different microhabitats: bare rock, cracks and crevices, vernal pools, soil depressions, seeps, and glade-like margins. It is likely that each microhabitat will exhibit a different sensitivity to outside threats. The bare rock is the most resilient of these microhabitats but also the least diverse. Acid rain and traffic from ATVs and foot traffic are the most significant threats to the lichen and moss community (Sigal and Johnston, 1986; Caspary and Rickard, 2007). Vernal pools and “sunnybell seeps” appear to be very sensitive environments, where cattle grazing, sedimentation, campfires, trash dumping and invasive exotics pose significant threats (Patrick, 1997; Caspary and Rickard, 2007). The marginal glade-like ecotone appears to be an important, but overlooked, gateway for threats to enter and alter the plant community of the rock outcrop habitat.

The Role of Invasion and Fire in Rock Outcrops

Invasion on outcrops increases competition for already scarce resources. According to the resource availability theory (Davis, Grime, and Thompson, 2000), invasion by exotic species strongly correlates with resource availability. Disturbance creates an opportunity for invasion by increasing resource availability and decreasing cover and vigor of surrounding competitors (Hobbs, 1989). Harrison et al. (2006b) set out to determine if communities with an especially rich flora of native, as well as key endemic and rare species, are particularly threatened by exotics, because exotic species have been observed to be more prevalent in areas where native species richness is highest. They performed their research in the California serpentine flora, characterized by rocky soils with low nutrient levels, and found that exotic species invasion was higher in sites with a greater richness of native species. Van Wilgen and Richardson (1985) point out that within the South African fynbos vegetation, alien shrubs increase the fuel load and lower the moisture content of live foliage, thereby increasing the frequency and intensity of fire.

Invasion by exotic flora has undoubtedly changed the plant species composition of the southeastern United States, resulting in loss of native species, changes in community structure and function, and alterations of the physical structure of plant communities (Meffe and Carroll, 1997). Exotics affect ecosystem function through competition for resources, prevention of recruitment, and altered fire regimes (Csurhes and Edwards, 1998). Weed invasions can alter vegetation structure and have been implicated in the extinction of threatened and endangered species (Leigh, 1992). No publications appear to address the effects of invasive species on the flora of granite rock outcrops of the southeastern United States.

Two exotic plants, *Ligustrum sinense* (Chinese privet) and *Lonicera japonica* (Japanese honeysuckle) are the most visible and dominant invasives observed within the marginal areas of the granite outcrops of the Southeast (personal observation). *Ligustrum sinense* typically forms dense thickets of shading shrubs and seems to be displacing the native *Forestiera ligustrina*. According to Baskin and Baskin (1998), competition and shading are the main limiting factors in outcrop vegetation. Thus, *L. sinense* may be changing the structure of the outcrop plant community from the margins inward by altering the available light levels in the forest understory. Transpiration by *L. sinense* may contribute to drying of shallow soils, and, in an environment that already has significant water stress, this competition for soil moisture could cause mortality of other plant species.

Threats posed by exotic plant invasions are intricately tied to our ability to manage these systems with controlled burns. Fire has also been a central force in shaping the forests of the Georgia Piedmont and burning has undoubtedly affected the structure and pattern of Piedmont vegetation (Jones, 1988). Consequently, fire management has long been prescribed for Piedmont plant communities (Whelan, 1995). There appears, however, to be a delicate balance in the use of fire to manage exotic invasions near outcrop habitat. A study performed by Hopper (2000a) in Western Australia suggests that outcrops, occurring as a natural fire break in the wild, provide a refuge for fire-sensitive vegetation. However, the outcrops of Western Australia act as catchments for water in the middle of highly flammable eucalypt scrub vegetation. The oak-hickory forests of the southeastern Piedmont are very different. They surround desert-like granite rock outcrop habitats in an envelope of mesic soils. Invasive species have also been documented to alter fire regimes (van Wilgen and Richardson, 1985; Brooks et al., 2004). Invading Chinese privet (*Ligustrum sinense*) has the potential to change the fuel loads and fire behavior along the rock perimeter of an outcrop.

Exploring the Ecotone of Granite Outcrop Systems

The term ecotone is used to describe “areas or zones of transition in the landscape from one habitat to another” (Risser, 1993). Other terms used synonymously include edge, transitional zone, margin, and boundary. In Western Australia, this microhabitat is commonly called the “apron” (Main, 1997). Historically, ecotones were assumed to support relatively high species diversity because they encompassed elements of both adjacent habitats as well as distinctive microhabitats found only in the ecotone (Odum, 1983; Delcourt and Delcourt, 1992). Ecotones may be a source of evolutionary novelty, play a critical role in maintaining diversity, and also be important areas regulating ecosystem function (Risser, 1993).

Ecotones can play a key role in altering ecological processes and community dynamics because they can restrict or facilitate the movement of organisms or propagules between areas along some sort of boundary. “Although boundaries occupy a relatively small portion of the total mosaic volume, they are important control points, modulating connecting flows” (Cadenasso et al., 2003). They may be sensitive areas for detecting changes in behavior and distribution of those species at the limit of their ecological tolerance (Risser, 1995). The ecotone concept also translates across all kinds and scales of ecological systems.

The ecotone of granite outcrops is a distinctive microhabitat where the soil depth and floristic composition is similar to the deeper soil islands on the outcrop surface, but increased shade and groundwater seepage make this habitat more mesic (Shure, 1999; Edwards, Kirkman, and Ambrose, 2012 (in press)). Many of the plant species that occur in this zone are mesophytes with no special adaptations to extreme habitat conditions, but because they are poor competitors, these plants find optimal areas for growth in a thin band around the margin of the rock island (McVaugh, 1943). Seepage from rocks upslope contributes to the formation of small bog-like areas in the ecotone that can be continuously wet through the winter and spring months but will desiccate after the typical drought-like conditions of mid-summer in the southeastern United States. The periodicity of high evapotranspiration rates during dry summers paired with shade from the encroaching canopy is believed to have prevented the establishment and spread of non-outcrop species in the ecotone (Murdy, 1968).

A rock outcrop ecotone is a microhabitat with its own distinct plant community (Wharton, 1978). Allison (1994) describes an ecotone forest flanking the northwestern side of Stone Mountain as "rich with rare or distinctive species that are not typical to rock outcrop bare exposures or to the oak-hickory forest of the Piedmont." The following is a description of the rock outcrop ecotone based on personal observations:

At the rock edge, rock outcrop endemics such as *Croton willdenowii*, *Diamorpha smallii*, *Helianthus porteri*, *Minuartia uniflora*, *Phacelia maculata*, *Phemeranthus teretifolius*, and *Sedum pusillum* thrive in beds of *Polytrichum*, *Sphagnum* and *Grimmia* moss amongst an abundance of *Cladonia* lichens. As you progress toward the forest, soils deepen and grasses such as *Andropogon virginicus*, *Chasmanthium sessiliflorum*, *Danthonia sericea*, and *Dichanthelium laxiflorum* create a glade-like cover that mixes with outcrop endemic herbs like *Anemone berlandieri, Tradescantia hirsuticaulis*, *Liatris microcephala*, and *Packera tomentosa*. Other old field weeds like *Krigia virginica*, *Nothoscordum bivalve*, *Nuttallanthus canadensis*, and *Triodanis perfoliata* also persist in the mid-zone of the typical outcrop ecotone. Vines of *Gelsemium sempervirens*, *Parthenocissus quinquefolia*, *Smilax bona-nox*, and *Vitis rotundifolia* creep in as soils deepen and help to close the canopy above. Shrubs such as *Callicarpa americana*, *Forestiera ligustrina*, and *Vaccinium arboreum* dominate the middle of the ecotone, with trees like *Pinus taeda* and *Juniperus virginiana* contributing to the canopy. The ecotone closes in with a canopy of *Carya* spp. and *Quercus* spp. This typical outcrop ecotone interior is spotted with occasional *Carex* spp. or *Asplenium platyneuron* fern, but remains mostly clear of a ground-layer herbaceous component.

The plant community of these marginal zones has historically been considered relatively stable (McVaugh 1943; Quarterman, Burbanck et al. 1993; Shure 1999), but this is the area that could be the most vulnerable to compromise by surrounding systems. It may therefore serve as a gateway to threats to interior outcrop microhabitats. Wharton (1978) documented disturbance in the ecotone in the form of historic cuts made for cotton field terraces or contours, trenches or gullies and non-native plants such as privet and honeysuckle. A review by the U.S. Fish and Wildlife Service (2007) cites quarrying and quarry dust, farm animals, vehicular traffic, dumping, and recreational impacts as threats to rock outcrop habitat. In the ecotone, invasion and fire suppression appear to play an important role in shaping plant community composition (Mawson, 2000).

Most of the studies performed on granite outcrops in the southeastern Piedmont have focused on succession in the plant communities of depression pools within the interior of rock outcrop habitats (Burbanck and Platt 1964; Shure and Ragsdale 1977; Burbanck and Phillips 1983). Little attention, however, has been paid to the changing composition of the ecotone community, where many endemic species are found. Endemic species can occur in small, isolated populations, which puts them at risk for extinction. The rock outcrop habitat can serve as a model system to explore whether the ecotone is essential to protect the integrity of the plant community as a whole. If viewed as a transitional zone, this ecotonal area has a distinctive set of physical conditions that supports an assembly of rare plants found nowhere else (Baskin and Baskin, 1988). Current changes in rock outcrop plant community composition may largely depend on changes occurring within the ecotone areas of these systems. The resource pulse from soil erosion and fertilizer drift from nearby development, combined with fire suppression, is likely to increase the susceptibility of these edge habitats to opportunistic invasions from exotic flora.

Research Objectives

Shure (1999) divided the rock outcrop habitat into four different microhabitats: 1) soil islands; 2) exposed rock; 3) temporary pools; and 4) outcrop margins. The first two communities have been studied extensively and the third supports few species. The outcrop margins have not been studied in great detail, even though this microhabitat appears to be the most sensitive to some of the threats to the plant community: namely, fire suppression and invasion by exotic plants.

The overall goals of this research project are to identify emerging threats to southeastern Piedmont granite rock outcrop plant communities and to clarify how these threats may be fundamentally changing the plant community composition, structure, and function of the rock outcrop habitat. Specifically, I looked at threats to these systems caused by anthropogenic disturbance, exotic species invasion, and fire suppression. The close urban proximity to the majority of granite rock outcrop habitats makes this distinctive plant community extremely vulnerable to disturbance pressures associated with human use and development. Invasion by a number of exotic plants is also threatening the ecotones of these habitats (between granitic and non-granitic soils) where many of the endangered species occur. Aided by fire suppression in the Piedmont, exotic species appear to be increasing. Using a combination of field surveys and regional geographic analysis, I evaluated the status of these systems and defined the threats that are instrumental in modifying their habitat.

To accomplish this work, I performed surveys of the vegetation along the transitional zone of rock outcrop habitat where prescribed burning is taking place and where active invasion is occurring. I used a Geographic Information Systems (GIS) analysis of rock outcrop habitat and associated flora to define spatial patterns in rare species distributions and to identify trends in habitat destruction.

Research Hypotheses

1. Fire suppression in the ecotone of rock outcrop systems encourages a dense understory layer where increased competition and lower light levels discourage establishment and maintenance of native rock outcrop flora.
2. Competition from invading exotic flora creates an environment where lower light levels and limited resource availability discourage the persistence of native outcrop flora.
3. Increasing regional development increases direct and indirect anthropogenic impacts, resulting in additional habitat loss, increased disturbance to remaining habitats, and increased pressure on rare endemic outcrop species.

A current assessment of granite rock outcrop habitat is needed to clarify the status of these systems, determine the health of their plant communities, and inform management decisions in conserved sites. Some of the most pristine remaining rock outcrop habitat is managed by The Nature Conservancy (TNC) and the State of Georgia, Department of Natural Resources (DNR), leaving land managers with the responsibility to properly care for these refuge communities. The vast majority of rock outcrop habitat acreage is held by private landowners. Understanding how depletion and destruction of habitat, fire suppression, and exotic plant invasions are restructuring and redefining outcrop habitat is critical to set conservation priorities and management protocols. By quantifying the degree to which granite rock outcrop habitat in the southeastern Piedmont is disappearing, findings from this research project can highlight hotspots of concern and determine if there are any spatial trends to this habitat destruction. This work will help to define the role of the marginal community in terms of its contributions to biotic diversity and to appraise the strength of influence from outside disturbance.

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Table 1.1: Granite Outcrops of North America

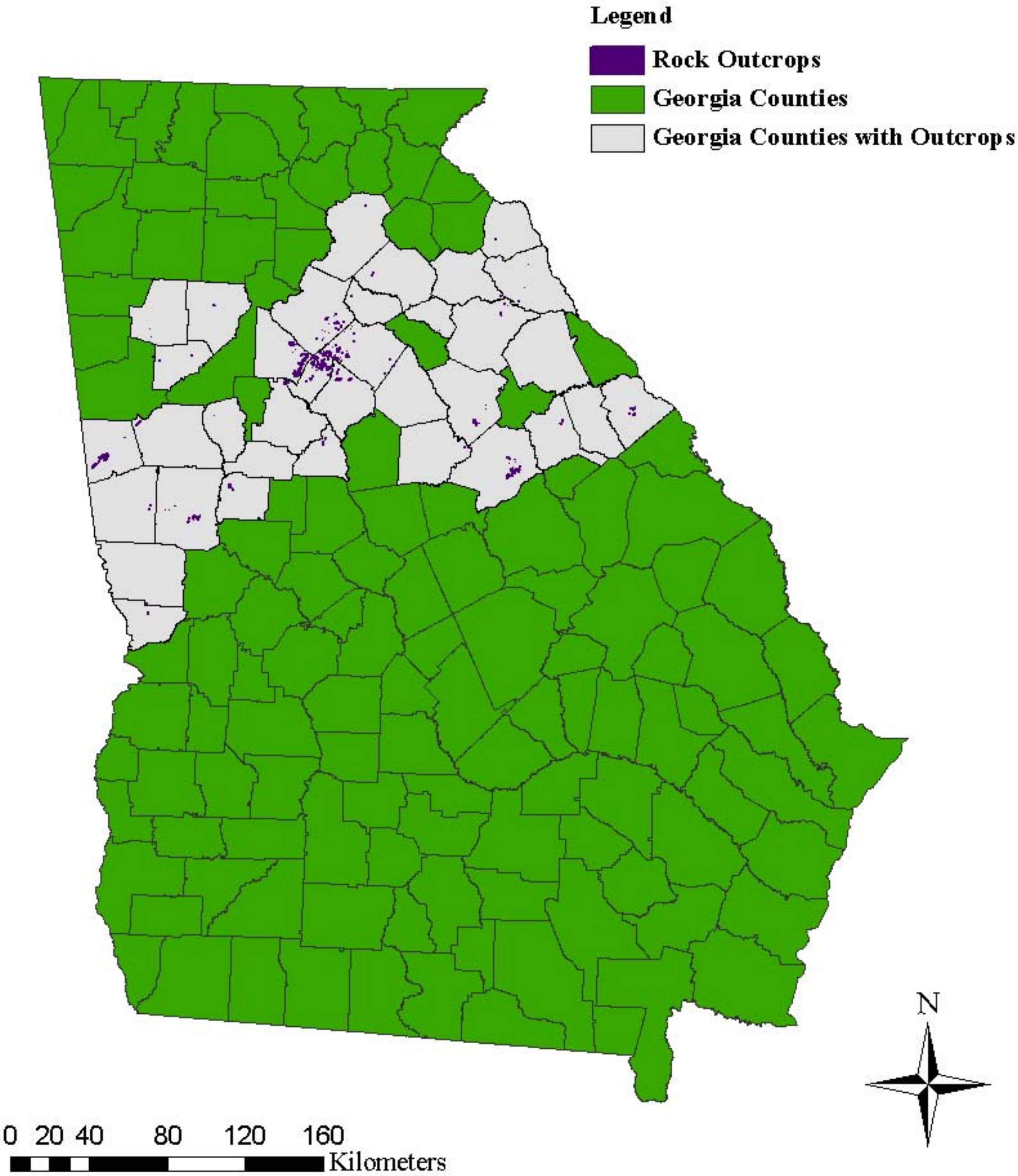
Habitat Type	Region	# of Endemics[a]	Study
Limestone and Dolomitic Glades	Southeastern US (AL, GA, KY, TN, VA, & WV)	21	Harper 1926
			Freeman 1933
			Harper 1939
			Quarterman 1950
			Baskin and Baskin 1985
			Bridges and Orzell 1986
			Baskin and Baskin 1999
	Ketona Glades, AL	+8	Allison and Stevens 2001
	Ozark region of AR and MO	na	Steyermark 1934
			Erickson et al. 1942
			Kucera and Martin 1957
			Hicks 1981
			Ware 2002
Sandstone Glades	Ozarks (AR and MO)	3	Palmer 1916
			Nelson and Ladd 1983
			Solecki 1983
			Jefferies 1987
	Cumberland Plateau in SE TN and NE AL	na	Harper 1906a
			Perkins 1981
	Coastal Plain of GA	na	Harper 1905
			Harper 1906b
			Harper 1906c
	Coastal Plain of W FL	na	Harper 1911
Chert Glades	Southwestern Missouri	na	Palmer 1910
			Palmer 1916
			Palmer and Steyermark 1935
Igneous Glades	Southeastern Missouri	na	Nelson and Ladd 1983
Alvars	Great Lakes Region (MI, NY and ON)	7	Catling and Brownell 1999a
Granite Barrens	Southern Ontario	0	Catling and Brownell 1999b
Granite Outcrops	Southern California	na	Rundel 1975
Rock Outcrops or Granite Barrens	Oklahoma, Wichita Mountains		Eskew 1938
			Diehl 1953
			Buck 1964
			Crockett 1964
Granite Outcrops	Texas (central mineral region)	6	Walters and Wyatt 1982

Table 1.1: Granite Outcrops of North America

Habitat Type	Region	# of Endemics[a]	Study
Granite Outcrops	Southeastern Piedmont U.S. (AL, GA, NC, SC and VA)	34	Oosting and Anderson 1937 Oosting and Anderson 1939 McVaugh 1943 Keever et al. 1951 Baker 1956 Burbanck and Platt 1964 Murdy 1968 Taggart 1973 Harvill 1976 Shure and Ragsdale 1977 Wyatt and Fowler 1977 Burbanck and Phillips 1983 Houlé 1987 Shure 1999 Wyatt and Allison 2000
Granite Outcrops	High Elevation Outcrops, NC	na	Wiser and White 1999
Serpentine Barrens	California	73	Whittaker 1954 Denton 1979 Kruckeberg 1984 Harrison 1997 Harrison 1999 Kruckeberg 1999 Harrison et al. 2000 Harrison et al. 2006a Harrison et al. 2006b
Serpentine Barrens	Washington, Oregon, and Northwestern CA	43	Kruckeberg 1999
Serpentine Barrens	Eastern North America (DE, MD, NC, NY, PA, and VT)	na	Harshberger 1904 Miller 1977 Barton and Wallenstein 1997 Tyndall and Hull 1999
Shale Barrens	Mid-Appalachian Mtns. (MD, PA, and VA)	19	Platt 1951 Keener 1970 Keener 1983 Braunschweig et al. 1999

[a] Species designated in articles as strict, near, half, or restricted endemics, na denotes endemic summaries that were not available in any available publications

Figure 1.1: Rock Outcrops of Georgia



Granite rock outcrop polygons suppied by the Natural Resources Spatial Analysis Labratory (NARSAL 2006).

CHAPTER 2

THE INFLUENCE OF FIRE ON TWO GRANITE ROCK OUTCROP ECOTONES IN THE PIEDMONT OF THE SOUTHEASTERN UNITED STATES

Abstract

The effect of fire was observed on the ecotonal plant community of two granite rock outcrops in the Piedmont of Georgia in the southeastern United States. Rock outcrop habitats in this region host a many endemic and rare plant species, but it is unknown whether these endemic species are sensitive to fire. The ecotones of these communities have not been extensively studied and the proportion of the edge community that is made up of species endemic to the rock outcrop habitat is not known. Rock outcrops, glades, and barrens typically host a very different community than the surrounding forest matrix, and it is important to understand how these systems will be affected by fire management. A burn treatment was applied in the winter of the first year of the study, after both study sites had been surveyed for one spring and fall season. Occurrence of vegetation classes and species richness were assessed along random stratified belt transects that began at the rock edge and traversed into the forest interior. Hemispherical photographs, soil nutrients, and soil depth were used to monitor changes initiated by the prescribed burn. Burning did not significantly alter soil nutrients or pH and fire had no apparent effect on the abundance of different vegetation classes. Fire did not significantly change rare flora or invasive species richness. Abundance of the threatened outcrop endemic, *Sedum pusillum*, did decrease after burning, but plants appeared to be recovering to pre-burn levels after the second year of the study. Most vegetation classes demonstrated spatial trends from the rock edge to the forest interior, but the invasive species class did not. Burning did not significantly increase canopy openness or noticeably alter total species richness or woody plant species richness in the two-year period after the prescribed burn. Prescribed fire has historically been used as an effective tool for managing forests in the Piedmont and does not appear to be a threat to the rare rock outcrop endemics, *Anemone berlandieri* and *Sedum pusillum*. Provided outside propagule pressure of exotic species is minimal, winter burns may be an appropriate management tool for rock outcrop ecotones, but should be evaluated on a site-specific basis.

Keywords: ecotone, endemic species, fire refuge, granite outcrop, management, prescribed burning, species richness

Introduction

Stretching from Alabama to Virginia within the Piedmont physiographic province, granite rock outcrops support specialized endemic and rare species that have evolved to withstand the temperature and moisture extremes that accompany life on the rock surface. The majority of granite outcrops lie within 100 km of the Atlanta urban area, which makes this distinctive plant community extremely vulnerable to disturbance pressures associated with human use and development. Much of this habitat has already been lost to quarrying activities, and future demand for crushed gravel used in highway road construction is expected to only make that figure grow (Wyatt and Allison, 2000; USFWS, 2007). Due to their isolated and restricted distributions, endemic species of granite outcrops may be particularly vulnerable to extirpation and, ultimately, extinction (Shure, 1999). Previous work has focused on what these communities can teach us about early plant succession and plant competition (Burbanck and Platt, 1964; Sharitz and McCormick, 1973; Shure and Ragsdale, 1977; Phillips, 1981; Burbanck and Phillips, 1983), but little research has been done in recent years to understand how disturbance may be restructuring the plant communities of rock outcrops. The majority of the rock exposures are privately owned and unprotected, but some of the most important Piedmont rock outcrops have been conserved. It remains unclear how these properties should be managed to maintain optimal community preservation.

Some of the most interesting and potentially valuable areas for conservation are the ecotones. This isn't surprising given the important role ecotones play in moderating biological communities. Ecotones often display high species diversity (Delcourt and Delcourt, 1992b; Risser, 1995). Smith et al. (1997) suggest that the ecotone as an independent ecological community can be a source for evolutionary novelty and may be an important contributor to maintaining biological diversity. The land adjacent to the rock exposure plays a key role in buffering disturbance and may be the critical link in blocking invasion and preserving the integrity of the plant community as a whole. Cadenasso and Pickett (2001) demonstrated that an edge with intact vegetation can serve as a barrier to seeds of colonizing invasive species. In the southeastern United States, little work has been done to explore the role of the ecotone between rock outcrops and upland forest in moderating disturbance or in shaping the plant community of the rock outcrop. A clear understanding of the ecotone community is vital for management of the rock outcrop ecosystem. To steward these communities properly and ensure that they serve their intended purpose as *in situ* conservation sites, we must determine if the ecotone community is rich in endemic, fire-sensitive species. Applying controlled burns to outcrop sites may create forest gaps that could further encourage establishment of invasive species.

Two growing threats to this system are exotic plant invasions and invasion and intrusion by native upland vegetation. Outcrop plant species that occupy the thinnest soils are known to have high light requirements and to be poor competitors (Sharitz and McCormick, 1973; Baskin and Baskin, 1988). *Ligustrum sinense* and *Lonicera japonica* have emerged as the most dominant exotic colonizers of rock outcrop margins and are known for crowding out native understory plants (Oosting and Livingstone, 1964; Wharton, 1978; Wyatt and Allison, 2000). Without the aid of fire, species richness may decline as early to mid-successional species are replaced by more shade tolerant competitors. Invasions have been known to change fuel properties which can in turn alter fire behavior and fire regime characteristics such as frequency, intensity, extent, type, and seasonality (van Wilgen and Richardson, 1985; Mawson, 2000; Brooks et al., 2004).

Prescribed fire is a useful tool for restoring plant communities that have endured years of fire suppression and invasion by exotic flora. Controlled burns have been used to reduce fuel loads, restore historical disturbance regimes, improve forage habitat, and promote biodiversity (Whelan, 1995). Nevertheless, it is unclear how rock outcrop endemic species will respond to burning. Terrestrial rock islands and barrens have been theorized to support a community of fire refuge species and are thought to serve as natural fire breaks in the wild (Hopper, 2000; Clarke, 2002). Most of the research that has been done on the relationship between fire and rock outcrop habitat has been performed on the outcrops of Western Australia (Mawson, 2000, Yates *et al.*, 2003) which act as catchments for water in the middle of highly flammable eucalypt scrub vegetation. The granite rock outcrop habitats of the Piedmont region of the southeastern United States is very different, occurring like desert islands in a sea of mesic vegetation. If rock outcrop endemics persist beyond the rock boundaries and are not fire tolerant, managing these communities with fire could unintentionally transform community composition. Altering or eliminating seed sources on the periphery of the outcrop could, in turn, change abundance or species richness in the interior of the rock outcrop. Fire events could also minimize seed dispersal and therefore impede colonization and immigration events from rock island to rock island.

Fire can be used to control invasive species but may also provide the disturbance required for establishment of invasives. Increasing fire intensity or frequency on the edge of a rock outcrop would create an unnatural fire regime, which could favor non-native species and impact ecosystem properties such as soil characters and nutrient cycling (Brooks et al., 2004; DiTomaso et al., 2006). Disturbance regimes are important controls for species richness and community composition and may free previously sequestered resources to be used by competitive opportunists (Grime, 1979; Huston, 1979; Pickett and White, 1985; Davis, Grime, and Thompson, 2000; Burke, 2003). Disturbance may also be important in maintaining species diversity in the ecotone. According to the intermediate disturbance hypothesis, intermediate levels of disturbance will promote species diversity by reducing competitive exclusion (Connell, 1978; Huston, 1979). A comparison of pre- and post-disturbance has been advocated as a means to observe the processes that guide immigration, extinction, and survivorship and ultimately, to understand the mechanisms that drive changes in species richness in a community (MacArthur and Wilson, 1967; Brown and Peet, 2003).

I studied the effects of fire on species richness and percent cover of native and invasive plant species, soil nutrients, and site openness by comparing burned and unburned transects in the ecotone of two granite outcrops. I wanted to determine how prescribed burning would change the canopy cover, soil nutrients, species richness and spatial patterns of distinct vegetation cover classes in the rock outcrop plant community, especially with respect to invasive and rare plant species. This burning treatment is expected to increase overall species richness and decrease abundance of invasive species. Moreover, fire is expected to negatively impact small herbs and endemic species like *Sedum pusillum* that persist in the shallowest of soils and are theorized to be fire refuge species. Lichens and mosses are expected to be negatively impacted by burning because they naturally occur in the shallowest soils near the rock edge and their biomass is exclusively aboveground. Previous studies have demonstrated that grass and herb response to burning is tied to fire intensity and season, but a winter burn is typically expected to increase grass and forb cover (Heisler, Briggs, and Knapp, 2003; Towne and Kemp, 2003). Woody plant and vine mortality closest to the rock is expected as a result of burning, but where soils are deeper, the burn is expected to trigger resprouting and stimulate a flush of new growth.

Methods

Study sites

Heggie’s Rock, a 40.9-hectare Nature Conservancy preserve, is located in Columbia County, Georgia (33° 32' 35” N, 82 15'13” W, ca. 5.6 km from Appling, Georgia). A tongue of glade and secondary oak-hickory forest on a section of the northwestern perimeter was burned on January 31, 2007. The burn spread at ground level with low intensity. Heggie's Rock is a monolith of porphyritic granite rising 21 m above the surrounding topography. The soils at the site are a combination of granite sands and Cecil sandy loam with a slope of 1-10% (U.S.D.A., 2008). Mean annual temperatures at the Appling weather station (located ca. 5.6 km west of Heggie's Rock) ranged from 24°C as a high and 10°C as a low for study-years 1 and 2 to 22°C as a high and 9°C as a low in the third year of the study. Annual rainfall recorded for the area (from the Appling weather station) was 1.024 m for year 1, 1.016 m for year 2, and 1.143 m for year 3.

Camp Meeting Rock, a 51.8-hectare Nature Conservancy preserve, is located in Heard County, Georgia (33° 14' 32" N, 85° 8' 49" W, ca. 5.6 km from Franklin, Georgia). In a 1.2-hectare section of oak-hickory forest on the southwestern perimeter of the site, a ground level fire of low to moderate intensity was applied on February 13, 2007. The exposed rock is a combination of biotite gneiss and granitic gneiss. Pacolet soils with slopes from 0-15 % characterize the marginal forest community (U.S.D.A., 2006). Mean annual temperatures at the Plant Wansley weather station (located ca. 2.1 km northeast of Camp Meeting Rock) ranged from 23-24°C as a high and 10°C as a low for all three years of the study. Annual precipitation from the Plant Wansley weather station ranged from 1.334 m in the first year, 0.899 m in the second year, and 1.135 m in the third year. It is estimated that both study areas have not experienced fire for at least 15-20 years.

Botanical survey

At each site, plants were surveyed in the spring and fall of 2006 (pre-burn, study-year 0), spring and fall of 2007 (post-burn, study-year 1) and spring and fall of 2008 (post-burn, study-year 2). Plants were surveyed across 18 belt transects, each comprised of 20 1x1-m contiguous plots. A random number was used to specify the position of the first transect along a compass heading parallel to the rock edge and subsequent transects were placed at 10 m intervals. These transects extended from the edge of the rock exposure into the surrounding forest. Plots 1, 10, and 20 were surveyed for vegetation class occurrence and species richness where plot one captured flora at the rock edge, plot ten was located in the middle of the ecotone, and plot 20 recorded flora at the end of the ecotone where the oak hickory habitat begins (Figure 2.1). Half (9 of 18) of the belt transects were burned between sampling seasons 0 and 1 in the winter of 2007 and the rest remained unmodified as control plots. Burn treatment transects were randomly assigned (Figures 2.2 & 2.3). The entire burn unit at both sites was approximately 1 ha in size, and fire was kept out of control plots using fire lines raked 2 m around the periphery of no-burn transects. In cases where the fire did not naturally carry all the way to the rock edge, transects were burned with a propane tank to ensure compete coverage.

Vegetation class occurrence was recorded for the plant categories of grasses, herbaceous plants, ferns and fern allies, invasive flora, lichens, mosses, rare flora, vines, and woody plants. The Georgia Exotic Pest Plant Council (2011) has classified invasives in the state of Georgia, according to their known degree of aggressiveness. Only plants listed in their highest level category for invasiveness (Category One) were included in the invasive percent cover category. Rare plants were analyzed in plots 1, 2, 3, 4, 5, 6, 8, 10, 12, 14, 16, 18, and 20 of each transect to increase the likelihood of destroying these uncommon species.

Soil nutrients and depth

Soil samples were taken in plots 1, 10, and 20 of each transect at each site. Soil was sampled at a depth of 0-5 cm, at the O horizon level, if present, and collected before and after the burn treatment. I chose to look at the macronutrient potassium (K), which is expected to volatilize with an intense burn, and the micronutrients calcium (Ca), magnesium (Mg) and sodium (Na) which are expected to become more biologically available after fire (Allan, 2006).

Canopy light

Hemphispherical photography, an indirect optical technique, was used to characterize canopy openness. Hemispherical photographs were taken in plots 1, 6, 10, 16, and 20 in each transect at each site using a Nikon F 35 mm digital camera and a Nikkor 8 mm f2.9 fisheye lens. The resulting image had a 183 degree view. Photographs were taken at ground level with the top of the image oriented north in fall of 2006 before the burn treatment and in fall of 2008 after the burn treatment.

Data Analysis

Frequency data for each vegetation class is reported by site, study-year, treatment, plot-distance, and study-year x treatment, where frequency is defined as the number of plots in which each vegetation class occurred. Vegetation occurrence was analyzed using logistic regression in a stepwise selection on the experimental variables of treatment, study-year, the interaction of treatment and study-year, and plot-distance. An example of the logistic regression model used is:

$$\ln(P/1\text{-}P) = \alpha + s_i + T_j + t_{k(i)} + Y_l + TY_{jl} + P_m + e_{ijklmn}$$

where P is the probability that Vegetation Class X=1 (occurs in plot) in observation *n* with all experimental factors as specified in the model: α is the intercept of the model, s_i is the random effect of site i (CMR or HR), T_j is the effect of treatment j (burn or no-burn), $t_{k(j)}$ is the effect of transect k within site j, Y_l is the effect of study-year l (0 or 2, where 0 is prior to the burn), TY_{jl} is the interaction of study-year and treatment (which is important because this interaction must exist in order to show that differences between burn/no-burn transects are actually due to the burn and not some other differences), P_m is the effect of plot-distance m (1 m, 10 m, or 20 m), and e_{ijklmn} is the random error of observation n with all experimental factors as specified. Once the important experimental factors were identified using this model, the additional continuous covariates of soil depth, site openness, pH, Ca, Mg, K, and Na were tested for significance in the model. The Tukey-Kramer adjustment was used to modify the p-value where multiple comparisons were made. Generalized Linear Models (GliM) were created for each cover class as a way of removing plot-distance from the models, and focusing on the relationship between plot cover and the continuous covariates of soil depth, canopy openness and soil nutrients. This was done because many of these continuous covariates were related to plot-distance, and may be, in combination, a more natural explanation for the response when plot-distance is not included.

Total species richness and woody species richness were measured during each survey. Shannon's diversity index was used as an additional measure of species richness for woody plants. A Poisson distribution was used to normalize the data. A stepwise multiple linear regression analysis was used to determine the relative influence of soil depth, canopy openness, Ca, Mg, Na, and K on total species richness and woody species richness. Tukey-Kramer adjustments were used to allow for multiple comparisons while maintaining an α value < 0.05. Generalized Linear Models were also used to analyze the relationship between total species richness and woody species richness and the continuous covariates of soil depth, canopy openness, and soil nutrients.

Soil nutrient variables Mg, K, Na, and Ca were transformed using natural logarithms because they demonstrated extremely right-skewed distributions. Atomic absorption was used to measure pH. In the data analysis, soil nutrient data were log transformed to help normalize data distribution. Soil depth was averaged from three independent measurements taken in plots 1, 6, 10, 16, and 20 in each transect at each site.

Soil depths were not normally distributed and were therefore transformed by taking the square root of the three independent observations for each plot, and then averaging over those observations to create the variable "square depth average (sqdepthav)". An Analysis of Variance (ANOVA) was used to calculate differences between sites and for plot-distance.

Gap Light Analyzer (GLA) software was used to compute the parameter percent site openness (Frazer, Canham, and Lertzman, 1999). Percent site openness is percentage of open sky beneath the forest canopy, with the influence of surrounding topography taken into account.

Results

Soil depth, soil nutrients, and light

Average soil depths did not vary significantly among transects at each site, but the total site average soil depth was significantly higher at Camp Meeting Rock (Table 2.1).

Individual plot average soil depths were significantly different, demonstrating a pattern of increased depth with increased distance from the rock edge.

Table 2.2 reports the amount of cations and pH in the soil before and after the burn treatment. Linear regression indicates that all five of the soil analysis variables, pH, Mg, K, Na, and Ca, appear to be similarly related to the variables of burn treatment, study-year, study-year x burn treatment interaction, and plot-location (Table 2.3). None of the soil nutrients or pH appear to have been significantly altered by the burn treatment. In the cases of pH, Mg, and Na, only plot-distance and year affected the responses significantly. All departures from the general trend are due to other differences between the years and are not a result of the burn treatment.

Analysis of the hemispherical photograph data through linear regression indicates that the burn treatment did not significantly change site openness (Table 2.2 and Table 2.3). The study-year and plot appear to have influenced the site openness response. Study-year 0 had significantly higher site openness than study-year 2, and plot 1 had significantly higher site openness than all other plots; but the burn treatments were not significantly different from control transects.

Vegetation Class

Frequency data for each vegetation class is reported by site, study-year, treatment, plot-distance, and study-year x treatment in Table 2.4. Vegetation class occurrence did not appear to be significantly impacted for any of the vegetation classes by the burn treatment (Table 2.5 and Table 2.6). The herb/fern/fern ally and grass class occurrences were significantly higher at Heggie's Rock, whereas vines were significantly greater at Camp Meeting Rock. The invasive class only occurred in Heggie's Rock transects and was the only vegetation class that did not demonstrate a correlation to plot-distance. Lichens, mosses, and herbs/ferns were recorded in much larger proportions close to the rock edge. Woody plants and vines were observed in greater numbers proportionally distant to the rock edge. The rare class was only recorded in plots at Heggie's Rock and most often occurred between plots 8 and 14. Occurrence of the rare plant class did not appear to change significantly year-to-year or with burn treatment.

Species richness

A total of 154 species at Heggie's Rock and 89 species at Camp Meeting Rock were recorded during the survey. This species total includes both vascular and non-vascular plants, and a list of plant species is provided in Appendix A. At Heggie's Rock, 21 endemic species were identified in the ecotone (13.6 % of total species richness), whereas at Camp Meeting Rock only five species (5.6 %) were identified as granite outcrop endemics as defined by Wyatt and Allison (2000).

The burn treatment did not appear to affect total species richness or woody plant species richness significantly, although total count numbers throughout the study decreased for both total species richness and woody species richness (Figures 2.4 and Figure 2.5**)**. Woody species richness increases proportionally to increased distance from the rock edge and study-year also appears to be significant in determining species richness (Table 2.6 and Table 2.7). The intercept model based on values calculated using Shannon's diversity index reinforce that study-year is significantly correlated with woody species richness (p-value = 0.0162). Plot-distance is extremely significant in determining total species richness (Table 2.7). Study-year is also highly significant, but there is no significant study-year x treatment interaction, therefore the burn treatment is not responsible for the difference. The burn treatment did not appear to significantly affect total species richness or woody plant species richness, although total count numbers throughout the study decreased for both total species richness and woody species richness.

Only two representative species occurred in the plots for the rare vegetation class (R), *Anemone berlandieri* and *Sedum pusillum*, and these only occurred at Heggie's Rock. Invasive plants were only recorded in transects at Heggie's Rock and consisted of the following four species: *Ligustrum sinense*, *Lonicera japonica*, *Murdannia keisak*, and *Rosa multiflora*. Species representation for invasive and rare vegetation classes was the same before and after the burn treatment.

Discussion

Burn treatments did not appear to negatively impact species richness or vegetation class occurrence in the ecotone community of the treatment sites. Fire does not carry well in thin soils, and controlled burning alone may not be the most effective tool for managing these systems. According to Graham et al. (2004), physical setting, weather, and fuels combine to determine wildfire intensity and severity. Both prescribed burns were administered as winter burns due to the amount of time these areas had been without fire and insufficient knowledge about how rock outcrop species might respond to burning. The Heggie's Rock burn did not carry well from the forest interior through to the rock edge and the behavior of that particular burn was patchy and of low intensity and severity. At the Camp Meeting Rock site, the fire burned with greater intensity and even carried through portions of transects A-F as a crown fire. Ecotone vegetation recovered relatively quickly and did not appear to sustain any significant changes to community assembly or structure.

Soil nutrients in a rock outcrop community are known to be limited, and a fire could spur a nutrient-level shift that might favor certain plant species. This treatment, however, did not significantly alter nutrient levels or change soil pH. Fire is expected to volatilize nitrogen and potassium but deposit nutrients such as calcium, sodium, and magnesium on the soil surface and make them more biologically available. Soil pH is expected to be higher following a burn and can affect the availability of certain nutrients to plants (Allan 2006). Potassium and pH, while not significantly different due to the burn, were significantly different in year 2 of the study but contrary to their projected direction. Low ground fires in forests burn at 200-300°C, so the winter burn preformed at both sites probably did not reach the temperatures in excess of 500°C required to volatilize K (Boerner, 1982; DiTomaso et al., 2006).

Hemispherical photographs indicated that canopy openness decreased after the burn. Site openness should increase dramatically after the burn due to physiological effects of the fire which causes mortality of herbs and saplings and dieback of woody plants. Woody trees and shrubs typically resprout heavily at ground level in response to burning. Since hemispherical photographs were taken at ground level, canopy openness should respond by decreasing once plant productivity increases. The change observed in canopy openness was not significant. This lack of response could be attributed to a need for more time to pass in order for the treatment to take effect or a need to repeat the treatment in order to observe the effects of burning. Hemispherical photographs could have also been taken at more intervals throughout the study to better measure vegetation response.

Outcrop ecotones are ideal communities to explore spatial patterns in vegetation classes because they host a wide range of grasses, lichens, mosses, herbaceous and woody plants, and vines. Spatial trends detected in vegetation classes throughout the study confirm that grasses, herbs, mosses, and lichens appear in greater abundance closer to the rock surface in shallower soils. As light levels decrease and soils become deeper, woody plants and vines become the successful occupants. Plot-distance from the rock edge appears to be a consistent predictor of vegetation class, which was expected given the trends identified in previous studies (Burbanck and Platt, 1964, Burbanck and Phillips, 1983, Shure and Ragsdale, 1977). This trend holds true when plot-distance is removed from the model, and soil depth remains consistently significant across vegetation classes. The fact that invasive species show no spatial trends with distance from the rock edge is noteworthy.

The state of Georgia experienced a severe drought from May 2007 through December 2007. Dry periods and drought are periodic events in rock outcrop communities, but this drought combined with the winter season burn likely killed a number of saplings in shallow soil plots. According to nearby weather station records, the total precipitation level for 2007 of 0.899 m at Camp Meeting Rock appears to be substantially lower both than for years 2006 (1.334 m total annual precipitation) and 2008 (1.334 m total annual precipitation) and for the corresponding total annual precipitation at Heggie's Rock (1.016 m). These extreme variations in precipitation may have confounded the outcome of the burn treatment. The burn and drought of 2007 likely initiated a reversion back from the tree-shrub to herb-shrub seral stages for many patches in the ecotone. A fire study performed on the Piedmont National Wildlife Refuge suggested that rainfall affected vegetative cover more than burn treatment (Rideout, Rickard, and Wade, 2003).

While there were concerns about rare, endemic species surviving the prescribed burn, they seemed to be surprisingly tolerant of fire. *Sedum pusillum*, a Georgia state-threatened outcrop endemic located in some transects at Heggie's Rock has succulent leaves and appeared to be a good example of a species using rock exposures as a refuge from fire. This species was expected to decline severely in response to burning, but observations after the fire confirmed that *Sedum pusillum* recovered from the burn without apparent issue. In terms of abundance, the number of plants declined considerably after the burn, but by study-year 2, the number of individual plants had increased to approximately half of the original number. *Anemone berlandieri* is a plant reported as special concern in the state of Georgia. Because this plant persists in the deeper mesic soils of the ecotone, it was not expected to experience significant mortality in response to burning. In fact, this species increased to five times the original number of stem counts in the spring after the burn, but plant abundance dropped down to just two times original amounts by the second year of the survey. Given that the rare flora responded with resilience to a winter treatment burn combined with severe drought conditions, it is possible that prescribed burning could be an effective management tool in areas of rare species occurrence along rock outcrop margins. Fire response should be evaluated on a species-specific basis to evaluate with certainty whether it is a safe and effective tool to use in these sensitive systems.

Controlled winter burns may be safe to use in the presence of rare endemic species, but the kind of prescribed burning required to impact exotic species may not be appropriate or effective in rock outcrop ecotones. Fire can be used to control *Lonicera japonica* and *Ligustrum sinense*, but effective fires must burn hot enough to prevent resprouting or plants will return to pre-burn levels within a few years (Oosting and Livingstone, 1964; Nuzzo, 1997; Batcher, 2000). In the southeastern United States, severe burns have been documented to increase weeds and decrease grasses (Rideout, Rickard, and Wade, 2003). DiTomaso et al. (2006) recommend an integrated strategy in which burn timing is critical and herbicide applications are also employed as a tool against invasion of exotic flora.

While many of the edges of rock outcrops have become substantially modified by invasion of exotic flora, both sites in this study are protected by The Nature Conservancy and have benefited from that active management. The opportunity to perform this research arose when prescribed winter burns were planned in pre-determined areas of the outcrop margins. Unfortunately, the lack of invasive species made it difficult to observe their response to fire. No Category One invasive species (as defined by GA-EPPC (2011)) were encountered during the surveys at Camp Meeting Rock. The surveyed area of the Heggie's Rock site was relatively uninvaded with four category one invasive species recorded; although portions of the rock ecotone were heavily impacted by *Lonicera japonica*. This invasive vine is particularly intrusive in the canopy layer, where it forms substantial cover and diminishes light availability for native understory plants. Exotic species occurrence outside the Category One designation was minimal to nonexistant at both sites. Burning did not appear to reduce the invasion cover class significantly.

There is considerable debate over whether rock outcrops should be managed with fire at all. Wyatt and Fowler (1977) suggest that mortality and turnover in the herb-shrub and shrub-tree communities on the rock outcrop surface is based on lack of moisture, rather than disturbance from erosion, wind, or fire. Water seems to run off the rock surface in fall, winter, and early spring, collecting in seeps along downstream edges and creating an effective buffer to discourage the spread of fire. Some of the glade-like ecotone areas, however, are essentially grasslands that have a propensity to carry and tolerate fire well. The strongly seasonal aspects of outcrop plant communities support winter annuals which germinate in the fall and overwinter as tiny rosettes on the soil surface (Burbanck and Platt, 1964). It is unclear how those fragile rosettes persist after a winter burn of low to moderate intensity. Several studies on Australian rock outcrops suggest that frequently burned outcrops are likely to become invaded by fire-tolerant species (Porembski, Seine, and Barthlott, 1997; Hopper, 2000; Pigott, 2000). Mawson (2000) found that immediately after fire, factors such as nutrient-rich ash, soil enriched by smoke, full sun at ground level, and removal of long-lived woody dominants favored fast growing, opportunistic herbaceous species. In a study performed by Hester and Hobbs (1992), however, fire did not necessarily increase invasion and was recommended as an effective management tool if care was taken to minimize other disturbances. Repeated burning of moderate intensity would be required to decrease *Ligustrum sinense* (Batcher, 2000), but this burn regime may cause mortality to endemic *Forestiera ligustrina*, a sister oleaceous shrub that occupies the niche *Ligustrum sinense* is invading.

While analysis of the data demonstrated that the burn treatment did not significantly affect total species richness and woody species richness, species richness did decrease throughout the study period. Change in species richness can be an important indicator of community health. This change could indicate that the granite outcrops do harbor fire-sensitive species along their margins and that fire is not an appropriate tool for managing these systems. It is also possible that the study was not long enough either to detect long-term trends in plant community shifts or to catch the recovery from the prescribed burn, especially when confounded by anomalous climate conditions.

Increased species diversity is believed to improve an ecosystem's resilience and resistance to environmental change (Chapin et al., 2000), but ecotones of rock outcrop systems in the Piedmont of Georgia may be more stable with a less diverse, but better adjusted community. According to Odum (1985), communities subjected to continual stress are expected to exhibit decreased species diversity. Much of the mortality that was witnessed throughout the survey from fire and drought were seedlings and saplings of species that do not necessarily belong on dry upland slopes. *Acer rubrum*, *Ulmus alata*, *Celtis occidentalis*, and *Ligustrum sinense* are all plants that experience high turnover and capitalize on abundant, effective seed dispersal to colonize new areas. These plants are not physiologically adapted to the rock outcrop environment and extreme physical or climatic disturbance may clear the way and make resources available for a more specialized flora. It is likely that the ecotones that encompass the area between two unique plant communities serve as a gateway for disturbance and invasion and will elicit their own unique dynamic in response to environmental change.

The three-year period of time for this study is only enough to gain a snapshot into this complicated community. Rock outcrop plant communities have been evolving over millions of years to respond to the severity of physical disturbance by fire and stress from dry climate conditions. Although the product of invasion is evident in many rock outcrop ecotones, I was unable to demonstrate and document that change during the timeframe of this study. It appears that, while fire may not be integral to the management needs of these systems, there was no evidence in this study to suggest prescribed burning is doing any harm either by stimulating an increase in the invasive community or by causing mortality of rare species.

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Table 2.1: Fire Study Average Soil Depth By Plot

Site	**Function**	**Plot 1** (cm)	**Plot 6** (cm)	**Plot 10** (cm)	**Plot 16** (cm)	**Plot 20** (cm)	**Total**
Camp Meeting Rock	Mean	2.6	25.8	35.8	43.4	56.5	31.6
	S.D.	2.4	15.6	17.6	22.1	28.2	30.7
	Max	6.7	53.3	71.7	76.7	100.0	100.0
	Min	0.3	1.7	10.0	8.3	3.3	0.0
Heggie's Rock	Mean	4.0	13.4	20.5	25.6	21.0	15.3
	S.D.	2.8	8.5	10.7	13.6	8.4	11.3
	Max	9.7	30.7	41.3	63.0	35.3	44.0
	Min	0.3	0.0	3.7	7.3	9.7	0.0
T-test soil depth and site							
P-Value							**0.0004**
T-test soil depth and treatment transects							
P-Value							0.998
ANOVA soil depth and plot							
P-Value							**<.0001**

Table 2.2: Fire Study Soil Nutrients, pH, and Percent Site Openness

Site	Transect Type/ Function	% Ca		% Mg		% K		% Na		pH		% Site Openness	
		Year 0	Year 2	Year 0	Year 2	Year 0	Year 2	Year 0	Year 2	Year 0	Year 2	Year 0	Year 2
Camp Meeting Rock	No-Burn Trans.												
	Mean	0.0029	0.0031	0.0015	0.0014	0.0259	0.0260	0.0066	0.0050	5.08	4.62	38.78	24.81
	S.D.	0.0096	0.0096	0.0004	0.0006	0.0051	0.0067	0.0027	0.0024	0.49	0.51	6.36	6.14
	Max	0.0499	0.0473	0.0023	0.0029	0.0437	0.0429	0.0132	0.0122	6.02	6.04	49.47	39.01
	Min	0.0000	0.0000	0.0007	0.0003	0.0181	0.0156	0.0026	0.0019	4.06	3.85	28.48	16.54
Camp Meeting Rock	Burn Trans.												
	Mean	0.0019	0.0023	0.0012	0.0013	0.0245	0.0283	0.0056	0.0051	4.93	4.69	40.04	25.01
	S.D.	0.0055	0.0069	0.0005	0.0007	0.0074	0.0119	0.0032	0.0018	0.62	0.50	5.73	5.51
	Max	0.0271	0.0326	0.0026	0.0025	0.0483	0.0570	0.0161	0.0082	6.30	6.73	49.10	42.79
	Min	0.0000	0.0000	0.0003	0.0002	0.0148	0.0099	0.0024	0.0019	3.88	4.17	26.31	14.94
Heggie's Rock	No-Burn Trans.												
	Mean	0.1858	0.1009	0.0007	0.0015	0.0230	0.0254	0.0036	0.0037	5.68	5.48	40.60	41.62
	S.D.	0.1678	0.1457	0.0004	0.0006	0.0054	0.0058	0.0015	0.0020	0.57	0.44	9.50	6.89
	Max	0.4270	0.4469	0.0018	0.0027	0.0316	0.0417	0.0062	0.0087	6.72	6.70	58.50	57.06
	Min	0.0001	0.0001	0.0004	0.0006	0.0127	0.0174	0.0015	0.0011	4.28	4.84	22.89	29.38
Heggie's Rock	Burn Trans.												
	Mean	0.1936	0.1065	0.0006	0.0015	0.0264	0.0275	0.0037	0.0034	5.65	5.72	36.67	42.53
	S.D.	0.1762	0.1530	0.0003	0.0004	0.0048	0.0072	0.0024	0.0022	0.61	0.62	13.35	9.93
	Max	0.4116	0.4832	0.0018	0.0025	0.0375	0.0418	0.0101	0.0098	7.11	7.06	71.55	63.22
	Min	0.0010	0.0002	0.0004	0.0006	0.0173	0.0161	0.0014	0.0008	4.59	4.62	20.31	22.06

Table 2.3: Fire Study Soil Nutrient, pH, and Site Openness Model Results

Effect	Study-year	Plot	Estimate	Standard Error	P-Value
pH (Linear Mixed Model)					
Intercept			5.4019	0.4025	0.0473
Plot		1/10	-0.1037	0.08289	0.4246
Plot		1/20	-0.4733	0.08289	**<.0001**
Plot		10/20	-0.3696	0.08258	**<.0001**
Study-year	0/2		0.2154	0.0676	**0.0017**
lnCa (Linear Mixed Model)					
Intercept			-2.6324	2.1814	0.4405
Plot		1/10	-1.8152	0.4607	**0.0003**
Plot		1/20	-2.9515	0.5166	**<.0001**
Plot		10/20	-1.1363	0.3203	**0.0014**
Study-year	0/2		0.7043	0.2567	**0.0067**
lnMg (Linear Mixed Model)					
Intercept			-6.8333	0.1373	0.0128
Plot		1/10	0.2137	0.07755	**0.0177**
Plot		1/20	0.3491	0.07755	**<.0001**
Plot		10/20	0.1354	0.07698	0.1865
Study-year	0/2		-0.3516	0.06316	**<.0001**
K (Linear Mixed Model)					
Intercept			-3.4429	0.04718	0.0087
Study-year	0/2		-0.06447	0.03107	**0.0395**
lnNa (Linear Mixed Model)					
Intercept			-5.8172	0.2429	0.0266
Plot		1/10	0.4401	0.05907	**<.0001**
Plot		1/20	0.5714	0.05907	**<.0001**
Plot		10/20	0.1312	0.05859	0.0675
Study-year	0/2		0.1033	0.0481	**0.0331**
Site Openness (Linear Mixed Model)					
Intercept			33.5178	2.9496	0.0559
Plot	1		6.4734	1.4549	**<.0001**
Plot	6		-2.2135	1.4497	0.1278
Plot	10		-2.8995	1.4497	**0.0464**
Plot	16		-1.0484	1.4497	0.4701
Plot	20		0	.	**.**
Study-year	0/2		4.1587	0.9162	**<.0001**

Table 2.4: Fire Study Frequency of Vegetation Classes

Test	Vegetation Class % Representation[a]							
All Plots	**G**	**HF**	**I**	**L**	**M**	**R[b]**	**V**	**W**
HR	74.07	49.07	30.56	12.96	45.37	12.04	49.07	62.96
CMR	48.15	12.04	0	8.33	53.7	0	83.33	61.11
Total	61.11	30.56	15.28	10.65	49.34	12.04	66.2	62.04
By Treatment								
Burned	61.11	25	12.04	11.11	47.22	9.12	66.67	63.89
Not-Burned	61.11	36.11	18.52	10.19	51.85	9.33	65.74	60.19
By Plot								
1	45.83	41.67	8.33	29.17	77.78	8.33	27.78	18.06
10	68.06	33.33	19.44	2.78	47.22	12.04	79.17	75
20	69.44	16.67	18.06	0	23.61	0	91.67	93.06
By Study-year								
Year 0	60.19	31.48	18.52	11.11	48.15	8.97	71.3	67.59
Year 2	62.04	29.63	12.04	10.19	50.93	9.08	61.11	56.48
By Study-year and Treatment								
Not Burned x Year 0	50.8	58.8	55.0	41.7	53.9	29.2	50.7	50.7
Not Burned x Year 2	49.3	59.4	69.2	54.6	50.9	42.1	48.5	45.9
Burned x Year 0	49.2	41.2	45.0	58.3	46.2	70.8	49.4	49.3
Burned x Year 2	50.8	40.6	30.8	45.5	49.1	57.9	51.5	54.1

[a] Vegetation classes are G for grasses, HF for herbaceous plants, ferns, and fern allies, I for invasive, L for lichens, M for mosses, R for rare flora, V for vines, and W for woody plants.

[b] NA designates there were not enough occurrences recorded to perform an

Table 2.5: Logistic Regression for Vegetation Classes with Tukey-Kramer Adjustment

Effect	**Plot**	**Estimate**	**Standard Error**	**P-Value**
Grass Class				
Intercept		5.516	2.7886	0.298
Plot	1/10	-1.2344	0.4019	**0.007**
Plot	1/20	-1.5075	0.4193	**0.0012**
Plot	10/20	-0.273	0.3891	0.7628
lnMg		0.7845	0.3433	**0.0235**
lnCa		-0.1721	0.07936	**0.0314**
Herb/Fern/Fern Ally Class				
Intercept		-9.9183	3.1866	0.1979
Plot	1/10	0.7907	0.4317	0.1623
Plot	1/20	1.8801	0.4787	**0.0004**
Plot	10/20	1.0894	0.4463	**0.0412**
lnK		-2.1214	0.7928	**0.0082**
Invasive Class				
Intercept		-2.9363	0.5559	<.0001
Study-year 0		0.7227	0.1689	**<.0001**
Study-year 2		0	.	.
Soil Depth		0.3572	0.08406	**<.0001**
Plot	5	1.6114	0.3677	**<.0001**
Lichen Class				
Intercept		-4.1209	0.8042	0.1227
Plot	1/10	3.0656	0.8075	**0.0002**
Moss Class				
Intercept		-2.11	0.4969	0.1472
Plot	1/10	1.5261	0.3901	**0.0001**
Plot	1/20	2.4827	0.4175	**<.0001**
Plot	10/20	0.9566	0.3797	**0.0127**
lnCa		-0.169	0.06104	**0.0062**
Rare Class				
Intercept		-0.1309	1.2251	0.9153
Soil Depth		-1.4434	0.159	**<.0001**
Plot		4.5689	1.1205	**<.0001**
Vine Class				
Intercept		3.0834	1.8956	0.3509
Study-year 0		0.9487	0.4266	**0.0274**
Study-year 2		0	.	.
Plot	1/10	-3.9198	0.7312	**<.0001**
Plot	1/20	-5.2452	0.8214	**<.0001**
Plot	10/20	-1.3254	0.5737	0.057
Woody Plant Class				
Intercept		5.6187	3.9292	0.3885
Study-year 0		1.3646	0.4555	**0.0031**
Study-year 2		0	.	.
Plot	1/10	-3.2233	0.5644	**<.0001**
Plot	1/20	-4.8907	0.7455	**<.0001**
Plot	10/20	-1.6674	0.5985	**0.0163**
lnK	.	2.8237	0.8953	**0.0019**
lnNa		-1.3054	0.501	**0.01**

Table 2.6: Fire Study Generalized Linear Models for Vegetation Classes and Species Richness

	Max. Likely. Est.	Standard Error	P-value
Grass Class			
Intercept	-0.7185	2.0509	0.7261
pH	0.9526	0.242	**<.0001**
Sqdepthav	0.1619	0.0633	**0.0105**
lnMg	0.6454	0.2963	**0.0294**
Herb/Fern/Fern Ally Class			
Intercept	-12.3095	2.7666	<.0001
Site openness	0.0325	0.0174	0.062
lnK	-3.3039	0.8042	**<.0001**
Sqdepthav	-0.3159	0.0931	**0.0007**
lnCa	0.1356	0.055	**0.0136**
Lichen Class			
Intercept	-0.1914	0.3934	0.6265
Sqdepthav	-0.6799	0.1583	**<.0001**
Moss Class			
Intercept	0.2355	0.3658	0.5197
Sqdepthav	-0.393	0.0715	**<.0001**
lnCa	-0.2383	0.052	**<.0001**
Invasive Class			
Intercept	-10.5825	3.6386	0.0036
lnK	-2.631	0.9705	**0.0067**
Vine Class			
Intercept	4.3503	2.5805	0.0918
Sqdepthav	0.8473	0.1327	**<.0001**
lnCa	-0.2072	0.0635	**0.0011**
lnK	2.0949	0.7766	**0.007**
Woody Class			
Intercept	-0.48564	1.04841	0.644
lnMg	-0.39373	0.15073	**0.01**
W Sp Richness			
Intercept	-4.9803	0.8596	<.0001
Sqdepthav	0.1585	0.0241	**<.0001**
lnMg	-0.2742	0.1092	**0.012**
lnNa	-0.4746	0.1155	**<.0001**
Total Sp Richness			
Intercept	-0.3665	0.327	0.2625
lnNa	-0.2021	0.0607	**0.0009**
pH	0.1522	0.0464	**0.001**
Sqdepthav	0.0249	0.0128	0.0515

Table 2.7: Fire Study Logistic Regression for Species Richness with Tukey-Kramer Adjustment

Effect	Plot	Estimate	Standard Error	P-Value
Woody SR				
Intercept		0.5615	0.1181	0.132
Study-year 0		0.4027	0.1174	**0.0008**
Study-year 2		0	.	.
Plot	1/10	-2.1282	0.2731	**<.0001**
Plot	1/20	-2.3734	0.27	**<.0001**
Plot	10/20	-0.2451	0.1189	0.1011
Total SR				
Intercept		1.4292	0.2453	0.1082
Study-year 0		0.2027	0.06128	**0.0011**
Study-year 2		0	.	.
Plot	1/10	-0.4002	0.07627	**<.0001**
Plot	1/20	-0.3813	0.07928	**<.0001**
Plot	10/20	0.01885	0.07043	0.9613
lnCa		-0.03806	0.0148	**0.011**

Figure 2.1: Fire Study Transect Design Layout

🟩 1
2
3
4
5
🟪 6
7
8
9
🟩 10
11
12
13
14
15
🟪 16
17
18
19
🟩 20

🟩 **Variables Used for Statistical Analysis:**
Vegetation Classes: Grasses, Herbs, Ferns, and Fern Allies, Invasives, Lichens, Mosses, Rare Plants, Vines, and Woody Plants
Total Species Richness and Woody Species Richness
Environmental Data:
a) Soil Nutrients & pH
b) Soil Depth
c) Hemispherical Photograph

🟪 **Variables Used for Statistical Analysis:**
Vegetation Classes: Grasses, Herbs, Ferns, and Fern Allies, Invasives, Lichens, Mosses, Rare Plants, Vines, and Woody Plants
Environmental Data:
a) Soil Depth
b) Hemispherical Photograph

Figure 2.2: Camp Meeting Rock Transect Locations

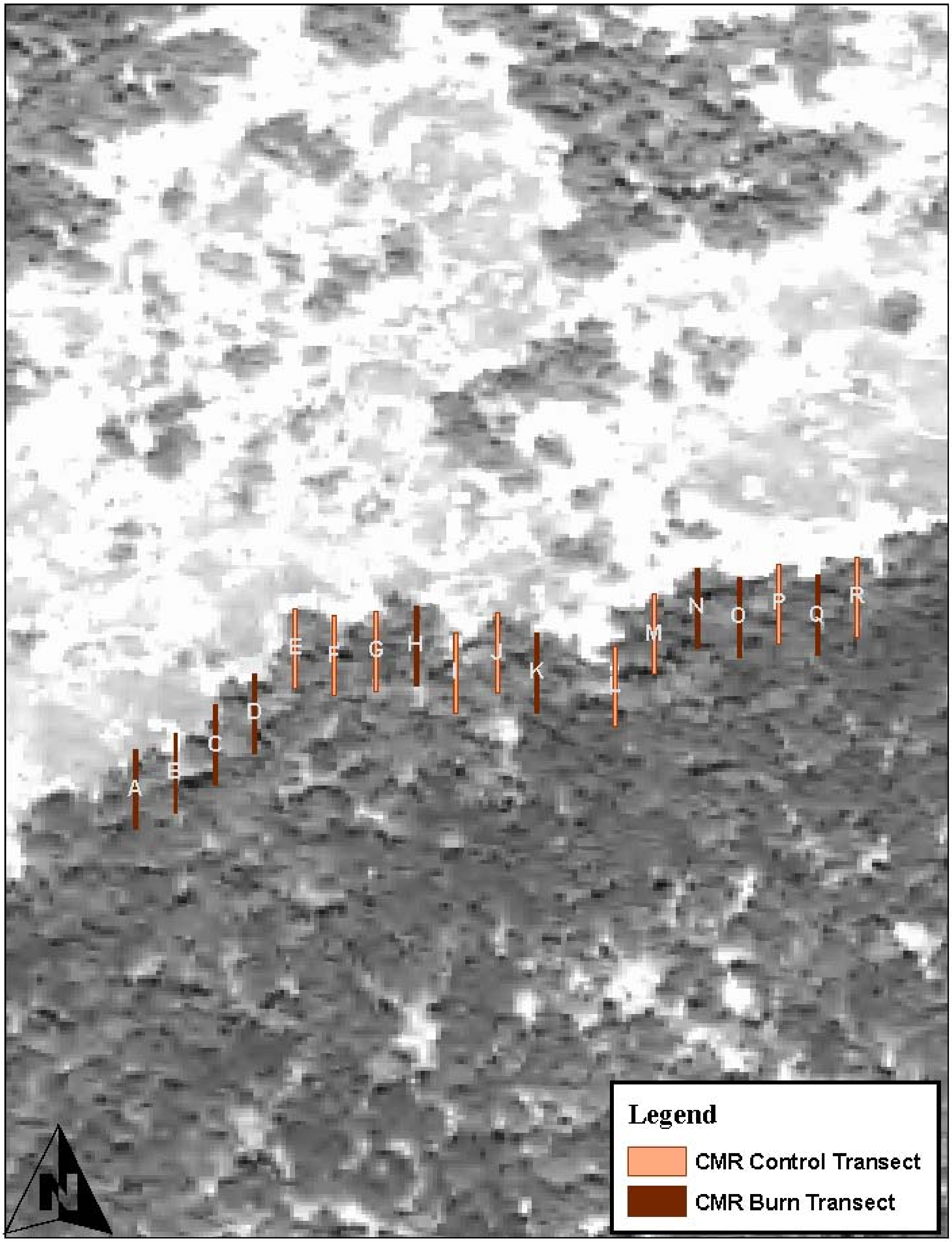

A
B
C
D
E
F
G
H
I
J
K
L
M
N
O
P
Q
R
Legend
CMR Control Transect
CMR Burn Transect
N

Figure 2.3: Heggie's Rock Transect Locations

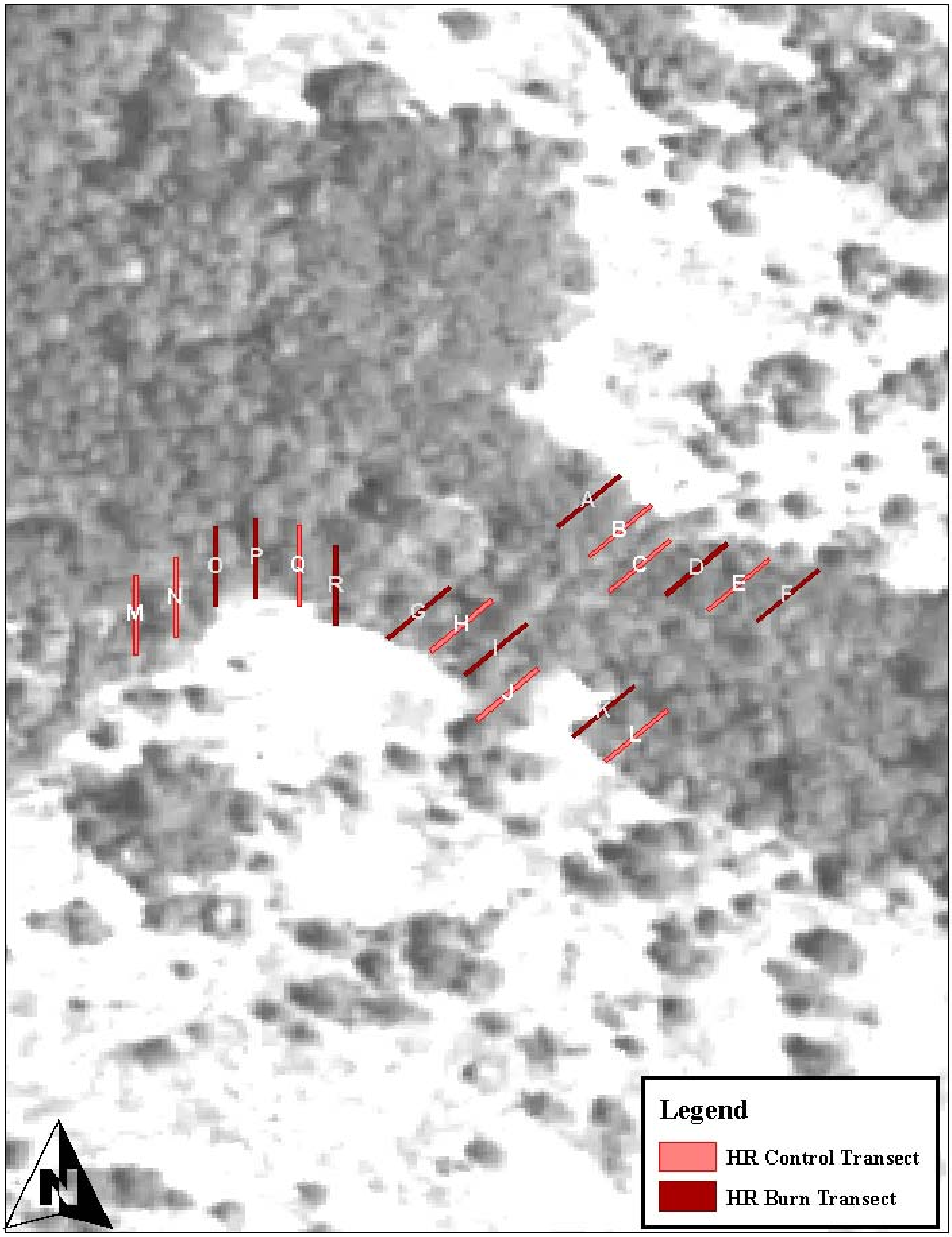
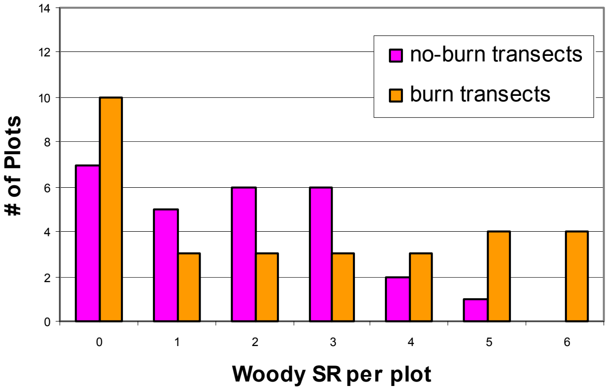
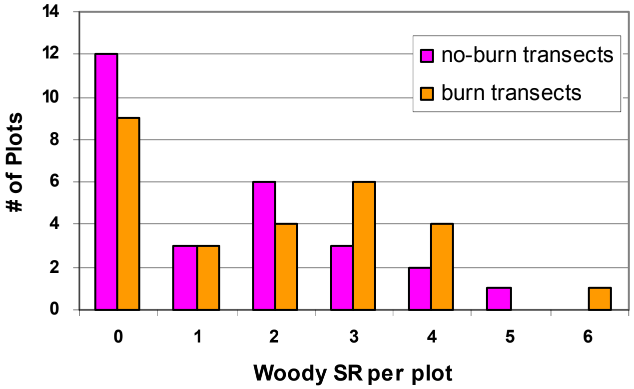

A
B
C
D
E
F
G
H
I
J
K
L
M
N
O
P
Q
R
Legend
HR Control Transect
HR Burn Transect
N

Figure 2.4: Woody Plant Species Richness for Camp Meeting Rock and Heggie's Rock, Before and After Burn Treatments

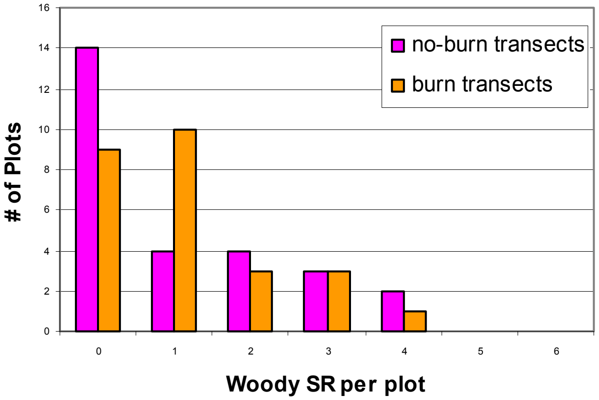
Woody Species Richness
Camp Meeting Rock
Before Burn Treatment



Woody Species Richness
Heggie's Rock
Before Burn Treatment



Woody Species Richness
Camp Meeting Rock
After Burn Treatment



Woody Species Richness
Heggie's Rock
After Burn Treatment

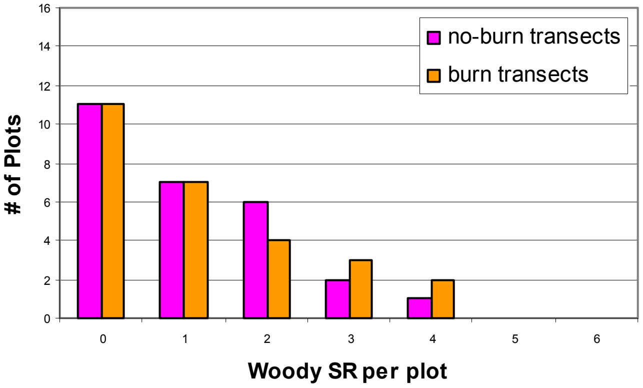
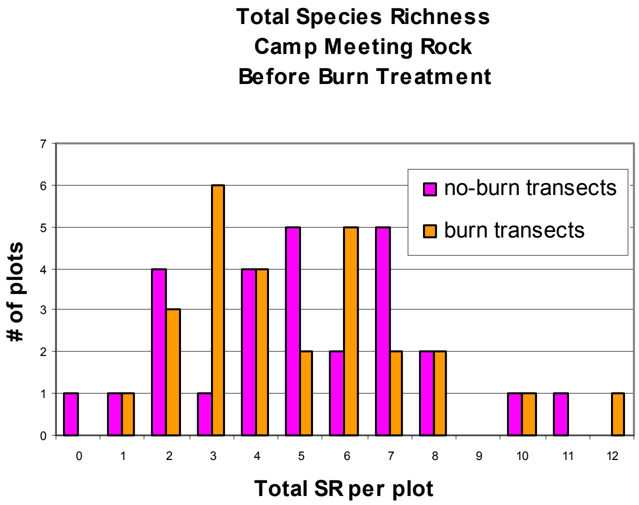
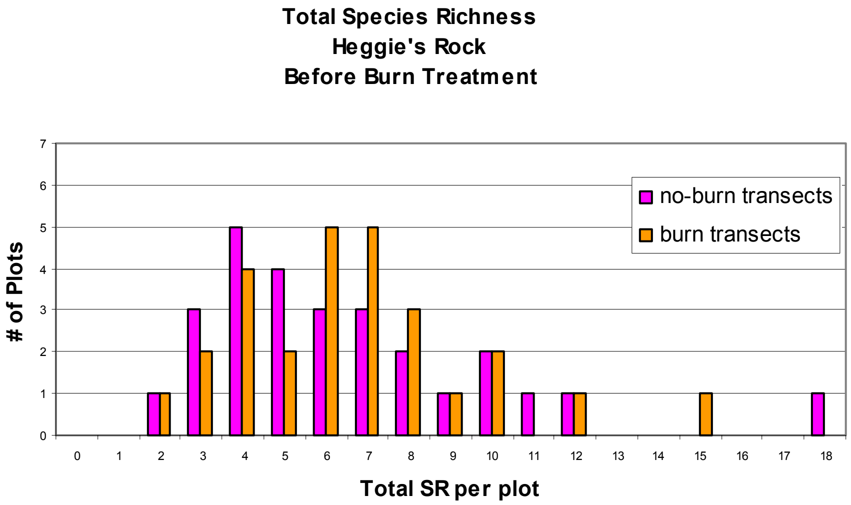
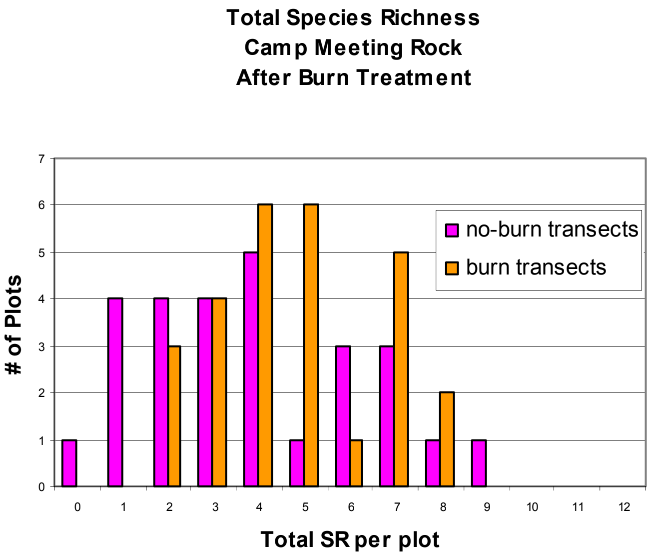


Figure 2.5: Total Species Richness for Camp Meeting Rock and Heggie's Rock, Before and After Burn Treatments


Total Species Richness
Camp Meeting Rock
Before Burn Treatment
# of plots
0
1
2
3
4
5
6
7
no-burn transects
burn transects
0
1
2
3
4
5
6
7
8
9
10
11
12
Total SR per plot


Total Species Richness
Heggie's Rock
Before Burn Treatment
# of Plots
0
1
2
3
4
5
6
7
no-burn transects
burn transects
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
Total SR per plot


Total Species Richness
Camp Meeting Rock
After Burn Treatment
# of Plots
0
1
2
3
4
5
6
7
no-burn transects
burn transects
0
1
2
3
4
5
6
7
8
9
10
11
12
Total SR per plot


Total Species Richness
Heggie's Rock
After Burn Treatment
# of Plots
0
1
2
3
4
5
6
7
8
no-burn transects
burn transects
0
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
Total SR per plot

CHAPTER 3

EXOTIC INVASION ON TWO GRANITE ROCK OUTCROP ECOTONES IN THE PIEDMONT OF THE SOUTHEASTERN UNITED STATES

Abstract

Piedmont granite rock outcrops in the southeastern United States support a plant community rich in endemic and rare plant species. It is unclear what effect increasing pressure from exotic invasion may have on the ecotone of this community. I removed invasives from the ecotones of two granite rock outcrops and monitored species richness and plant occurrence along random stratified belt transects that began at the rock edge and traversed into the forest interior. Hemispherical photographs and soil depth were used to monitor changes in light availability and soil nutrients. Vegetation classes demonstrated spatial distribution patterns related to distance from the rock edge and soil depth, but removal of invasives had no apparent effect. Sodium, calcium, and magnesium were all found to be significant predictors of invasive species occurrence. Species richness did not increase significantly over the three year study period.

Keywords: ecotone, endemic species, invasion, granite outcrop, management, species richness

Introduction

In the southeastern United States, outcrops of bare rock persist in the midst of oak-hickory forests. The insular nature of these exposures encourages allopatric speciation, which has contributed to endemism and rarity in the outcrop flora and resulted in a unique assemblage of plant species (Wyatt and Allison, 2000). Most of this granite rock outcrop habitat lies in the Piedmont of Georgia, with the majority of outcrops occurring just outside the Atlanta metropolitan area. The predominant threat to these exposures has been quarrying, but they are secondarily impacted by exotic species invasion, cattle grazing and eutrophication, dumping, quarry dust deposition, fire building, vandalism, foot traffic, and off-road vehicles. Development that accompanies a growing urban center is now increasing anthropogenic disturbance on these rock exposures and creating new opportunities for invasion of surrounding habitats.

Rock outcrops of the southeastern Piedmont can be gently sloping flatrocks or towering monadnocks of stone and are collectively referred to as granite rock outcrops although many may be comprised of granite, granite schist or granite gneiss. Species on granite outcrops have display numerous physiological and life history adaptations to withstand the environmental extremes of a life in thin soils on the rock surface. Outcrops also host a number of species whose range extends to Piedmont only on such sites. In Georgia, the Piedmont province lies between the Coastal Plain, and Ridge and Valley and Blue Ridge Mountain provinces. This position in the region has allowed recruitment of species from the surrounding provinces into the rock outcrop community.

The rock outcrop ecotone can be considered a distinct community with its own suite of characteristic endemic species, such as *Forestiera ligustrina*, *Sedum pusillum*, and *Anemone berlandieri*, and specific range of environmental conditions. Wyatt and Allison (2000) listed a number of endemic, near endemic, or disjunct endemic species found on Piedmont rock outcrops as well as commonly encountered “weed” species. The outcrop ecotone habitat is typically more mesic than the soil islands of the bare rock surface because of runoff captured from the rock. The outcrop ecotone plant community represents a mixture of endemic rock outcrop species and Piedmont oak-hickory forest flora.

Colonization of the edge communities of many rock exposures by exotics is startling. In a survey of 52 outcrops from Alabama through Georgia and into South Carolina, Caspary and Rickard (2007) reported that 39 were impacted by invasive species. *Ligustrum sinense* and *Lonicera japonica* appear to be the most dominant colonizers of rock outcrop margins and are known to crowd out native understory plants (Oosting and Livingstone, 1964; Wharton, 1978; Wyatt and Allison, 2000). A number of studies have documented that the presence of *Ligustrum sinense* resulted in decreased species richness and abundance in southeastern Piedmont forests (Kittell, 2001; Wilcox and Beck, 2007). Baskin and Baskin (1988) determined that granite outcrop endemics are not shade tolerant, so a change in canopy cover could trigger a change in species community composition. Invasion results in resource competition, prevention of recruitment of endemic species, and altered fire regimes (Csurhes and Edwards, 1998), and can also reduce the amount of light reaching the forest floor. Moreover, competition from exotic species for resources can transform the species distributions in ecotones.

Competition is believed to play a major role in structuring communities (Gause, 1934; Hutchinson, 1959; MacArthur and Levins, 1967). Changes in the abundance of species, especially those that influence water and nutrient dynamics, trophic interactions, or disturbance regime, are known to affect the structure and functioning of ecosystems (Chapin et al., 1997; Mack and D'Antonio, 1998). Research documenting interspecific competition on rock outcrops has identified soil depth, soil moisture, and biotic competition as significant factors influencing the population dynamics of endemics (Sharitz and McCormick, 1973; Houlé, 1990). Competition and shading have been identified in previous studies as the main limiting factors in southeastern United States outcrop vegetation (Barbour, Burk, and Pitts, 1987; Baskin and Baskin, 1988).

Exotic species invasion is expected to have altered the plant community structure on the rock outcrop ecotones of the southeastern Piedmont and perhaps species richness. Due to the insular nature of the rock outcrop habitat, rapid colonization by invasive opportunists may ultimately lead to extirpation of native species. MacArthur and Wilson (1967) introduced the theory of island biogeography which proposes that increased immigration of exotic species will fuel species turnover and drive native species to extinction. On Western Australia outcrops, weeds have become a threat to conserving the diversity of the granite outcrop community (Pigott, 2000). Wyatt (1997) predicted that the open nature of the site and low competition in outcrop habitats present an opportunity for invasion by many weedy species. Invasion has been correlated with species richness (Lonsdale, 1999; Levine, 2000; Brown and Peet, 2003). Harrison et al. (2006) explored invasion on serpentine barrens, systems with soils derived from ultra mafic rock that have low calcium to magnesium ratios and are generally understood to be nutrient poor and high in heavy metals. These systems are particularly rich in endemic flora and Harrison et al. suggested that diversity hot spots may be even more under attack. They found that invasion did correlate with overall species richness, but it was negatively correlated with serpentine endemic and rare herbs.

Invasive exotics can be promoted by disturbance (Mack and D'Antonio, 1998; Mawson, 2000). Disturbance can initiate a change in resource availability by exposing bare soil and freeing bound nutrients, which creates an opportunity for exotic species to enter a system (Hobbs, 1989; Davis, Grime, and Thompson, 2000). Exotic species have been known to alter disturbance regimes (Hobbs and Huenneke, 1992). Outcrops in Western Australia have become vulnerable to opportunistic invasions through fertilizer drift and disturbance from fire and recreational activities. If the character of the ecotone community changes, those changes may carry through to the rest of the rock outcrop community. No publications appear to address the invasion of exotic flora within the granite rock outcrops of the Southeast, and Baskin and Baskin (1988) urged study of the interaction between aggressive weeds and rock outcrop endemics.

Rock outcrop communities are an ideal habitat in which to research edge-effect dynamics and spatial trends because the transition from the forest edge to the rock surface is so abrupt. It is unclear whether ecotone habitats serve to moderate or facilitate disturbance. I surveyed the vegetation along the ecotone of rock outcrop habitats where active invasion is occurring. I studied the effects of removing invasive flora on species richness and percent cover of native vascular and non-vascular flora, and site openness, by comparing invaded, control, and removal transects in the ecotones of two granite outcrops. I expect that a removal treatment will ultimately lead to increased species diversity at small scales for herbs, grasses, and woody plants. I chose to look at cations in the soil to see if I could detect any spatial trends between cation levels and vegetation class occurrence or species diversity. Removing the invasive layer should free resources for native flora and result in increased occurrence of outcrop vegetation classes.

Methods

Study Sites

Panola Mountain, a 370-hectare Georgia Department of Natural Resources State Conservation Park, is located in Rockdale County, Georgia (33° 14' 32” N, 85° 8' 49" W, ca. 10.5 km from Lithonia, Georgia). Panola Mountain is a monadnock that rises 55 m above the surrounding landscape. The vegetation surveys, however, were performed in an adjacent area of rolling flatrocks with a much less pronounced slope. The park is underlain by Panola Granite, a non-porphyritic igneous rock (USGS, 2000). Ashlar sandy loam, Pacolet sandy loam, and Ashlar-Pacolet-Wedowee complex soils characterize the forest community surrounding the rock (USDA, 2006a). Mean daily maximum and minimum temperatures were 23.8ºC and 10.9ºC for 2006, 24.1ºC and 11.1ºC for 2007, and 23ºC and 10.4ºC for 2008 at the Jonesboro weather station (located ca. 17 km southwest of Panola Mountain). Total rainfall measurements for 2006, 2007, and 2008 were 1.100 m, 0.770 m, and 0.980 m, respectively.

Rock and Shoals, jointly owned by the Georgia Department of Natural Resources and Athens-Clarke County, is located in Clarke County, Georgia (33° 32' 35” N, 82° 15'13” W, ca. 8 km from Athens, Georgia). Rock and Shoals is a glade flatrock, with *Ligustrum sinense*, *Elaeagnus umbellata*, and *Lonicera japonica* invasion documented at the site (Nourse and Nourse, 2004). The rock is a combination of biotite gneiss and felspathic biotite gneiss (Survey, 1999). Pacolet sandy clay loams with slopes varying from 0-15 % and Madison Louisa complex soils surround the rock exposure (USDA, 2006b). Weather data recorded by the Watkinsville RAWS weather station (located ca. 3.2 km southwest of Rock and Shoals) measured mean daily maximum and minimum temperatures at 23.6ºC and 10.5ºC for 2006, 23.9ºC and 10.9ºC for 2007, and 22.6ºC and 10.2ºC for 2008. Total rainfall for 2006, 2007, and 2008 was measured at 1.130 m, 0.782 m, and 0.960 m, respectively.

Botanical survey

At each site, 18 stratified random belt transects, 1 m wide by 20 m long, were surveyed in the fall of 2006 and spring of 2007 (study-year 0), the fall of 2007 and spring of 2008 (study-year 1) and fall of 2008 and spring of 2009 (study-year 2) (Figure 3.1). A random number was used to specify the position of the first transect along a compass heading parallel to the rock edge and subsequent transects were placed at 10 m intervals. These transects extended from the edge of the rock exposure into the surrounding forest (Figures 3.2 and 3.3). Six of the belt transects served as control transects and were located in relatively uninvaded ecotonal areas. The remaining 12 transects were located in a heavily invaded ecotone, and six of these were chosen at random to be cleared of invasive species. The Georgia Exotic Pest Plant Council maintains a ranked list of invasive plants and only exotic species with the highest designation (Category One) were removed from transects (GA-EPPC, 2011). Removal was conducted by hand pulling and cutting and painting stems with Glyphosate. Every effort was made to minimize the disturbance caused by the removal process. Each removal transect was cleared 2 m beyond the existing transect in all directions and invasives were regularly cleared throughout the study period. Drought and high summer temperatures kill many plants, so surveys were performed in the spring and fall of the research period. Total species richness and woody species richness were measured for plots 1, 10, and 20 of each transect during each survey session. Vegetation class occurrence was recorded in plots 1, 10, and 20 for the plant categories of grasses; herbaceous plants, ferns and fern allies; invasive flora; lichens; mosses; vines; and woody plants.

Soil nutrients and depth

Soil samples were taken at a depth of 0-5 cm in the O horizon of plots 1, 10, and 20 of each transect at each site. Soils were analyzed for calcium (Ca), magnesium (Mg), potassium (K), and sodium (Na) using atomic absorption and for pH. In the data analysis, soil nutrient data were log transformed to help normalize data distributions. Soil depth was averaged from three independent measurements taken in plots 1, 6, 10, 16, and 20 in each transect at each site.

Canopy light

Hemphispherical photography, an indirect optical technique, was used to characterize canopy openness. Hemispherical photographs were taken in plots 1, 6, 10, 16, and 20 in each transect at each site using a Nikon F 35 mm digital camera and a Nikkor 8 mm f2.9 fisheye lens. The resulting image had a 183 degree view. Photographs were taken at ground level with the top of the image oriented north in fall of 2006 before the burn treatment and in fall of 2008 after the burn treatment.

Data Analysis

Soil depths were not normally distributed and were transformed by taking the square root of the three independent observations for each plot, and then averaging over those observations to create the variable “sqdepthav”. An Analysis of Variance (ANOVA) was used to calculate differences between sites and for plot-distance. Soil nutrient variables Na and Ca were transformed using the natural log because they demonstrated extremely right-skewed distributions. The linear model used is:

$$Y_{ijklmn} = \mu + s_i + T_j + t_{k(i)} + P_l + \alpha SD_{ijkl} + e_{ijklm}$$

where Y is the is the soil variable in observation *n* with all experimental factors as specified in the model, μ is the intercept, s_i is the effect of site i (i = Panola Mountain or Rock and Shoals), T_j is the effect of treatment j (j = Control, Invaded, Removed), $t_{k(i)}$ is the effect of transect k at site i, P_l is the effect of plot l (l = 1, 10, or 20), and e_{ijklm} is the error for the observation.

Gap Light Analyzer (GLA) software was used to compute the parameter "percent site openness" (Frazer, Canham, and Lertzman, 1999). Percent site openness is percentage of total sky area that is found in overlapping gaps in the canopy.

A linear mixed model with a natural log-transformed response was used to analyze the classes of grasses; herbs, ferns, and fern allies; invasives; vines; and woody plants. This model uses the variables of effect of the site (PM or RS), treatment (Control, Invaded, or Removed), transect (A-R), study-season (Fall 0, Spr 0, Fall 2, Spr 2), plot-distance (1, 10, or 20), and the interaction of study-season x treatment. A logistic regression was used to analyze the lichen class and a two-step analysis using a logistic regression and a linear mixed model was used to perform an analysis on the moss and invasion classes, due to a large number of plots with no occurrence for these vegetation classes. The invasive class variable for the invasive linear mixed model was square-root transformed to help normalize data distributions.

A Poisson regression was performed to determine the relative influence of soil depth, canopy openness, Ca, Mg, Na, and K on herbs, ferns, and fern ally species richness, grass species richness, woody species richness, and total species richness.

Tukey-Kramer adjustments were used to allow for multiple comparisons with an α value < 0.05.

Results

Soil depth, soil nutrients, and light

Average soil depths did not vary significantly between sites or among transects at each site (Table 3.1). Plots at distance 1 had significantly less soil depth than those at distances 6, 10, 16, and 20; plots at distance 6 had significantly less soil depth than 10, 16, and 20; plots at distance 10 had significantly less soil depth than plots at distance 20 but were similar to 16. Plots at distance 16 and 20 are not statistically different. The general trend is a pattern of increased depth with increased distance from the rock edge.

Table 3.2 shows the percent available soil nutrients, pH, and percent site openness for control, invaded, and removal transects at each site. Table 3.3 displays the linear mixed model results for soil nutrients, pH, and site openness. Soil depth demonstrated a positive significant correlation with pH. Plot-distance is a significant predictor of K, Ca, and Na. Plots at distance 1 have significantly lower K and Ca, on average, than at distance 10, but all other comparisons between plots are not statistically different. Plot 1 has a significantly lower expected Na than plot 20, but all other pairwise comparisons are not significantly different. The variables of treatment, plot, or sqdepthav did not show any significant relationship with Mg. Analysis of the hemispherical photographs through linear regression indicates that the site openness does not show any significant correlation with soil depth. Study-year 0 was significantly more open than study-year 2 (Table 3.2 and Table 3.3) but no interaction between study-year and treatment could be found, so the change in canopy openness cannot be attributed to the invasive removal treatment.

Vegetation Class

Endemic species encountered in the ecotone are listed in Appendix A. Sodium, calcium, and magnesium were all found to be significant predictors of invasive occurrence. Table 3.4 demonstrates that in most cases plot was the most significant variable in vegetation class occurrence. The Tukey-Kramer adjustment was used to modify the p-value where multiple comparisons were made.

Vegetation class occurrence was not significantly impacted by the removal treatment. Most vegetation classes demonstrate a significant correlation with plot-distance from the rock. Plot 1 in a given transect typically displays a very different plant community than expected from random chance when compared to plot 20. Grasses, herbs and ferns, and vines demonstrate a positive relationship with magnesium and calcium. Vines also correlate positively with pH. Mosses display a negative correlation with calcium. Woody plants appear to have a positive relationship with available magnesium and a negative relationship with sodium in the soil. For the invasive cover class, the control treatments had significantly less invasive occurrences than the invaded transects and invasive species did demonstrate a spatial distribution, in which they were more likely to occur at a greater distance from the rock edge. Vines showed a significant difference between the treatments of Control and Invasive and Control and Removal, but not a significant difference between Removal and Invasive, so these differences cannot be attributed to the removal treatment.

Species richness

A total of 136 species of vascular and non-vascular flora were identified in the surveys at Panola Mountain, and 10 of these or 7.4% are rock outcrop endemics as defined by Wyatt and Allison (2000) (Appendix A). The survey performed at Rock and Shoals documented a total of 165 species with 13 of these (7.4 %) identified as rock outcrop endemic species. Species with at least five recorded appearances are reported at given plot-distances of 1, 6, 10, 16, and 20 for each site in Appendix A. The table also reports associated vegetation classes for each species and level of endemism as determined by Wyatt and Allison (2000). Rock and Shoals also has a higher diversity of invading species represented in the sampled transects, with *Microstegium vimineum*, *Lonicera japonica*, *Ligustrum sinense*, *Lespedeza cuneata*, and *Elaeagnus umbellata* all in abundance. *Nandina domestica* and *Lonicera mackii*, two recently recognized invasive threats (GA EPPC Category 2 (2011), but classified in the woody vegetation class), were also identified in the survey transects at Rock and Shoals outcrop, but they were not encountered in great abundance.

Table 3.5 shows the Poisson models for grass, herb/fern/fern allies, woody plant, and total species richness. Figure 3.4 illustrates how the richness of these vegetation classes changed over the study period. The removal treatment did not significantly affect species richness. Grass species richness was negatively correlated with calcium. Survey season was the only significant predictor of herbaceous/fern/fern ally diversity, likely because many of the forbs encountered at rock outcrops are annual plants that reproduce in the spring. Woody species richness is positively correlated with calcium and magnesium in the soil. Soil depth is a significant predictor of both woody plant species richness and total species richness. Survey-season and distance from rock are also significant predictors of overall species diversity. Total species richness amounts increased throughout the study, but this change was not significant. More change in species richness is represented in control and removal plots, but this increase was not significant.

Discussion

Removing exotic invasives from treatment plots did not significantly alter species diversity, or significantly alter vegetation class distributions or abundance. Hemispherical photographs revealed a significant difference in canopy openness, but these changes occurred evenly across the different transects and cannot be attributed to treatment. These differences could be attributed to annual variation. Extreme variation in precipitation during the study period may have confounded the outcome of the removal treatment. The state of Georgia experienced a severe drought from May to December of 2007. Nearby weather stations reported total annual precipitation levels of 0.77 m for Panola Mountain (330 mm less than 2006 annual precipitation totals) in 2007 and 0.782 m for Rock and Shoals (348 mm less than 2006 levels) in 2007. Moisture in the ecotone can vary, as site topography encourages run-off from the rock to channel through some areas but not others. Therefore, a patchy response to low moisture is expected in the ecotone. Observations of the study site confirm that heavier mortality was experienced in some areas due to shallower soils and lack of rain. Rock outcrop plant communities experience periodic fluctuations in moisture and temperature. Thus, it is possible that the environmental extremes in the course of the study period did little to alter long-term trends in the plant community structure of the outcrop ecotone. It is also possible that time or area scales were too small to capture a community response to the treatment.

Soil depth has been found to be a predictor of vegetation class (Burbanck and Platt, 1964). In this study, soil depth correlated with distance from the rock edge, and soil depth predicted vegetation class distributions as well. The rock outcrop ecotone habitat at both study sites is very glade-like, with persistently shallow soils between rock islands. Some species of grasses, mosses, herbs, and ferns typically occur in the first couple of meters from the rock edge, but this suite of plants was found at a greater distance from the rock edge because of a continued trend in shallow soils.

Soil nutrients in a rock outcrop community are known to be limited, and cation availability in the soil appears to be a predictor of plant occurrence. The resource availability hypothesis suggests that invasive species will be positively correlated with available nutrients (Davis, Grime, and Thompson, 2000). In serpentine communities, Ca and Mg have been positively correlated with abundance of herbs and species richness (Harrison, 1999; Safford and Harrison, 2004). Harrison (1999) found that herbaceous species (grasses were included in this species class designation) diversity on serpentine outcrops showed significant correlations with soil calcium levels. In this study, calcium and magnesium levels were predictive of grass, herb/fern, and vine vegetation class occurrence. Invasive plant occurrence was tied to calcium, magnesium, and sodium. Woody vegetation class and woody species diversity also showed a spatial correlation with magnesium. Herb/fern species diversity, however, did not correlate with calcium levels, only with study season.

Increased species diversity has been linked with ecosystem stability and health since Elton related invasion probability to low diversity systems (Elton, 1958; Chapin et al., 1997). The diversity stability hypothesis predicts that species-rich ecosystems will be more resistant to disturbance (Tilman and Downing, 1994; Tilman, 1996). In this study, control plots were not significantly more species-rich than invaded plots, and removal plots did not appear to change in species composition once the invasive elements were eliminated. Therefore, no link between diversity and invasion was found. Harrison et al. (2006) discovered that invasion on California serpentine barrens was positively correlated with native species richness. Levine and D'Antonio (1999) found that the diversity-invasibility relationship is often ambiguous, but they recognized a consistent positive correlation between invasive abundance and species diversity.

A number of studies have suggested that aboiotically harsh and unproductive environments tend to be less invaded than more productive ones (Zartman, Norton, and Stern, 1964; Crawley, 1987; Zolyomi, 1987; Hobbs, 1989; Mack, 1989). The results of this study fit with that pattern, with greater invasion documented at greater distances from the rock edge. Baskin and Baskin (1988) and McVaugh (1943) suggested the harsh environment of the rock outcrop tames weedy species into less vigorous aggressors. Studies have shown that in serpentine barrens, limitations in soil nutrients combined with plant competition have made these environments more resistant to invasion (Williamson and Harrison, 2002; Daehler, 2003; Going, Hillerislambers, and Levine, 2009). Huenneke et al. (1990) posited that this nutrient availability may be more important than physical disturbance in determining community invasibility. These findings provide encouragement that, whereas the ecotone environment may be affected, the soil islands on the rock surface may remain more intact.

Many weedy species are known to occur on rock outcrops, and Wyatt and Allison (2000) have even suggested that some of these species may have used outcrops as an evolutionary springboard. The relationship between these "weedy" species and endemic rock outcrop flora appears to be very different from the relatively recent invasion by several aggressive exotics. Many species such as *Nuttallanthus canadensis*, *Oenothera biennis*, *Plantago virginica*, *Triodanis perfoliata*, and *Krigia virginica* are considered old field weeds, but appear to coexist with outcrop flora without restructuring environmental conditions.

Ligustrum sinense, *Lonicera japonica*, *Microstegium vimineum*, *Lespedeza cuneata*, *Rosa multiflora*, and *Elaeagnus umbellata* are the Category One invasive species encountered in the surveys. *Ligustrum sinense* is by far the greatest in abundance and does the most to visibly modify a rock outcrop community by closing the mid-story canopy. *Ligustrum* sp. are known for their abundant production of highly dispersible fruits, wide range of adaptability, and limited natural predators (Batcher, 2000). Invasive Category One species were removed throughout the study period. After the initial clearing, removal transects required some investment of energy to remove exotics in the following year, but this effort was not considerable. *Ligustrum sinense* appeared to seed back in to plots from the overhanging canopy, and is expected to take over removal plots again, but reestablishment of invasives is expected to be slower than previously thought.

Personal observations suggest that, even with the high mortality of invasives caused by extreme environmental conditions in rock outcrop habitats, an established buffer of parent plants provide a means of quick replacement. There were very few occurrences of exotic species outside the “Category One Invasive” designation so these plants are not expected to persist at the exclusion of other native species.

It is commonly stated that disturbance facilitates invasion (Grime, 1979; Hobbs and Huenneke, 1992). Williamson and Harrison (2002) found disturbance and seed supply to be equally important in promoting the spread of invasion in the serpentine soils of California. Hester and Hobbs (1992) applied an invasive removal treatment after prescribed burning on southwestern Australian outcrops and were able to document increased plant vigor and seed-set. Removal experiments with invasive *Acer platanoides* revealed that other exotic species spread where tree removals opened the canopy (Webb, Iv, and Dwyer, 2001). Species diversity may strengthen the resilience and resistance of ecosystems to environmental change, as suggested by Chapin et al. (2000), but the introduction of disturbance will weaken ecosystem defenses to invasion. Disturbance is expected to homogenize a community (Velland et al., 2007). In the case of rock outcrops in the Southeast, if increasing anthropogenic disturbance doesn't destroy the habitat outright, increased nutrient inputs from waste and runoff or physical disturbance from recreational traffic and grazing threaten to overwhelm these sites with exotics.

If the ecotone of the rock outcrop system changes into a closed canopy, wind-dispersed seed flow from one island-like rock to another will be affected. Cadenasso and Pickett (2001) have demonstrated how the structure of the forest edge can regulate seed dispersal. Local dynamics within populations will invariably determine the processes of migration, extinction, and colonization (Husband and Barrett, 1996). Island hopping colonization events will be required to weather the coming storm of climate change for many of the relict endemic species of outcrop habitats. More work is required to understand better how these long-distance dispersal events take place with rock outcrop endemics and how invasion may be changing immigration success. Until we understand better these dispersal processes, it is critical that we act to protect the remaining population reserves of rock outcrop endemic species.

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Table 3.1: Average Soil Depth by Plot and Soil Depth ANOVA

Site	**Function**	**Plot 1** (cm)	**Plot 6** (cm)	**Plot 10** (cm)	**Plot 16** (cm)	**Plot 20** (cm)	**Total**
Panola Mountain	Mean	6.0	28.2	46.7	67.7	64.9	42.7
	S.D.	3.8	11.3	16.1	29.3	32.5	31.6
	Max	16.0	51.0	89.0	100.0	100.0	100.0
	Min	0.0	4.0	18.0	12.0	2.0	0.0
Rock and Shoals	Mean	7.0	22.9	30.6	32.5	41.7	35.4
	S.D.	10.2	23.7	24.1	20.2	25.2	25.3
	Max	51.0	100.0	100.0	89.0	100.0	100.0
	Min	0.0	1.0	2.0	4.0	11.0	1.0
ANOVA soil depth and site							
P-Value							0.1958
ANOVA soil depth and transects							
P-Value							0.5242
ANOVA soil depth and plot							
P-Value							**<.0001**

Table 3.2: Invasion Study Soil Nutrients, pH, and Percent Site Openness

Site	Transect Type/ Function	% Ca	% Mg	% K	% Na	pH	% Site Openness Year 0	Year 2
Panola Mountain	Control Plot							
	Mean	0.0275	0.0020	0.0307	0.0046	5.1500	47.59	41.09
	S.D.	0.0828	0.0005	0.0047	0.0021	0.4602	6.97	7.11
Panola Mountain	Invaded Plot							
	Mean	0.0026	0.0019	0.0289	0.0037	5.3333	47.44	41.39
	S.D.	0.0049	0.0004	0.0060	0.0014	0.5308	7.25	7.17
Panola Mountain	Removal Plot							
	Mean	0.0011	0.0022	0.0285	0.0036	5.1006	48.41	42.58
	S.D.	0.0019	0.0005	0.0047	0.0010	0.4712	6.85	6.25
Rock and Shoals	Control Plot							
	Mean	0.1200	0.0010	0.0296	0.0024	5.6159	40.24	35.52
	S.D.	0.1548	0.0005	0.0046	0.0006	0.6273	8.94	5.91
Rock and Shoals	Invaded Plot							
	Mean	0.0335	0.0011	0.0261	0.0025	5.4712	39.65	36.00
	S.D.	0.0634	0.0003	0.0068	0.0011	0.4983	6.44	7.72
Rock and Shoals	Removal Plot							
	Mean	0.0808	0.0010	0.0268	0.0026	5.7767	42.03	36.76
	S.D.	0.1545	0.0003	0.0081	0.0009	0.5719	8.91	6.45

Table 3.3: Invasion Study Soil Nutrient, pH, and Site Openness Model Results

Effect	Study-year	Plot	Estimate	Standard Error	P-Value
pH (Linear Mixed Model)					
Intercept			5.2191	0.2413	0.0294
Plot		1	0.03512	0.01766	**0.0507**
lnCa (Linear Mixed Model)					
Intercept			-5.8489	1.1386	0.1224
Plot		1	-1.4068	0.5873	**0.0194**
Plot		10	0.03322	0.5826	0.9547
Plot		20	0	.	.
Mg (Linear Mixed Model)					
Intercept			0.001529	0.000488	0.1966
K (Linear Mixed Model)					
Intercept			0.02845	0.001183	0.0265
Plot		1	-0.00171	0.001248	0.1763
Plot		10	0.001573	0.001227	0.2041
Tukey-Kramer Adjustment					
Plot		1/10	-0.00328	0.001248	**0.0283**
Plot		1/20	-0.00171	0.001248	0.3641
Plot		10/20	0.001573	0.001227	0.41
lnNa (Linear Mixed Model)					
Intercept			-5.9521	0.2253	0.0241
Plot		1	0.2427	0.08303	**0.0047**
Plot		10	0.1557	0.08179	**0.0611**
Plot		20	0	.	.
Tukey-Kramer Adjustment					
Plot		1/10	0.08699	0.08303	0.5496
Plot		1/20	0.2427	0.08303	**0.0129**
Plot		10/20	0.1557	0.08179	0.1453
Site Openness (Linear Mixed Model)					
Intercept			38.8783	3.2117	0.0525
Study-year	0		5.336	0.7095	**<.0001**
Study-year	2		0	.	.

Table 3.4: Invasion Study Vegetation Class Model Results

Effect	Treatment or Season	Plot	Estimate	Standard Error	P-Value
G Class (Linear Mixed Model (ln trans.)) with Tukey-Kramer Adjustment					
Intercept			-5.1182	1.7115	0.2054
Plot		1/10	0.9492	0.2	**<.0001**
Plot		1/20	1.2183	0.2265	**<.0001**
Plot		10/20	0.269	0.1239	0.0774
sqdepthav			0.1509	0.03647	**<.0001**
lnCa			-0.1211	0.02621	**<.0001**
K			66.7349	13.9912	**<.0001**
lnNa			-0.4859	0.1956	**0.0134**
H Class (Linear Mixed Model (ln transformed))					
Intercept			1.0789	0.5978	0.3221
lnCa			0.1088	0.0245	**<.0001**
Mg			430.71	142.07	**0.0026**
L Class (Logistic Regression) with Tukey-Kramer Adjustment					
Intercept			-0.3653	0.9639	0.7694
Plot		1/10	1.4917	0.5783	**0.0276**
Plot		1/20	1.9606	0.6491	**0.0076**
Plot		10/20	0.4688	0.3851	0.4435
sqdepthav			-0.3348	0.126	**0.0082**
M Class (Linear Mixed model)					
Intercept			15.872	9.4063	0.3406
Plot		1/10	6.3301	4.5991	0.3548
Plot		1/20	20.4833	5.1656	**0.0003**
Plot		10/20	14.1532	3.366	**0.0001**
lnCa			-2.3072	0.5911	**0.0001**
sqdepthav			-2.6107	0.9104	**0.0045**
I Class (Logistic Model for comparing Control to Invaded transects)					
Intercept			5.054	0.9477	0.118
Treatment	C		-5.051	1.0935	**0.0001**
Treatment	I		0	.	.
Plot		1	-3.7909	0.6664	**<.0001**
Plot		10	0.7664	0.5143	0.1374
Plot		20	0	.	.

Table 3.4 Continued: Invasion Study Vegetation Class Model Results

Effect	Treatment or Season	Plot	Estimate	Standard Error	P-Value
I Class (Lin. Mixed Model for Invaded trans.) with Tukey-Kramer Adjustment					
Intercept			10.5813	2.0774	0.1234
Plot		1/10	-2.4479	0.2318	**<.0001**
Plot		1/20	-2.4689	0.2646	**<.0001**
Plot		10/20	-0.021	0.2339	0.9956
lnNa			1.0474	0.3321	**0.002**
lnCa			0.2523	0.06721	**0.0003**
Mg			854.08	327.97	**0.0103**
V Class (Linear Mixed Model (ln transformed))					
Intercept			2.5711	0.7368	0.1777
Treatment	C		-0.6298	0.1765	**0.0012**
Treatment	I		-0.00106	0.175	0.9952
Treatment	R		0	.	.
Plot		1	-0.7313	0.1869	**0.0001**
Plot		10	-0.03477	0.1061	0.7432
Plot		20	0	.	.
Season	Fall_0		-0.2687	0.1188	**0.0242**
Season	Spr_0		0.08172	0.1188	0.4919
Season	Fall_2		0.0132	0.1188	0.9116
Season	Spr_2		0	.	.
sqdepthav			0.1294	0.03076	**<.0001**
lnCa			0.09863	0.02594	**0.0002**
pH			-0.2955	0.1015	**0.0038**
Mg			370.51	142.61	**0.0097**
W Class (Linear Mixed Model (ln trans.)) with Tukey-Kramer Adjustment					
Intercept			-2.5374	1.0066	0.2404
Plot		1/10	-0.364	0.1723	**0.0887**
Plot		1/20	-0.1827	0.1993	0.6301
Plot		10/20	0.1813	0.1156	0.2608
lnNa			-0.4215	0.1535	**0.0063**
Sqdepthav			0.1814	0.0335	**<.0001**
Mg			249.31	112.93	**0.0279**

Table 3.5: Species Richness Model Results

Effect	Treatment or Season	Plot	Estimate	Standard Error	P-Value
G Species Richness (Poisson regression)					
Intercept			0.3953	0.4106	0.5121
lnCa			-0.0363	0.01486	**0.015**
HF Species Richness (Poisson regression) with Tukey-Kramer Adjustment					
Intercept			0.9608	0.5241	0.3179
Season	Fall_0 vs. Spr_0		-0.4146	0.08728	**<.0001**
Season	Fall_0 vs. Fall_2		0.09119	0.09804	0.7887
Season	Fall_0 vs. Spr_2		-0.4763	0.08623	**<.0001**
Season	Spr_0 vs. Spr _2		0.5058	0.08975	**<.0001**
Season	Spr_0 vs. Fall_2		-0.06169	0.07668	0.8523
Season	Spr_2 vs. Fall_2		-0.5675	0.08874	**<.0001**
W Species Richness (Poisson regression) with Tukey-Kramer Adjustment					
Intercept			0.4109	0.2454	0.3427
Plot		1/10	-0.4415	0.1456	**0.0073**
Plot		1/20	-0.3188	0.1668	0.1368
lnCa			0.08954	0.02172	**<.0001**
sqdepthav			0.09644	0.02616	**0.0003**
Mg			217.45	109.01	**0.0468**
Total Species Richness (Poisson regression) with Tukey-Kramer Adjustment					
Intercept			2.2267	0.1705	0.0486
Season	Fall_0 vs. Spr_2		-0.1526	0.04059	**0.0011**
Season	Spr_0 vs. Spr _2		0.145	0.04101	**0.0026**
Season	Spr_0 vs. Fall_2		-0.02465	0.03926	0.923
Season	Spr_2 vs. Fall_2		-0.1696	0.04078	**0.0002**

Figure 3.1: Invasion Study Transect Design

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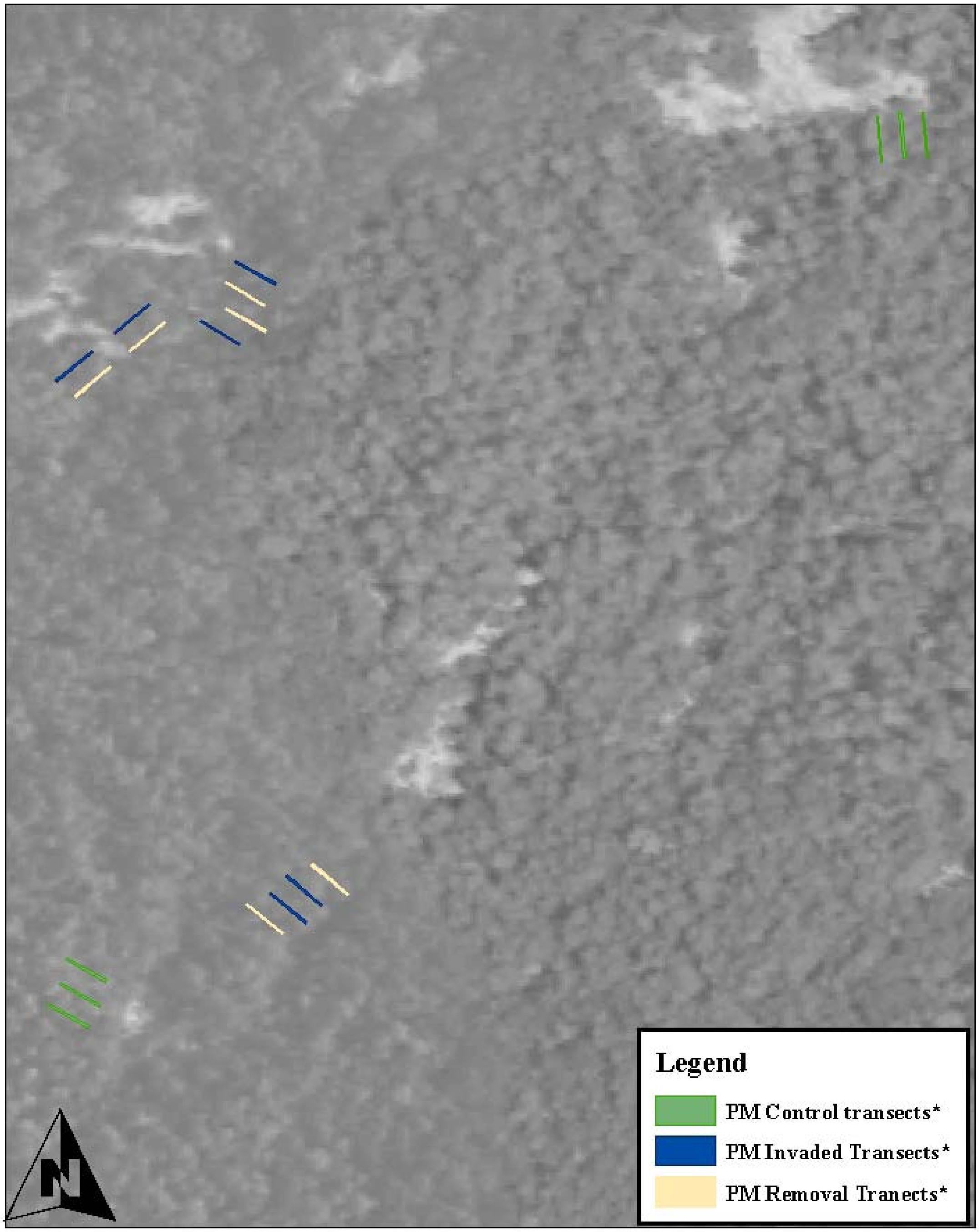
🟩 **Variables Used for Statistical Analysis:**
Vegetation Classes: Grasses, Herbs, Ferns, and Fern Allies, Invasives, Lichens, Mosses, Vines, and Woody Plants
Total Species Richness and Woody Species Richness
Environmental Data:
d) Soil Nutrients & pH
e) Soil Depth
f) Hemispherical Photograph

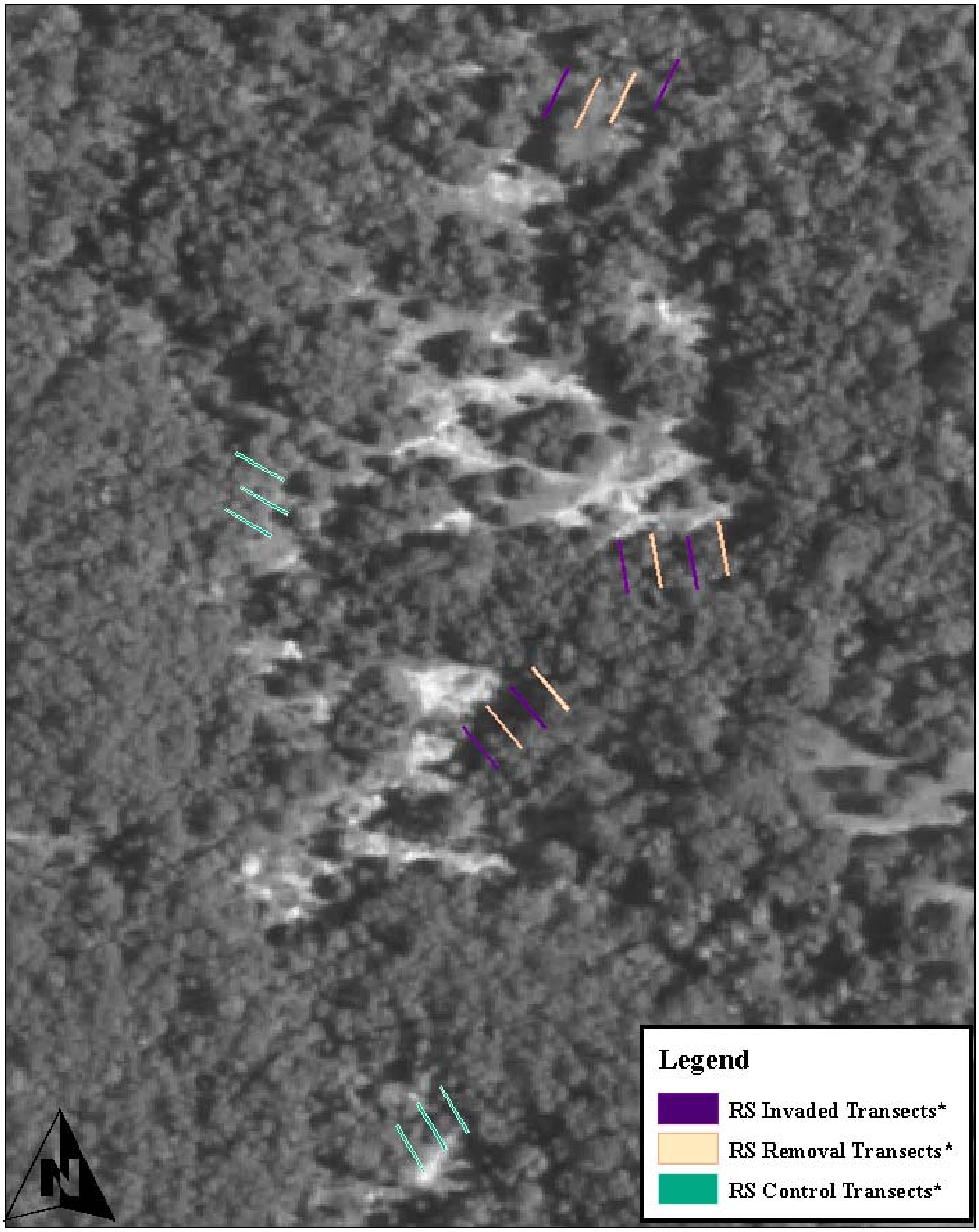
🟪 **Variables Used for Statistical Analysis:**
Vegetation Classes: Grasses, Herbs, Ferns, and Fern Allies, Invasives, Lichens, Mosses, Vines, and Woody Plants
Environmental Data:
c) Soil Depth
d) Hemispherical Photograph

Figure 3.2: Panola Mountain Transect Locations



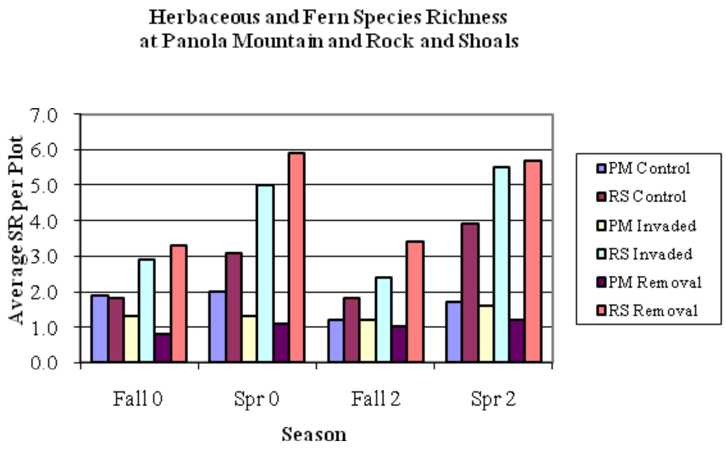
*Transect outlines have enhanced weight for visual clarity

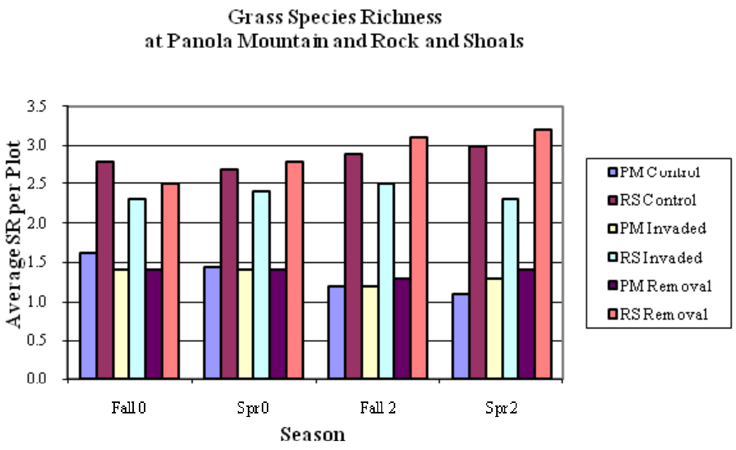
Figure 3.3: Rock and Shoals Transect Locations

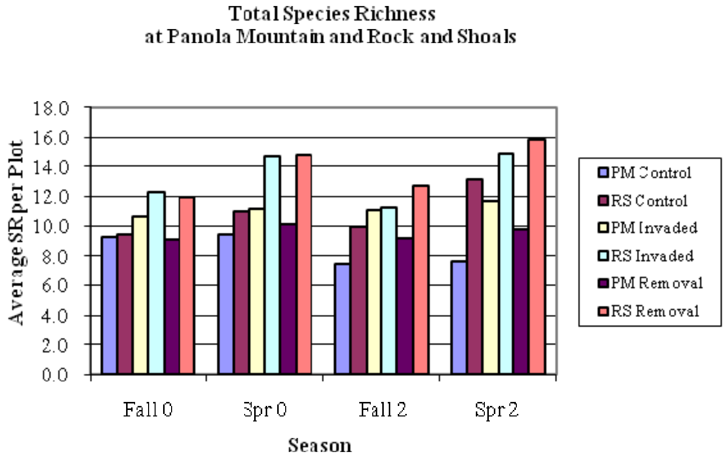


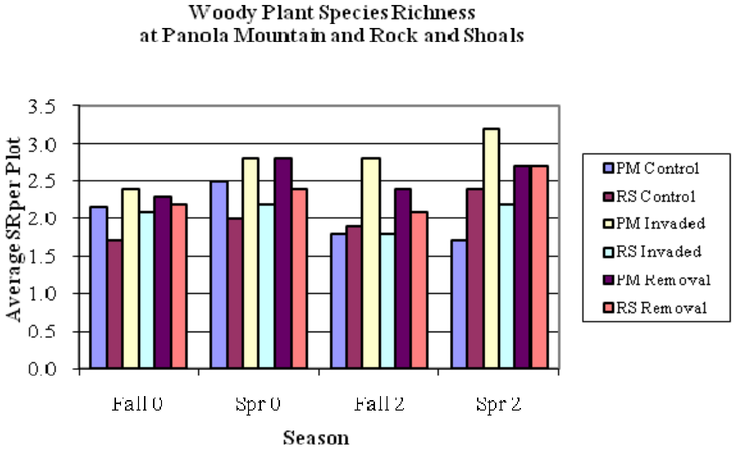
*Transect outlines have enhanced weight for visual clarity

Figure 3.4: Species Richness for the Herbs and Ferns, Grasses, Woody Plants, and Total Plants at Panola Mountain and Rock and Shoals


Herbaceous and Fern Species Richness at Panola Mountain and Rock and Shoals


Grass Species Richness at Panola Mountain and Rock and Shoals


Total Species Richness at Panola Mountain and Rock and Shoals


Woody Plant Species Richness at Panola Mountain and Rock and Shoals

CHAPTER 4

USING SPATIAL ANALYSIS TO PREDICT HABITAT THREATS AND RARE SPECIES OCCURRENCE IN GRANITE ROCK OUTCROPS

Abstract

The granite rock outcrops of the southeastern Piedmont host a unique plant community rich in endemic and rare flora. The center of distribution of these rock exposures lies within the developing reach of the Atlanta urban center and this close proximity poses a constant threat for the rock outcrop habitat and its associated flora and fauna. Data on tree canopy, impervious surface, and landcover produced by the University of Georgia Natural Resources Spatial Analysis Laboratory were used to examine a 33-year snapshot of landcover change surrounding Georgia outcrops. A 1 km buffer was analyzed around outcrops and quarries in Georgia to determine land use patterns, assess present habitat threats, and predict future development effects. Predictive models of species geographic distributions were used to determine the key environmental variables that determine local and regional species patterns and to assess the relevance of patch size and distribution, habitat complexity, and population density for outcrop and rare plant occurrence. Landcover trends reveal increasing urban development and deforestation in all areas that were analyzed. Impervious surface surrounding rock outcrops significantly increased from 1991 to 2005. Latitude, longitude, elevation, and outcrop spatial complexity were predictors of rare plant occurrence, whereas outcrop size or isolation were not reasonable predictors of rare species occurrence. Population density, shape complexity, and geology were predictive of outcrop occurrence. Outcrop habitats that are more irregular and located toward the southeast at higher elevations are more likely to have associated rare plant species. These models may be applied to specific habitats to assist in identifying threats.

Keywords: Georgia Piedmont, landscape ecology, rare species, rock outcrop, spatial analysis

Introduction

The granite rock outcrops of the southeastern United States are floristically diverse and support many endemic and rare plants (McVaugh, 1943; Natureserve, 2010). As replicated terrestrial islands, they present an opportunity to improve our general understanding of threats to rare species persistence. As a consequence of their isolated and restricted distributions, the endemic species of this habitat type may be particularly vulnerable to extirpation and, ultimately, extinction (Wyatt and Allison, 2000). The integrity of the granite rock outcrops has been challenged by quarrying, dumping, and traffic from all-terrain vehicles, horses, and humans (USFWS, 2007). Southeastern rock outcrops extend from Virginia to Alabama in an irregular broken band within the Piedmont physiographic province and the majority of outcrop habitat is concentrated near the Atlanta metropolitan area (McVaugh, 1943). With increasing urban pressure, there is an immediate need to identify the threats to this distinctive habitat. The present condition of the granite rock outcrops in the southeastern Piedmont is not well understood or documented. These systems have been used to as models to study plant community structure and composition, particularly in regard to exploring the concept of plant succession (Keever, Oosting, and Anderson, 1951; Burbanck and Platt, 1964; Shure and Ragsdale, 1977; Phillips, 1981; Burbanck and Phillips, 1983). They have not been used extensively, however, to further our knowledge about patterns of species rarity.

Looking at regional landscape trends of these insular environments through the lens of spatial analysis presents an opportunity to study the ecological processes that govern species rarity and endemism and to identify threats present on larger scales. The comparison of rock outcrops or inselbergs to islands has been a common theme in the literature (Main, 1997; Hopper, 2000; Porembski and Barthlott, 2000a; Wyatt and Allison, 2000; Burke, 2003). Island metapopulation dynamics are believed to foster plant speciation (Carlquist, 1974), and Murdy (1968) pointed out that southeastern outcrops have long served as sites for active speciation in the region. The size and degree of isolation of separate patches may play an important role in shaping patterns of species diversity, either by increasing species diversity via speciation or by reducing diversity via increased rates of extinction. Whittaker (1998) recommended that the theory of island biogeography (MacArthur and Wilson, 1967) be applied to terrestrial insular landscapes like rock outcrops. Outcrops also have well-defined edges, which makes them more easily defined in geographic terms.

The outcrop landscape is visibly structured at a number of different spatial scales. From the most diminutive endemic flora to the largest landscape trees, or from tiny pools and to huge soil "islands," patterns seem to be replicated in this landscape. Levin (2000) points out that looking at an ecological system at multiple scales can tell us a lot about the interplay between pattern and diversity dynamics. Geographic technology enables us to take a landscape-level look at rock outcrop habitats and allows us to ask what the key environmental variables are that determine local and regional species patterns. Large experiments are critical because landscape-level processes are often not predictable from fine-grained studies (Carpenter, 1996; Clark et al., 1999). Geographic Information Systems (GIS), coupled with aerial imagery and landcover data availability, makes analysis of these habitats possible. This effort will help clarify emerging threats to rock outcrop habitat in terms of real land use changes and trends in the Georgia Piedmont.

Spatial habitat structure can influence rarity on regional scales. Previous studies have looked at biogeographical patterns and regional species diversity in outcrop habitats around the world (Porembski, Seine, and Barthlott, 1997; Harrison, 1999; Meirelles, 1999; Harrison, Viers, and Quinn, 2000; Burke, 2002), but few have used these environments to study regional trends in plant rarity. Wiser et al. (1998) paired species distribution patterns with spatial attributes to create models of rare species occurrence on high elevation outcrops in North Carolina. On rock outcrops, species diversity has been found to vary dramatically within habitats along ecological gradients and among different rock islands (Porembski, Seine, and Barthlott, 1997). In the southeastern United States, Baskin and Baskin (1988) confirmed that species diversity was not tied to edaphic conditions. Gaining a greater understanding of species distribution patterns will be extremely valuable in assessing future areas for conservation, assisting in present management of existing protected areas, and directing and defining priorities for future research.

I measured rock outcrop habitat representation in Georgia and analytically assessed habitat distribution patterns to determine trends in habitat structure, rare endemic species distribution, or shifting land use patterns that could be indicators of present and future threats. This research should help to quantify the degree of habitat destruction and the relative severity of different threats to these systems. Congruent with previous studies of islands (Hobohm, 2000; Loehle, 2006), I expect the least isolated, largest outcrops to have the most endemic species. I also identified relationships between certain plant species and outcrop locations and types.

Methods

Study System

Outcrops occur in the southeastern Piedmont from east-central Alabama through Georgia, the Carolinas, and into south central-Virginia. They are most numerous, however, in Georgia, and the greatest concentration of endemic vascular plant species occurs there (Murdy, 1968). The majority of these rock exposures lie within 50 km of Atlanta, one of the largest metropolitan areas in the United States.

Landcover Data

Working in conjunction with the University of Georgia Natural Resources Spatial Analysis Laboratory (NARSAL), all known Georgia outcrop and quarry exposures were mapped from previously digitized polygons from the Georgia Gap Analysis Project (GA-GAP). Summary statistics, including the results of a cluster analysis, for all digitized outcrop and quarry polygons were generated using Patch Analyst 4.2.10 in ArcGIS (Rempel, Carr, and D., 2008). Landcover data generated for the Georgia Land Use Trends (GLUT) Project from LANDSAT imagery is available for the years 1974, 1985, 1991, 1998, 2001, and 2005. Information for impervious surface data and tree canopy data in Georgia is also available for the years 1991, 2001, and 2005. Tree canopy represents tree density modeled for each individual pixel of aerial photography.

Impervious surface data are defined as "Man-made materials found in developed landscapes including rooftops, driveways, sidewalks, decks and other materials that prevent water from infiltrating into the ground," as well as natural impervious surface from rock outcrops (NARSAL, 2011). A 1 km buffer was used around each rock outcrop polygon for each landcover analysis.

Rare Plant Data

I used logistic regression to build predictive models about the probability of outcrops and rare species occurrence. Rare plant species locations were identified from Element of Occurrence records provided by the Georgia Department of Natural Resources (GA DNR), Nongame Conservation Section, and compared with granite outcrop maps to determine any geographic patterns in species area and species isolation. These Element of Occurrence records were largely collected and revisited by James Allsion, a Georgia State Botanist and recognized authority on granite rock outcrop habitats and rare plant species, so the quality of this information is expected to be high. The 18 rare species associated with rock outcrop habitat are listed in Table 4.1 along with their designation of endemism according to Wyatt and Allison (2000).

A 100 m buffer was created around a total of 903 outcrop polygons and 117 quarry polygons to build a dataset of 369 polygons. Predictor variables of longitude (center), latitude (center), sum of outcrop area, geology, soil type, population density, distance to nearest neighbor, and average mean shape index (a shape complexity variable) were used to predict species occurrence. Patch Analyst 4.2.10 and Hawth's Analysis Tools 3.27, in concert with ArcMap 9.3 and digital elevation models, were used to calculate the metrics of longitude, latitude, elevation, sum of outcrop area, distance to nearest neighbor, and average mean shape index (Beyer, 2004; Rempel, Carr, and D., 2008). Average mean shape index is a calculation of shape complexity and is equal to 1 when patches are circular, and increases with increasing shape irregularity (MSI = patch perimeter divided by the square root of patch area). Population density was calculated from census records for the state of Georgia for the year 2009. Geological classification was determined using the digital geographic map of Georgia, and soils were classified using STATSGO (STATSGO, 1998; GADNR, 1999).

All variable distribution properties were examined and transformed as necessary before performing statistical analyses. Models were built using logistic regression based on a critical P-value of 0.05. Variables that were not statistically significant were removed in a stepwise backward elimination procedure.

Results

Maps were created of all rock outcrops and quarries in Georgia (Figures 4.1 and 4.2). Rock outcrop polygons were used to calculate a total of 1,539.8 ha of rock outcrop habitat and a total of 1,067 ha of quarry land area in the state of Georgia. Summary statistics for rock outcrops and quarries are reported in Table 4.2. A cluster analysis revealed that outcrops are highly clustered.

For the years of 1991-2005, canopy cover did not change significantly, but impervious surface increased significantly in the 1 km area surrounding outcrop polygons (Figure 4.3, Table 4.3). Landcover progression was graphed to show land use trends for each landcover class for the 31-year time period (Figure 4.4). The landcover classes of clear-cut and sparse; quarries, strip mines, and rock outcrops; low-intensity urban; and high-intensity urban demonstrate an increasing trend over the 31-year period, whereas row crops and pastures; forested wetland; deciduous forest; and evergreen forest generally show a decreasing trend (Figure 4.4).

Logistic regression models for rare species occurrence and outcrop/quarry location are reported in Table 4.4. The best predictors in the rare species regression model were latitude, outcrop spatial complexity, elevation, and longitude. Surprisingly, neither outcrop size nor isolation were good predictors of rare species occurrence. Outcrop shape complexity and geology were predictive of outcrop occurrence.

Discussion

Gaining an understanding of how much rock outcrop habitat remains in Georgia (1,539.8 ha) and how that amount relates to the amount of habitat that has already been destroyed by quarrying is of major importance to land managers and stewardship agencies around the state. Much of the area currently occupied by quarries (1,067 ha) was likely outcrop habitat in the past.

Significant increases in the amount of total impervious surface reinforce findings that these areas are transitioning into more urban, developed environments. Tree canopy cover is concentrated in the higher percentage classes, revealing that most of the tree cover surrounding these outcrop sites is restricted to high tree density areas. Tree canopy appears to have decreased much more in the years of 1991-2001 than it did from 2001-2005. This corresponds to the overall landuse trends of decreasing forest cover classes and increasing clear-cut lands. Impervious surface consistently increased surrounding rock outcrop sites. Landcover changes show the most dramatic changes in deciduous forest, evergreen forest, and agricultural landcover transitioning into mixed forest, clear-cut and sparse vegetation, low-intensity and high-intensity urban landcover classes.

High-intensity urban land classes did not increase dramatically, but low-intensity urban landcover classes spiked considerably in the period 1974-2005. Given the surge of development in the region, it is likely that the urban and clear-cut/sparse vegetation landcover classes will succeed forest and agricultural landcover classes in many areas.

Tree canopy cover, impervious surface cover, and landcover classes are created by rasterizing aerial imagery using algorithms into a grid of 30 m cell sizes. A great deal of information can be gained from this type of landscape level analysis, but detail is lost in the process. Spatial analysis is a valuable tool, provided it is reinforced with on-the-ground investigation of predicted trends.

Habitat complexity, latitude, longitude, and elevation were the most significant predictors of rare species occurrence identified by logistic regression. The model suggests that outcrops that have irregular perimeters, occur at higher elevations, and are located toward the southeast have a higher probability of supporting a rare flora. Michael et al. (2008) found that species richness of reptiles depended on patch size and habitat structure. The significance of habitat complexity in the model suggests that ecotones may be even more important thant previously thought in fostering rare species in rock outcrop habitats. There is a recognized north-to-south gradient in plant richness (Qian, 1999; Loehle, 2006), so latitude was expected to correlate negatively with rare plant occurrence. Baskin and Baskin (1988) discovered that outcrop endemic plant physiological performance was not dictated by edaphic factors, so it is expected that geology and soil would not be good predictors of rare plant occurrence. Spatial autocorrelation is ubiquitous in geographically based data so variables like longitude, latitude, and elevation are expected to have a degree of correlation.

A positive correlation has been noted between outcrop size and species diversity (Meirelles, 1999; Porembski and Barthlott, 2000b). Houlé (1990) found that plant species richness was significantly positively correlated with island area in Southeastern outcrops. Wiser et al. (1998) found area and elevation to be the best predictors of species richness in high elevation outcrops of North Carolina. Murdy (1968) observed that the geographic "center of endemism" for granite outcrops in the Southeast correlates with the concentration of exposed rock, so outcrop area was expected to be an indicator of rare plant occurrence. A cluster analysis revealed that outcrop habitats are highly clustered, yet nearest neighbor distances did not emerge as a relevant predictor of rare plant occurrence. The comparatively low percentage of endemism (1%) observed on granite rocks in Texas was partially explained by the continuous nature of outcrop distribution (Walters and Wyatt, 1982). Studies of island floras show that time and isolation are correlated with numbers of endemics (Hobohm, 2000; Loehle, 2006), so it was expected that isolation would be a predictor of rare plant occurrence. Instead, I found no relationship with the variables of outcrop sum area and nearest neighbor. It is possible that when species diversity and Georgia Piedmont outcrop spatial distributions are analyzed, a relationship between richness and outcrop size and area may emerge.

Environmental patterns strongly influence ecological processes. Wyatt (1997) observed that most outcrop plants are not adapted for long-distance dispersal. Burke (2002) determined that stepping stone corridors are important particularly for taxa with short dispersal ranges. Strong genetic differentiation between outcrop populations of *Arenaria uniflora*, *Isoetes melanospora*, and *Isoetes tegetiformans* (Wyatt, 1984; Wyatt, Evans, and Sorenson, 1992; Van De Genachte, 1996) indicate that these plants have probably island-hopped from one exposure to another as local erosional forces created new outcrops and covered older ones (Wyatt and Fowler, 1977). Small, isolated habitats, where rescue effects are reduced, are expected to have higher rates of turnover (Wright, 1985; Hinsley, Bellamy, and Newton, 1995; Krieger, 2003). Extinction rates are expected to be lower in larger habitat islands because these areas offer a bigger target for dispersal (Jones and Diamond, 1976; Brown and Kodric-Brown, 1977). Spatial habitat structure influences the ability of local sites to sample from their regional pools (Harrison et al., 2006). Gonzalez et al. (1998) point out that, in a community in which metapopulation dynamics dominate, connectivity between habitat fragments is critical to retain genetic elasticity and viable populations. Drury (1974) suggested that the number of subpopulations is more important for population persistence than the overall population size because relict species are likely to harbor genetic diversity in isolated populations.

If the connectivity of rock outcrop populations is severed, it is likely that the associated plant community will not be able to respond to climate change. Due to the limited geographic distribution of rock outcrop habitat, it is conceivable that climate change will make outcrop species particularly susceptible to extinction. It is also possible that outcrop habitats may support flora that will be particularly tolerant of climate change, given the history of survival in climatic extremes (Krieger, 2003).

Combining the information generated in this study with future spatial data analysis will help to create a more complete large-scale picture of threats emerging for rock outcrop habitat in the southeastern Piedmont. Future areas of study could include an exploration of outcrop species diversity and richness in the Southeast. Spatial analysis efforts could be expanded to include more predictor variables such as precipitation and temperature. Climate models could be paired with rare species occurrence to increase predictive value for prioritizing conservation efforts. It would be advisable to combine spatial analysis information with research on genetic differentiation among populations when making decisions about conservation priorities.

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Table 4.1: Rare Species on Rock Outcrops in Georgia

Scientific Name	Common Name	Level of Endemism[a]
Allium speculae	flatrock onion	Disjunct Endemic
Amorpha schwerinii	Schwerin's false indigo	not listed
Amphianthus pusillus	snorkelwort	Strict Endemic
Amsonia ludoviciana	Louisiana bluestar	Disjunct Endemic
Anemone berlandieri	tenpetal thimbleweed	Half-Endemic
Arabis missouriensis	green rockcress	Half-Endemic
Eurybia avita (*Aster avitus*)	Alexander's rock aster	Strict Endemic
Cuscuta harperi	Harper's dodder	Incidental
Draba aprica	openground draba	Disjunct Endemic
Eriocaulon koernickianum	dwarf pipewort	Disjunct Endemic
Fimbristylis brevivaginata	glade fimbry	Near-Endemic
Isoetes melanospora	blackspore quillwort	Strict Endemic
Isoetes tegetiformans	mat-forming quillwort	Strict Endemic
Pilularia americana	American pillwort	"Provincial" endemic
Portulaca umbraticola ssp. coronata	wingpod purslane	Near-Endemic
Ptilimnium nodosum	harperella	Incidental
Pycnanthemum curvipes	Stone mountainmint	Half-Endemic
Sedum pusillum	granite stonecrop	Strict Endemic

[a] Endemic designation according to classification by Wyatt and Allison, 2000

Table 4.2: Georgia Outcrop and Quarry Summary Statistics

	Outcrops	Quarries
# of Polygons	903	117
Area (ha)	1,539.8	1,067.1
Mean (m^2)	17,052.1	91,200.7
Median (m^2)	6,163.0	35,151.0
Min (m^2)	397.2	691.5
Max (m^2)	1,626,888.7	1,582,247.2
Patch Size Coefficient of Variance[a] (m^2)	399.4	208.3
Patch Size Standard Deviation[b] (m^2)	68,098.8	189,980.1
Total Edge (m^2)	504,536.1	149,391.8
Edge Density (m/patch)	0.03	0.014
Mean Patch Edge (m)	558.7	1,276.9
Mean Shape Index[c]	1.53	1.45
Average Nearest Neighbor Distance	427.30	16,720.60
Moran's I[d]	0.01	0.05
Z-score[e]	2.0	2.06

[a] Patch size coefficient of variance is a measure of the covariance of the patch area

[b] Patch size standard deviation is a measure of the standard deviation of patch area

[c] Mean shape index is a measure of shape complexity where 1=circular and x >1 is a more irregular shape

[d] Moran's I is a measure of spatial autocorrelation where values range from −1 (indicating perfect dispersion) to +1 (perfect correlation). A zero value indicates a random spatial pattern

[e] Z-score is another way to evaluate spatial autocorrelation or clustering, where values > 1.96 or < −1.96 indicate spatial autocorrelation that is significant at the 5% level.

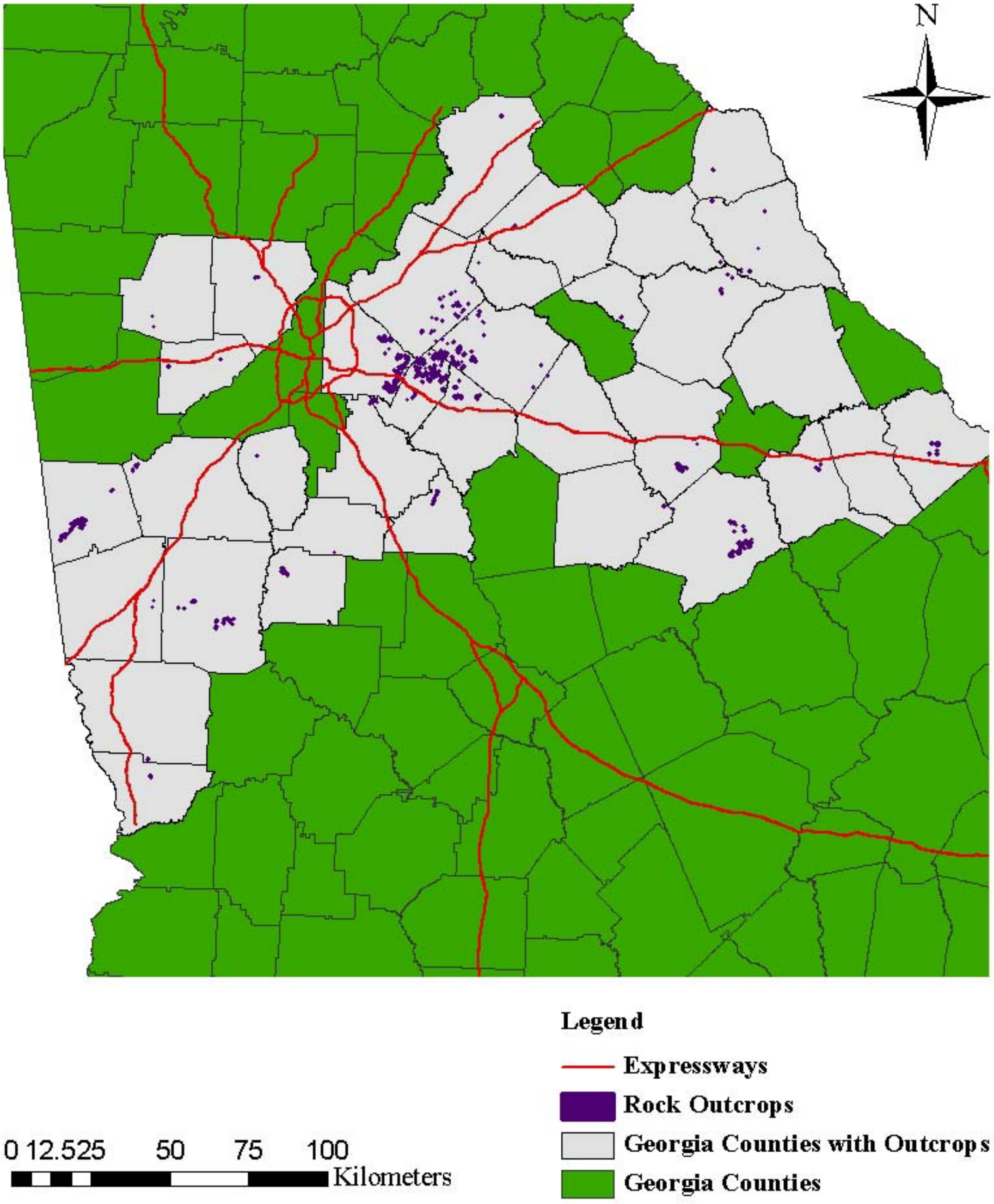
Table 4.3: Tree Canopy Cover and Impervious Surface Tests for Significance

	Total Change (ha)	Test	P-value
Tree Canopy			
1991 vs. 2001	-395	Mann-Whitney Rank Sum Test	0.597
2001 vs. 2005	-3	Mann-Whitney Rank Sum Test	0.642
1991 vs. 2005	-398	Mann-Whitney Rank Sum Test	0.870
Impervious Surface			
1991 vs. 2001	81	t-test	**0.015**
2001 vs. 2005	128	t-test	**0.005**
1991 vs. 2005	209	Mann-Whitney Rank Sum Test	**<0.001**

Table 4.4 Logistic Regression of Rare Species Occurrence and Outcrop Presence

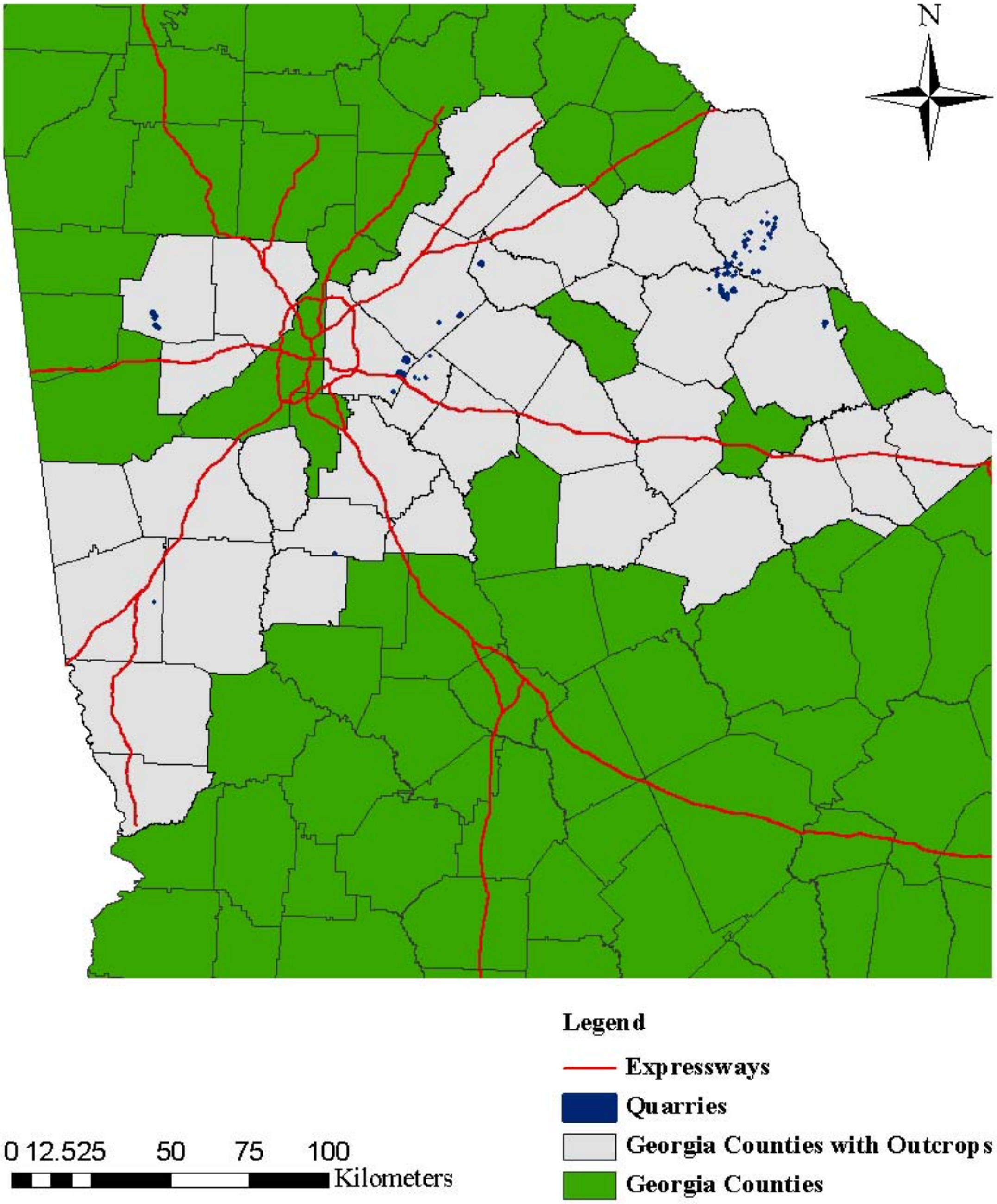
Dependent Variable	Independent Variable	Intercept	MSE	P-Value
Rare Species Occurrence	Constant	132.32	43.880	**0.003**
	Longitude	0.78	0.351	**0.026**
	Latitude	-2.16	0.563	**<0.001**
	Avg Mean Shape Index	1.48	0.405	**<0.001**
	Elevation	0.01	0.004	**0.006**
Outcrop occurrence	Constant	1.11	1.136	0.929
	Pop_area	1.00	0.000	**0.014**
	Avg Mean Shape Index	4.47	0.645	**0.021**
	Geology			**0.049**
	Geology: Amphibolite	0.93	1.294	0.952
	Geology: Biotite Gneiss	142700000.00	8427.505	0.998
	Geology: Granite	0.38	0.775	0.217
	Geology: Non-Poryph. Granite	600092699.60	12746.443	0.999
	Geology: Other	0.07	0.951	**0.005**

Figure 4.1: Map of Georgia Rock Outcrops



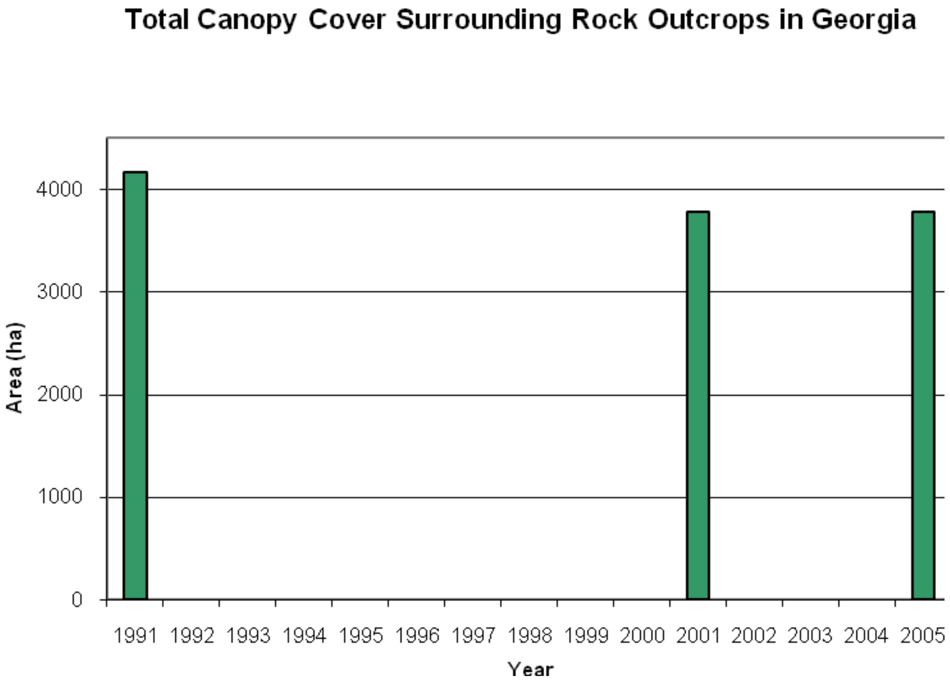
Granite rock outcrop polygons suppied by the Natural Resources Spatial Analysis Labratory (NARSAL 2006).

Figure 4.2: Map of Georgia Quarries



Granite rock outcrop polygons suppied by the Natural Resources Spatial Analysis Labratory (NARSAL 2006).

Figure 4.3: Total Tree Canopy Cover and Impervious Surface Area for 1km Surrounding Rock Outcrops in Georgia


Total Canopy Cover Surrounding Rock Outcrops in Georgia

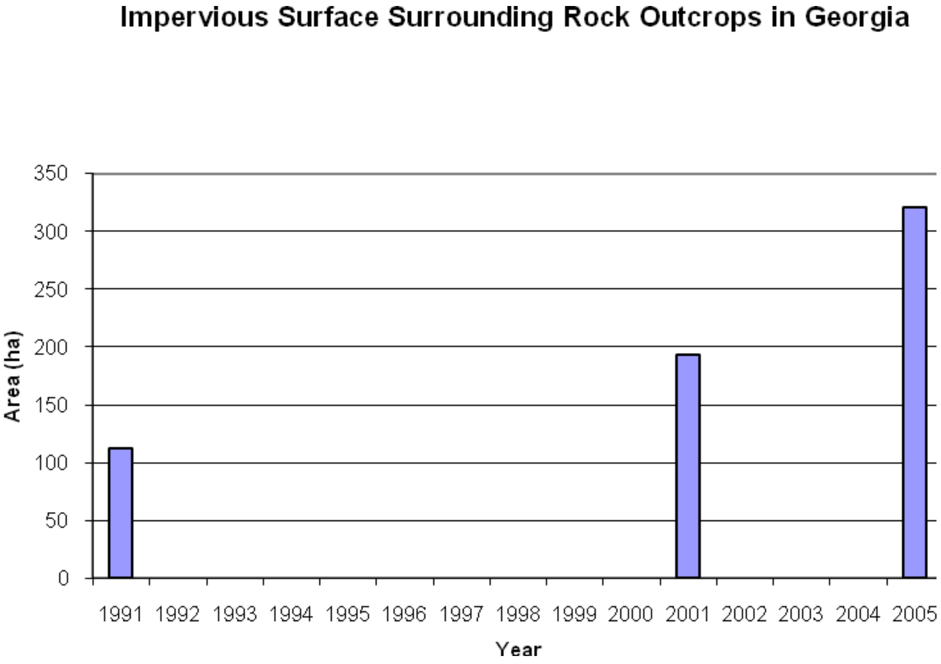
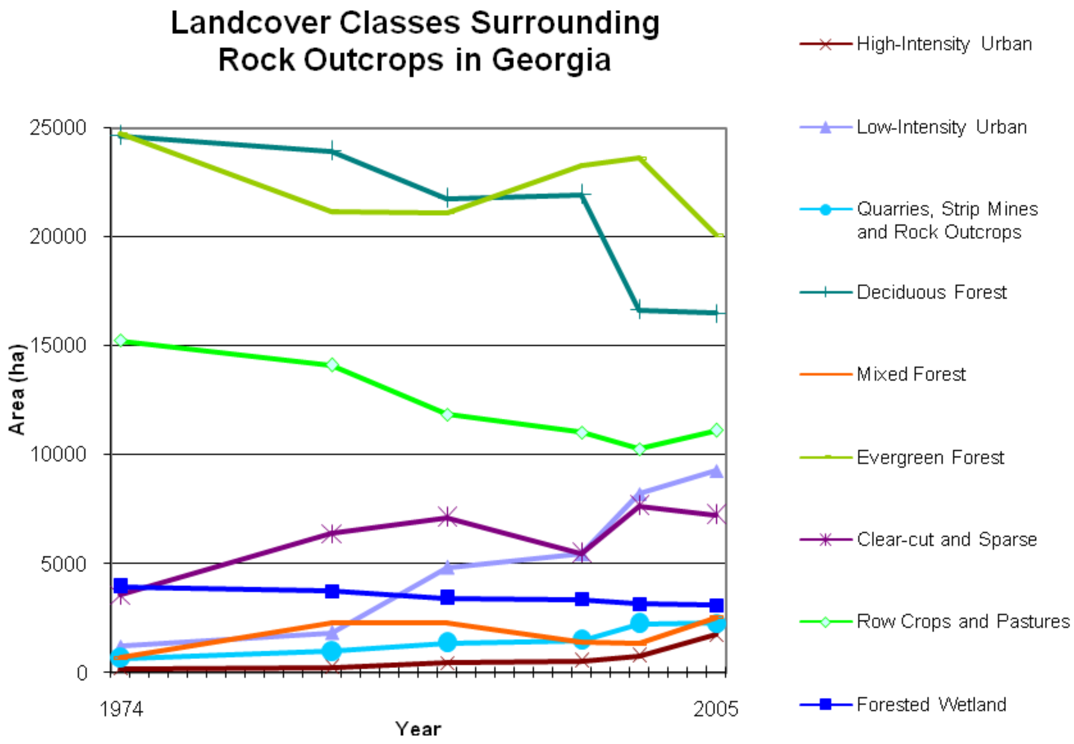
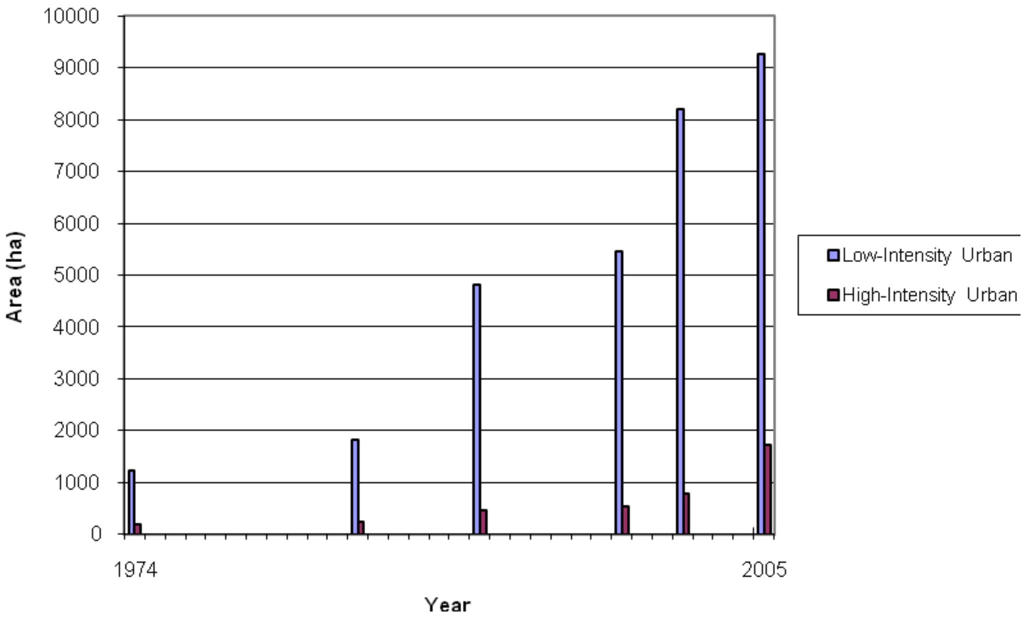

Impervious Surface Surrounding Rock Outcrops in Georgia

Figure 4.4: Landcover Classes for 1 km Surrounding Rock Outcrops in Georgia


Landcover Classes Surrounding Rock Outcrops in Georgia


High-Intensity and Low-Intensity Urban Landcover Classes Surrounding Rock Outcrops in Georgia

CHAPTER 5

MANAGEMENT RECOMMENDATIONS FOR GRANITE ROCK OUTCROP HABITAT IN THE PIEDMONT OF THE SOUTHEASTERN UNITED STATES

Granite outcrops of the southeastern United States are unusual landscapes where ecological and environmental extremes meld with scattered population distributions to create unique plant communities. This habitat has been diminished by almost 40% due to quarrying. Preliminary and historical surveys have documented that 64-79% of sites visited have suffered some degree of disturbance (Magee, 1987; Caspary and Rickard, 2007). This habitat is also impacted by invasion, grazing, vehicular traffic, and dumping. The outcrop ecotone appears to be changing due to fire suppression, invasion of exotic flora, and direct and indirect human disturbance. I sought to determine what role prescribed fire and invasion play in restructuring rock outcrop plant communities, particularly in the ecotone where these disturbances are most likely to occur. This research project was designed at multiple scales in order to gain the most complete picture of present-day threats to outcrop systems and their surrounding habitats and to identify patterns indicative of underlying ecological processes (Levin, 1992). These field studies were performed to gain a better understanding of local, short-term disturbance effects on microhabitats and niche relations in outcrop ecotone environments. I also wanted to study how spatial distribution patterns on regional, long-term scales inform ecological processes and see if these emerge as geographic predictors of rare plant occurrence.

It is believed that outcrops serve as a refuge for fire-sensitive species. Thus, there is concern that using fire to manage outcrop habitats may negatively impact rare and endemic species (Burke, 2002). Burn treatments, however, did not appear to negatively impact species richness or vegetation class occurrence in the ecotone community of Camp Meeting Rock or Heggie's Rock. These ecotones were burned during the winter and the prescribed burns did not carry well in the thin soils of outcrop margins. Hopper (2000) advised that fire should be avoided or rarely used on outcrops, but his advice is based on experience with outcrops in Western Australia, which are much different ecological environments with many more known fire refuge species than the outcrops of the Georgia Piedmont. The management of these systems with controlled burns is intricately tied to the disturbance history and presence of exotic species at the site. Controlled burning alone is not expected to be the most effective tool to manage these systems because resultant burns will be patchy and of insufficient intensity to remove invasive species from the system. The use of fire as a management tool in rock outcrop ecotones should be evaluated on a case-by-case basis where the consideration of rare species occurrence, site history, status of invasive exotics, and management goals are taken into account.

Competition can cause local extirpation, so there is concern that invasion of the ecotone of rock outcrops will modify the environmental character of the site and these invasive species will outcompete native flora for the precious few resources available in this stressed system. While invasion by exotic species is a valid concern, invasion rarely causes extinction (Elton, 1985; Sax et al., 2007). Daehler (2003) suggested that in resource-limited systems, native species will perform as well or better than their invasive counterparts. The invaded zones at Rock and Shoals and Panola Mountain had significantly higher invasive cover than control areas, confirming that invasion of some rock margins is patchy. Dominant invading species encountered in the surveys were *Ligustrum sinense* and *Lonicera japonica*. Sodium, calcium, and magnesium were all found to be significant predictors of invasive occurrence. Removing exotic invasives from treatment plots did not significantly alter species diversity or vegetation class distributions or abundance. Invasion correlated positively with distance from the rock edge. It is clear that active invasion is taking place at Rock and Shoals and Panola Mountain, but the biological repercussions of such change could not be documented in the time frame of this study. Invading exotics do not appear to be excluding native flora on a short time scale.

Several parallels can be drawn from the invasion and fire studies. Both studies demonstrated spatial distribution patterns, in which distance from the rock edge and soil depth correlated with vegetation class occurrence and woody species richness. Findings in the literature report that plant occurrence correlates with soil depth (Burbanck and Platt, 1964; Shure and Ragsdale, 1977; Burbanck and Phillips, 1983). The monitoring period may have been too short to capture effects of species recovery after invasive species were removed or transects were burned. Plant productivity, recruitment, or recovery that may have been stimulated by the study treatments also could have been compromised by the drought in 2007. Invasion and disturbance combined can be a recipe for disaster because invading species will use the light gaps and the resource pulse of disturbance as an opportunity for establishment. For management of invaded sites, DiTomaso et al. (2006) recommend an integrated strategy in which burning and herbicide applications are used in tandem, and the treatment times of these tools is considered a critical part of their effective use. Landscape managers should also attempt to remove invasives manually before and after burning takes place.

Pigott (2000) suggests that degraded outcrops will require intensive management, which includes fencing out livestock, fire management, and direct seeding of appropriate native species to restore pre-disturbance conditions. In the case of privet removal from southeastern landscapes, Hanula (2009) reinforces that opinion by stating that eradicating exotics from degraded sites requires a long-term commitment. The restoration potential of unmanaged areas invaded by *Ligustrum* spp. is considered low (Batcher, 2000). Kirkpatrick (1986) remains optimistic, however, suggesting that hand removal of serious environmental weeds can result in successful long-term control and improve recruitment of native species. Invasive removal efforts often produce disturbance, which ultimately facilitates the spread of other aggressive exotics (Webb, Iv, and Dwyer, 2001; Hanula, 2009). Restoration to pre-invasion communities will require future intervention or removal of all propagules of invasive exotics.

While changes in plant community structure on short-term, local scales were not observed in this study, regional spatial analysis confirmed that these communities are experiencing a shift in landcover associated with increased urban landcover and impervious surfaces. This is probably tied to suburban development of neighboring communities of Atlanta. At the turn of the next century, land use change is projected to have the largest global impact on biodiversity, followed by climate change, nitrogen deposition, species introductions, and changing concentrations of atmospheric CO_2 (Chapin et al., 2000; Sala et al., 2000).

The spatial distribution of outcrops has been compared to that of island archipelagoes in a sea of mesic forest (Wyatt and Allison, 2000). Local extinctions of a single patch are not expected to be restored by natural immigration on “reasonable” time scales (Loehle, 2006). These communities require change mediated over geologic time to achieve successful metapopulation dynamics, in which the immigration and emigration responsible for connecting isolated parts of the whole require longer time scales to play a part in recolonization events and replenish available habitat islands. The endemic species specialized to the unique conditions of the rock outcrop habitat have evolved adaptations that limit long-distance dispersal to discourage propagule mortality in the surrounding forest matrix (Wyatt, 1997). In this analysis, latitude, elevation, and outcrop spatial complexity were predictors of rare plant occurrence alone. These findings suggest that habitat complexity may be more important than previously recognized. Isolation and rock outcrop size were not found to be significant predictors of rare plant occurrence.

Future studies of rock outcrops in the southeastern United States could focus on a landscape level analysis of species richness. This information can also be combined with contemporary surveys and biological data to help identify conservation and management priorities and may very likely find different predictive variables than those that were significant to rare plant occurrence. Population isolation could be used as a potential predictor to explore further the idea that isolated outcrop populations are more vulnerable to extinction. These systems can also be used to understand better faunal invasion, particularly fire ants, and how modification of faunal and soil properties could be major contributing factors to habitat change.

Climate warming in the Southeast could cause a northward shift of outcrop populations, but suitable habitat doesn’t exist north of Virginia. Moreover, the present rate of climate change is too rapid to allow outcrop flora to disperse to new locations without assistance. Species extinctions in island systems have taught us that those species with specialized habitat requirements are often the most vulnerable to extinction threats (Ricklefs and Cox, 1972; Brown, 1995). Current rates of extinction are estimated to be 100–1,000 times greater than pre-human rates, and small endemic-rich areas are expected to contribute disproportionately to the total number of extinctions (Pimm et al., 1995; Chapin et al., 2000; Loreau et al., 2001). Populations that attempt to shift their geographic ranges to match habitat requirements via long-distance dispersal events are not expected to be able to keep pace with current climate trends (McLachlan, Clark, and Manos, 2005; Loehle, 2006). A climate shift, however, could favor, rather than condemn, these systems. Disjunct species have a rich history of claiming outcrop habitat as a home (Murdy, 1968). Rock outcrops, with their suite of species adapted to environmental extremes, could serve as a biological repository of flora for revegetating temperate regions of North America if human disturbance doesn’t destroy them first (Harper, 1981; Chapin et al., 2000). The possible effects of climate change among inselberg floras could also provide early warning indicators for climate change (Burke, 2003).

The stewardship of these unique and floristicly rich environments is our responsibility. Rock outcrops operate as disparate parts of a greater whole. Thus, we must make an attempt to protect individual outcrops because they have a much greater chance of representing a unique genetic component of the overall species composition.

Local extinctions appear imminent and unavoidable, especially on small outcrops close to Atlanta where development pressure is greatest. Because rock outcrops typically have low property value, however, they are reasonable targets for protection. We must insist that additional outcrop habitat be secured for protection and make every effort to minimize disturbance on existing habitats. When habitat destruction is imminent, every effort must be made to safeguard rare flora until suitable restoration habitat can be found. Some of these plant species may have the genetic variation needed to cope with future climate extremes, and it is important that we preserve what diversity is left.

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Appendix A: Typical Granite Outcrop Ecotone Species

Site: Heggie's Rock

Species		**Class**^{a}	**Plot**					**Level of Endemism**^{b}
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Agrostis hyemalis	winter bentgrass	G	X					
Andropogon virginicus	broomsedge	G	X	X	X	X	X	
Briza minor	little quakinggrass	G	X					
Carex sp.	sedge	G	X	X	X	X	X	
Chasmanthium sessiliflorum	woodoats	G		X	X	X	X	
Danthonia spicata	poverty oatgrass	G		X	X	X	X	
Dichanthelium boscii	Bosc's panicgrass	G			X	X	X	
Dichanthelium dichotomum	cypress panicgrass	G	X	X	X			
Dichanthelium laxiflorum	openflower rosette grass	G	X	X	X	X	X	
Juncus tenuis	poverty rush	G	X	X	X			
Melica mutica	twoflower melicgrass	G		X	X	X	X	
Piptochaetium avenaceum	blackseed speargrass	G					X	
Saccharum sp.	sugarcane plume grass	G		X				
Scleria sp.	nutrush	G		X	X	X	X	
Phacelia maculata	spotted phacelia	HF		X				strict endemic
Diamorpha smallii	elf orpine	HF	X					near endemic
Helianthus porteri	Porter's sunflower	HF	X	X				near endemic
Isoetes piedmontana	Peidmont quillwort	HF	X					near endemic
Lindernia monticola	piedmont false pimpernel	HF		X				near endemic
Minuartia uniflora	oneflower stitchwort	HF	X					near endemic
Oenothera fruticosa L. ssp. fruticosa	sundrops	HF		X				near endemic
Croton willdenowii	Willdenow's croton	HF	X					half endemic
Packera tomentosa	woolly ragwort	HF	X	X	X	X	X	half endemic
Phemeranthus teretifolius	quill fameflower	HF	X					half endemic
Schoenolirion croceum	yellow sunnybell	HF	X					half endemic
Tradescantia hirsuticaulis	hairystem spiderwort	HF		X	X		X	half endemic
Cheilanthes lanosa	hairy lipfern	HF	X	X	X			prov. endemic
Opuntia humifusa	prickly pear	HF	X			X		prov. endemic
Rhexia mariana	Maryland meadowbeauty	HF					X	prov. endemic
Selaginella rupestris	northern selaginella	HF	X	X				prov. endemic
Spiranthes cernua	nodding lady's tresses	HF				X		prov. endemic

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Heggie's Rock

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Agalinis tenuifolia	slenderleaf false foxglove	HF		X				
Asplenium platyneuron	ebony spleenwort	HF	X	X	X	X	X	
Chaerophyllum tainturieri	hairyfruit chervil	HF		X	X			
Desmodium nudiflorum	nakedflower ticktrefoil	HF				X	X	
Euphorbia pubentissima	false flowering spurge	HF			X	X	X	
Houstonia pusilla	tiny bluet	HF	X					
Hypericum gentianoides	pineweed	HF	X					
Krigia virginica	dwarfdandelion	HF	X	X				
Nothoscordum bivalve	crowpoison	HF	X	X	X			
Nuttallanthus canadensis	Canada toadflax	HF	X	X				
Oxalis dillenii	slender yellow woodsorrel	HF	X	X	X	X		
Oxalis violacea	violet woodsorrel	HF		X	X			
Packera anonyma	Small's ragwort	HF	X	X	X	X		
Plantago virginica	Virginia plantain	HF	X					
Polypodium polypodioides	resurrection fern	HF		X	X			
Ruellia caroliniensis	Carolina wild petunia	HF				X	X	
Scutellaria ovata	heartleaf skullcap	HF				X	X	
Selaginella apoda	meadow spike moss	HF		X				
Solidago nemoralis	gray goldenrod	HF		X	X			
Triodanis perfoliata	Venus' looking-glass	HF		X				
Yucca filamentosa	Adam's needle	HF			X		X	
Ligustrum sinense	Chinese privet	I			X			
Lonicera japonica	Japanese honeysuckle	I	X	X	X	X	X	
Murdannia keisak	wartremoving herb	I	X					
Cladonia caroliniana	Carolina cup lichen	L	X					
Cladonia rangiferina	greygreen reindeer lichen	L	X					
Cladonia subtenuis	reindeer lichen	L	X	X				
Cocculus carolinus	Carolina coralbead	L					X	
Grimmia laevigata	grimmia dry rock moss	M	X					
Hedwigia ciliata	ciliate hedwigia moss	M	X	X				
Leucobryum glaucum	leucobryum moss	M	X					
Philonotis sp.	philonotis moss	M	X	X	X			
Polytrichum sp.	polytrichum moss	M	X	X	X			
Sphagnum sp.	sphagnum moss	M	X	X	X			
Thuidium delicatulum	delicate thuidium moss	M	X	X	X	X	X	
Sedum pusillum	granite stonecrop	R	X	X				strict endemic
Anemone berlandieri	tenpetal thimbleweed	R		X	X	X		half endemic

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Heggie's Rock

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Parthenocissus quinquefolia	Virginia creeper	V	X	X	X	X	X	
Berchemia scandens	Alabama supplejack	V			X			
Gelsemium sempervirens	Carolina yellow jessamine	V		X	X	X	X	
Lonicera sempervirens	coral honeysuckle	V				X	X	
Rubus sp.	blackberry	V				X		
Smilax bona-nox	saw greenbrier	V	X	X	X	X	X	
Smilax smallii	lanceleaf greenbrier	V					X	
Toxicodendron radicans	eastern poison ivy	V			X	X	X	
Vitis rotundifolia	muscadine	V		X	X	X	X	
Forestiera ligustrina	upland swampprivet	W		X	X	X	X	prov. endemic
Callicarpa americana	American beautyberry	W		X	X	X	X	
Carya glabra	pignut hickory	W	X	X	X	X	X	
Celtis tennuifolia	dwarf hackberry	W			X	X	X	
Chionanthus virginicus	white fringetree	W			X	X		
Diospyros virginiana	common persimmon	W		X				
Frangula caroliniana	Carolina buckthorn	W		X	X	X	X	
Juniperus virginiana	eastern redcedar	W	X	X	X	X	X	
Pinus taeda	loblolly pine	W	X	X	X	X	X	
Quercus nigra	water oak	W			X	X	X	
Quercus stellata	post oak	W		X		X		
Rhus copallinum	winged sumac	W				X	X	
Ulmus alata	winged elm	W		X	X	X	X	
Vaccinium arboreum	sparkleberry	W		X	X	X	X	
Vaccinium stamineum	deerberry	W					X	
Viburnum rufidulum	rusty blackhaw	W				X		

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Camp Meeting Rock

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Andropogon virginicus	broomsedge	G		X				
Carex lurida	shallow sedge	G				X		
Carex sp.	sedge	G	X	X	X	X	X	
Chasmanthium sessiliflorum	longleaf woodoats	G	X	X	X	X	X	
Danthonia sericea	downy danthonia	G					X	
Dichanthelium boscii	Bosc's panicgrass	G				X	X	
Dichanthelium dichotomum	cypress panicgrass	G			X	X		
Dichanthelium laxiflorum	openflower rosette grass	G		X	X	X	X	
Piptochaetium avenaceum	blackseed needlegrass	G	X	X	X	X		
Helianthus porteri	Porter's sunflower	HF	X					near endemic
Croton willdenowii	Willdenow's croton	HF	X					half endemic
Iris verna	dwarf violet iris	HF				X		half endemic
Aletris farinosa	white colicroot	HF			X			prov. endemic
Rhexia mariana	Maryland meadowbeauty	HF			X			prov. endemic
Chimaphila umbellata	pipsissewa	HF				X		
Euphorbia pubentissima	false flowering spurge	HF					X	
Hexastylis arifolia	little brown jug	HF				X		
Hypoxis hirsuta	yellow star grass	HF				X		
Liatris microcephala	dwarf blazingstar	HF	X					
Mitchella repens	partridge berry	HF	X	X	X			
Cladonia pyxidata	cup lichen	L	X					
Parmelia sp.	shield lichen	L	X					
Aulacomnium palustre	aulacomnium moss	M	X	X				
Grimmia laevigata	grimmia dry rock moss	M	X					
Hedwigia ciliata	ciliate hedwigia moss	M	X					
Leucobryum glaucum	leucobryum moss	M	X	X	X	X	X	
Polytrichum sp.	polytrichum moss	M	X	X	X	X		
Sphagnum sp.	sphagnum moss	M	X	X	X	X	X	
Thuidium delicatulum	delicate thuidium moss	M				X		
Bignonia capreolata	crossvine	V	X		X			
Gelsemium sempervirens	Carolina yellow jessamine	V		X	X			
Rubus sp.	blackberry	V			X		X	
Smilax bona-nox	saw greenbrier	V	X	X	X	X	X	
Smilax smallii	lanceleaf greenbrier	V				X		
Vitis rotundifolia	muscadine	V	X	X	X	X	X	

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Camp Meeting Rock

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Acer rubrum	red maple	W	X	X	X	X	X	
Amelanchier arborea	serviceberry	W		X				
Asimina triloba	pawpaw	W		X	X		X	
Calycanthus floridus	sweetshrub	W		X	X	X	X	
Carya glabra	pignut hickory	W			X	X	X	
Fraxinus pennsylvanica	green ash	W					X	
Hypericum hypericoides	St. Andrew's cross	W		X				
Liquidambar styraciflua	sweetgum	W	X	X	X	X	X	
Liriodendron tulipifera	tulip poplar	W			X	X	X	
Nyssa sylvatica	black gum	W	X	X	X	X	X	
Oxydendrum arboreum	sourwood	W	X	X	X	X		
Photinia pyrifolia	red chokecherry	W		X	X	X		
Pinus taeda	loblolly pine	W	X	X	X	X	X	
Quercus alba	white oak	W		X	X	X	X	
Quercus nigra	water oak	W	X	X	X	X	X	
Quercus prinus	chestnut oak	W			X	X	X	
Quercus rubra	red oak	W		X	X	X	X	
Quercus stellata	post oak	W			X	X		
Quercus velutina	black oak	W			X	X		
Rhododendron canescens	Piedmont azalea	W	X	X	X	X	X	
Vaccinium arboreum	sparkleberry	W	X	X	X	X	X	
Vaccinium pallidum	lowbush blueberry	W	X	X	X	X	X	
Vaccinium stamineum	deerberry	W		X	X	X		
Viburnum acerifolium	mapleleaf viburnum	W			X		X	

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Panola Mountain

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Agrostis hyemalis	winter bentgrass	G	X					
Andropogon virginicus	broomsedge	G	X	X	X	X		
Aristida dichtoma	three awn grass	G	X					
Carex sp.	sedge	G	X	X	X	X	X	
Chasmanthium sessiliflorum	longleaf woodoats	G	X	X	X	X	X	
Danthonia sericea	silky oatgrass	G	X	X	X	X	X	
Danthonia spicata	poverty oatgrass	G	X	X	X	X	X	
Dichanthelium boscii	Bosc's panicgrass	G	X	X	X	X	X	
Dichanthelium laxiflorum	openflower rosette grass	G	X	X	X	X	X	
Eragrostis sp.	love grass	G	X					
Muhlenbergia schreberi	muhly grass	G	X					
Saccharum sp.	plumegrass	G					X	
Scleria sp.	nutrush	G		X	X	X	X	
Tridens flavus	purpletop	G					X	
Croton willdenowii	Willdenow's croton	HF	X					half-endemic
Liatris microcephala	scaly blazingstar	HF		X	X	X		half-endemic
Schoenolirion croceum	yellow sunnybell	HF	X					half-endemic
Tradescantia hirsuticaulis	hairystem spiderwort	HF	X					half-endemic
Oenothera fruticosa L. ssp. fruticosa	narrowleaf evening-primrose	HF	X					near-endemic
Helianthus porteri	Porter's sunflower	HF	X					near-endemic
Manfreda virginica	false aloe	HF	X					prov. endemic
Opuntia humifusa	prickly pear	HF		X				prov. endemic
Agalinis tenuifolia	slenderleaf false foxglove	HF			X	X		
Aquilegia canadensis	wild red columnine	HF		X				
Asplenium platyneuron	ebony spleenwort	HF	X	X	X	X	X	
Botrychium sp.	grapefern	HF		X		X		
Chimaphila umbellata	pipsissewa	HF			X			
Commelina erecta	whitemouth dayflower	HF			X			
Desmodium sp.	beggars ticks	HF		X	X			
Euphorbia sp.	flowering spurge	HF		X	X			
Galium pilosum	hairy bedstraw	HF		X	X			
Heuchera americana	alumroot	HF		X	X			

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Panola Mountain

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Hypericum gentianoides	pineweed	HF	X					
Hypericum punctatum	spotted St. John's wort	HF	X		X	X		
Krigia virginica	Virginia dwarfdandelion	HF	X	X				
Lespedeza procumbens	trailing bushclover	HF	X	X	X	X	X	
Lespedeza sp.	bushclover	HF		X	X	X		
Nothoscordum bivalve	crowpoison	HF	X	X			X	
Nuttallanthus canadensis	Canada toadflax	HF	X					
Oenothera biennis	evening primrose	HF	X					
Oxalis dillenii	slender yellow woodsorrel	HF	X	X	X	X		
Oxalis violacea	violet woodsorrel	HF		X	X			
Packera anonyma	Small's ragwort	HF	X	X	X	X		
Pityopsis graminifolia	narrowleaf silkgrass	HF				X		
Plantago virginica	Virginia plantain	HF	X					
Pleopeltis polypodiodes	resurrection fern	HF		X		X		
Potentilla canadensis	dwarf cinquefoil	HF		X	X	X	X	
Potentilla canadensis	dwarf cinquefoil	HF		X	X	X	X	
Solidago nemoralis	gray goldenrod	HF		X	X	X		
Stellaria media	chickweed	HF		X				
Tipularia discolor	crippled cranefly	HF					X	
Triodanis perfoliata	clasping Venus' looking-glass	HF		X				
Viola bicolor	field pansy	HF			X			
Yucca filamentosa	Adam's needle	HF		X	X	X	X	
Zephyranthes atamasca	atamasco lily	HF		X				
Elaeagnus umbellata	autumn olive	I				X		
Ligustrum sinense	Chinese privet	I	X	X	X	X	X	
Lonicera japonica	Japanese honeysuckle	I	X	X	X	X	X	
Rosa multiflora	multiflora rose	I		X				
Cladonia caroliniana	Carolina cup lichen	L	X					
Cladonia cristatella	British soldier	L	X	X	X	X	X	
Cladonia pyxidata	cup lichen	L	X	X				
Cladonia rangiferina	greygreen reindeer lichen	L	X					
Cladonia strepsilis	cup lichen	L	X	X				
Cladonia subtenuis	reindeer lichen	L	X	X	X			
Dermatocarpon sp.	earth and silverskin lichens	L	X		X	X	X	
Xanthoparmeilia sp.	scabrosa lichen	L	X		X		X	

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Panola Mountain

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Aulacomnium palustre	aulacomnium moss	M	X	X	X	X	X	
Grimmia laevigata	grimmia dry rock moss	M	X					
Leucobryum glaucum	leucobryum moss	M	X	X	X	X	X	
Philonotis sp.	philonotis moss	M	X	X	X	X		
Polytrichum sp.	polytrichum moss	M	X	X	X	X	X	
Sphagnum sp.	sphagnum moss	M		X	X	X	X	
Thuidium delicatulum	delicate thuidium moss	M	X	X			X	
Bignonia capreolata	crossvine	V		X	X		X	
Cocculus carolinus	Carolina coralbead	V			X			
Gelsimium sempervirens	Carolina yellow jessamine	V	X	X	X	X	X	
Parthenocissus quinquefolia	Virginia creeper	V	X	X	X	X	X	
Rubus sp.	Blackberry	V	X	X	X	X	X	
Smilax bona-nox	saw greenbrier	V	X	X	X	X	X	
Toxicodendron radicans	eastern poison ivy	V					X	
Vitis rotundifolia	muscadine	V	X	X	X	X	X	
Forestiera ligustrina	upland swampprivet	W		X	X	X	X	prov. endemic
Acer rubrum	red maple	W	X					
Amelanchier arborea	serviceberry	W		X	X	X	X	
Callicarpa americana	American beautyberry	W	X	X	X	X	X	
Carya glabra	pignut hickory	W	X	X	X	X	X	
Celtis occidentalis	common hackberry	W		X	X			
Chionanthus virginicus	white fringetree	W		X	X	X	X	
Crataegus sp.	hawthorn	W		X	X	X		
Euonymus americanus	Strawberry Bush	W			X			
Fraxinus pennsylvanica	green ash	W	X	X	X	X	X	
Juniperus virginiana	eastern redcedar	W	X	X	X	X	X	
Liquidambar styraciflua	sweetgum	W				X	X	
Philadelphus inodorus	scentless mock orange	W		X	X	X	X	
Pinus taeda	loblolly pine	W	X	X	X	X	X	
Prunus serotina	black cherry	W		X	X	X	X	
Quercus alba	white oak	W				X	X	
Quercus nigra	water oak	W	X	X	X	X	X	
Quercus stellata	post oak	W			X	X	X	
Quercus velutina	black oak	W		X	X	X	X	
Rhus aromatica	fragrant sumac	W		X	X	X	X	
Rhus copallinum	winged sumac	W		X	X			

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Panola Mountain

Species		**Class**[a]	**Plot**					**Level of Endemism**[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Symphoricarpos orbiculatus	coralberry	W		X	X	X	X	
Ulmus alata	winged elm	W	X	X	X	X	X	
Vaccinium arboreum	sparkleberry	W	X	X	X	X	X	
Vaccinium stamineum	deerberry	W			X			
Viburnum sp.	viburnum	W		X			X	

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Rock and Shoals

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Juncus georgianus	Georgia rush	G	X	X				strict endemic
Cyperus granitophilus	granite flatsedge	G		X				near-endemic
Andropogon ternarius	splitbeard bluestem	G	X	X				
Andropogon virginicus	broomsedge	G	X	X	X	X	X	
Carex lurida	shallow sedge	G	X	X		X		
Carex sp.	sedge	G	X	X	X	X		
Chasmanthium sessiliflorum	longleaf woodoats	G		X	X	X	X	
Danthonia sericea	silky oatgrass	G	X	X	X	X	X	
Danthonia spicata	poverty oatgrass	G		X	X	X	X	
Dichanthelium boscii	Bosc's panicgrass	G	X	X	X	X	X	
Dichanthelium dichotomum	cypress panicgrass	G	X	X	X	X		
Dichanthelium laxiflorum	openflower rosette grass	G	X	X	X	X	X	
Eragrostis sp.	love grass	G	X	X			X	
Melica mutica	twoflower melicgrass	G			X	X	X	
Panicum anceps	beaked panicgrass	G				X		
Panicum dichotomiflorum	fall panicgrass	G				X		
Piptochaetium avenaceum	blackseed speargrass	G						
Rhynchospora sp.	beaksedge	G	X	X				
Scleria sp.	nutrush	G		X	X	X	X	
Tridens flavus	purpletop	G				X		
Phacelia maculata	spotted phacelia	HF	X	X	X		X	strict endemic
Helianthus porteri	Porter's sunflower	HF	X	X	X	X	X	near-endemic
Isoetes piedmontana	Piedmont quillwort	HF	X					near-endemic
Oenothera fruticosa var. subglobosa	sundrops	HF	X					near-endemic
Croton willdenowii	Willdenow's croton	HF	X	X	X	X	X	half-endemic
Liatris microcephala	scaly blazingstar	HF					X	half-endemic
Schoenolirion croceum	yellow sunnybell	HF	X					half-endemic
Tradescantia hirsuticaulis	hairystem spiderwort	HF	X	X	X	X	X	half-endemic
Cheilanthes lanosa	hairy lipfern	HF	X					prov. endemic
Opuntia humifusa	prickly pear	HF	X	X				prov. endemic
Spiranthes cernua	nodding lady's tresses	HF		X	X	X	X	prov. endemic
Acalypha virginica	Virginia threeseed mercury	HF	X	X	X	X	X	
Ambrosia artemisiifolia	annual ragweed	HF			X			

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Rock and Shoals

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Antennaria plantaginifolia	plantain-leaf pussytoes	HF			X	X		
Asplenium platyneuron	ebony spleenwort	HF	X	X	X	X	X	
Belamcanda chinensis	blackberry lily	HF		X	X		X	
Botrychium biternatum	grape fern	HF			X			
Cardamine concatenata	cutleaf toothcup	HF			X			
Cardamine parviflora ssp. arenicola	sand bittercress	HF		X	X			
Carduus sp.	thistle	HF		X				
Chaerophyllum tainturieri	wild chervil	HF		X	X			
Commelina erecta	whitemouth dayflower	HF		X	X	X		
Coreopsis grandiflora	tickseed	HF	X	X	X	X	X	
Diodia teres	poorjoe	HF	X					
Galium pilosum	hairy bedstraw	HF		X	X	X	X	
Gamochaeta purpurea	spoonleaf purple everlasting	HF		X	X	X	X	
Geranium carolinianum	Carolina geranium	HF	X	X	X		X	
Heuchera americana	alumroot	HF		X	X			
Houstonia purpurea	woodland bluets	HF		X	X			
Houstonia pusilla	tiny bluet	HF	X	X	X		X	
Hypericum gentianoides	pineweed	HF	X					
Hypericum hypericoides	St. Andrew's cross	HF		X	X			
Krigia virginica	Virginia dwarfdandelion	HF	X	X	X	X	X	
Lespedeza angustifolia	narrowleaf lespedeza	HF		X		X	X	
Lespedeza repens	creeping lespedeza	HF		X	X	X	X	
Lespedeza sp.	bushclover	HF		X	X	X	X	
Leucanthemum vulgare	ox-eye daisy	HF			X			
Minuartia uniflora	oneflower stitchwort	HF	X					near-endemic
Nothoscordum bivalve	crowpoison	HF	X	X	X	X		
Nuttallanthus canadensis	Canada toadflax	HF	X	X			X	
Oxalis dillenii	slender yellow woodsorrel	HF	X	X	X	X	X	
Oxalis violacea	violet woodsorrel	HF		X	X			
Packera anonyma	Small's ragwort	HF	X	X	X	X	X	
Plantago virginica	Virginia plantain	HF	X		X		X	
Pleopeltis polypodiodes	resurrection fern	HF		X	X	X	X	
Potentilla canadensis	dwarf cinquefoil	HF	X	X	X	X	X	
Ruellia caroliniensis	Carolina wild petunia	HF		X	X	X	X	
Rumex hastatus	heartwing sorrel	HF		X				
Salvia lyrata	lyreleaf sage	HF		X	X	X	X	
Scutellaria ovata	heartleaf skullcap	HF			X			

Appendix A: Typical Granite Outcrop Ecotone Species

Site: Rock and Shoals

Species		Class[a]	Plot					Level of Endemism[b]
Scientific Name	**Common Name**		**1**	**6**	**10**	**16**	**20**	
Toxicodendron radicans	eastern poison ivy	V	X	X	X	X	X	
Vitis rotundifolia	muscadine	V		X	X	X	X	
Acer rubrum	red maple	W		X				
Callicarpa americana	American beautyberry	W		X	X	X	X	
Carya glabra	pignut hickory	W	X	X	X	X	X	
Celtis occidentalis	common hackberry	W		X	X	X	X	
Cercis canadensis	redbud	W		X	X	X	X	
Chionanthus virginicus	white fringetree	W			X			
Crataegus sp.	hawthorn	W			X	X	X	
Euonymus americanus	strawberry bush	W		X		X		
Fraxinus pennsylvanica	green ash	W	X	X	X	X	X	
Juniperus virginiana	eastern redcedar	W	X	X	X	X	X	
Lonicera maackii	bush honeysuckle	W			X		X	
Morus rubra	red mulberry	W				X		
Nandina domestica	heavenly bamboo	W			X			
Nyssa sylvatica	blackgum	W					X	
Pinus taeda	loblolly pine	W		X	X			
Prunus serotina	black cherry	W		X	X	X		
Quercus alba	white oak	W		X				
Quercus nigra	water oak	W	X	X	X	X	X	
Quercus stellata	post oak	W		X				
Quercus velutina	black oak	W		X		X	X	
Rhus copallinum	winged sumac	W		X				
Ulmus alata	winged elm	W	X	X	X	X	X	
Ulmus americana	American elm	W					X	
Vaccinium arboreum	sparkleberry	W	X	X	X	X	X	
Vaccinium elliottii	Elliott's blueberry	W				X		
Vaccinium stamineum	deerberry	W		X		X		
Viburnum sp.	viburnum	W					X	

[a] Vegetation classes are G for grasses, HF for herbaceous plants, ferns, and fern allies, I for invasive, L for lichens, M for mosses, V for vines, and W for woody plants.

[b] Level of endemism as determined by Wyatt and Allison, 2000

Species with an occurrence of five or more counts for the specified plot were reported in the table.