Assessing non-native invasive vegetation at reclaimed surface mine sites of the southern Shale Hills region of Alabama.

OSM cooperative agreement S09AC15438

Final Report

Reporting period - 8/1/09 to 12/1/11

Dawn Lemke, Yong Wang, Callie Schweitzer, Wubishet Tadesse, and Irenus Tazisong

Date of submission: January 2012

Alabama A&M University, 4900 Meridian Street, Normal, AL, 35761.
dawn.lemke@aamu.edu, 256 372 4562

Disclaimer

Abstract

Throughout the world, the invasion of non-native plants is an increasing threat to native biodiversity and economic productivity. Invasion is especially prevalent in areas affected by land transformation and anthropogenic disturbance. Surface mines are a major disturbance, and as such, may promote the establishment and expansion of invasive plant communities. Using the Shale Hills Region of Alabama as a case study, we examined the environmental and habitat factors that may contribute to favorable conditions for heightened plant invasion, and developed models for predicting the probability of occurrence of invasive plant species. . Mine lands were surveyed for all species defined by the United States Forest Service as invasive to the forest of the southern region. We conducted vegetation surveys, soil sampling, and environmental evaluation on the mined landscape. Canonical correspondence analysis suggested that the invasive community were predominantly associated with forest structure and composition. Chinese privet (*Ligustrum sinense*) and Japanese honeysuckle (*Lonicera japonica*) were positively associated with vegetation diversity, total canopy cover and hardwood density; autumn olive (*Elaeagnus umbellata)* and princesstree (*Paulownia tomentosa*) were positively associated with hardwood basal area. These parameters are features often associated with more established forests. Chinese lespedeza (*Lespedeza cuneata*) and shrubby lespedeza (*Lespedeza bicolor*) were negatively associated with the above characteristics. Logistic regressions with the three most common species, Chinese privet, Japanese honeysuckle and Chinese lespedeza, all had reasonable concordance (>75) and over 25% decrease in false omission rates and type II errors, suggesting useful models for predicting occurrence. Chinese lespedeza was more likely to be found in open or pine areas with higher magnesium levels in the soil and little or no midstory and downed woody debris. Japanese honeysuckle was found in high canopy cover with little midstory and in areas of high soil magnesium and higher diversity. Chinese privet had a strong positive relationship with canopy cover. At a landscape level invasive species occurrence data were assessed using logistic regression and maximum entropy modeling integrated with geographic information systems. We used an area under the curve value for the operator receiver characteristic of greater than 0.75 and decrease omission rate of more than 25% as defining a good model. The distance to forest had the highest overall contribution (19%) to the models, with three other variables contributing over 10%, distance to roads, Normalized Difference

Vegetation Index in 1987 and 2011. Species models were then applied to the mined landscape to assess the probable prevalence of each species across the landscape. Japanese honeysuckle had the highest probable prevalence at 48% (73% moderate potential), with princesstree having the lowest, at less than 1% (3% moderate potential). Overall 33% of the landscape is predicted to have no invasive plants, with 47% predicted to have one, 17% two and 3% to have three or more. All species we modeled, apart from princesstree and sawtooth oak, showed much higher occurrence on the reclaimed sites than that from across the broader region. We found that geospatial modeling of these invasive plants, at this scale, was useful and does offer potential for management, both in terms of identifying habitat types most and risk and identifying areas needing management attention. The accuracy of the predictive models and density of occurrence was probably affected by the planting of non-native, invasive species in this area.

Table of Figures

Figure 1: Percent of Forest Service survey plots within a county occupied by one to four invasive plants in the Southern region…………………………………… 8

Figure 2: Shale Hills study area location map…………………………………… 10

Figure 3: Relationship between habitat variables and the invasive community as assessed through Canonical Correspondence Analysis …………………. 24

Figure 4: Example of mapping distribution……………………………………… 34

Table of Tables

Table 1: Habitat variables measured at each sampling plot ………………………… 22

Table 2: Summary statistics of three invasive species from three logistic regression submodels (soil, ground and forest), combined models and final models.… 27

Table 3: Summary of resampling of final logistic model for 100 observations run 1000 times.……………………………………………………………….. 27

Table 4: Summary of significant variables for three invasive species from three logistic regression submodels (soil, ground and forest), combined models and final model with only variables that remain stables over resampling…. 28

Table 5: Summary of geospatial variables measured at each sampling site………... 29

Table 6: Summary statistics of six invasive species from logistic regression and MaxEnt models from 100 resample's…………..…………………………. 33

Table 7: Probable proportion of mined landscape invaded………………………… 33

Table 8: Correlation between species ……………………………………………… 34

Introduction

Land transformation and anthropogenic disturbance often facilitate the establishment and development of invasive plant community. Surface mining is one of the major forms of disturbance and has changed over 2.4 million hectares of terrestrial habitat in the United States since 1930 (Zeleznik & Skousen 1996). The changes include land transformation, alteration in ecosystems and geophysical characteristics (Holl 2002; McSweeney & Jansen 1984; Negley & Eshleman 2006; Shukla, Lal, & Ebinger 2005). Consequential impacts include interruption and change of energy flow, food webs, biodiversity, successional patterns, and biogeochemical cycling (Ripley, Redman, & Crowder 1996). Surface mining is distinct from most other land disturbances in that the disturbance is comprehensive, with native vegetation, soils, soil microbes, and seed banks being removed.

Since the introduction of the Surface Mining Control and Reclamation Act (SMCRA) in 1977, much of the land transformed by surface mining in the United States has been subjected to some reclamation, with efforts aimed at improving the quality of the land by restoring some of the pre-disturbance vegetation and functions (Bradshaw 1984). The SMCRA mandates that mined land be reclaimed and restored to its original use or a use of higher value. However, surface mine reclamation efforts rarely result in ecosystems that simulate pre-mined characteristics. In the past the focus on edaphic and hydrologic systems has not been conducive to achieving goals related to the recovery of the pre-mining biological communities or mitigation of landscape structure and ecological change (Burger, Scott, & Mitchem 2002). However, SMCRA states that mining operations shall establish "*a diverse, effective, and permanent vegetative cover of the same seasonal variety and native to the area and capable of self-regeneration and plant succession …, unless use of alien species is necessary to achieve the stated post-mining land us*", and industry goals of reclamation are changing (Zipper *et al.* 2011). Still, most legislation mandates evaluation of land reclamation success after a relatively short time period, if at all (Holl & Cairns 2002). This encourages reclamation approaches that address the short-term concerns of providing erosion control and minimizing acid mine drainage, but not long-term concerns of restoration of ecosystem services. For example, the success of coal surface mine reclamation efforts in the south eastern United States is usually evaluated within five years (McElfish & Beier 1990). Such practices have resulted in large patches of grassland not typical of the native landscape (Hardt & Forman 1989). It has been suggested that goals for short-term and long-term recovery of highly disturbed sites may conflict (Holl 2002). Planting aggressive non-native ground cover species to minimize short-term erosion may slow long-term recovery. Recent work (Zipper *et al.* 2011) has shown the cost of historically used restoration methods are comparable with those that develop a more diverse and ecologically sound community. However, the legacy of non-native plants remains, even though there is a transition to a more ecologically stable restoration approach.

Throughout the world, non-native plants are becoming an increasing threat to native biodiversity and ecosystem functions (Ricciardi 2007; Vitousek et al. 1997). Historically and still to some extent today, non-native species are used in reclamation, to stabilize land and quickly develop a vegetation community. In disturbed systems such as mined areas, non-native invasive plants can be a significant management concern reducing ecosystem services. Invasive plants can change ecosystem services and influence the long-term ecological and economic productivity of land (Webster, Jenkins, & Jose 2006). Invasiveness (traits that enable a species to invade a new habitat) and invasibility (the susceptibility of a community or habitat to the establishment and spread of new species) are key components for the occurrence and spread of non-native plants (Alpert, Bone, & Holzapfel 2000). The characteristics of plants that assist in some of the short-term goals of restoration, including land stabilization and nitrogen fixing are often the same traits that are associated with invasive plants. Some of the traits reclamationists favor in their choice of plants, including fast establishment, the ability to grow under harsh conditions, and adaptation to nutrient-poor soils also relate to invasive tendencies. Habitat attributes that are associated with invasibility are disturbance, early successional environments, low diversity of native species (Lodge 1993), and high environmental stress (Alpert et al. 2000; D’Antonio, Dudley, & Mack 1999; Skousen, Johnson, & Garbutt 1994). Mined sites often display these attributes and thus may have a high probability of being invaded by unwanted species.

Thus understanding the distribution across the landscape is an important component of invasive plant management and for evaluating the incidence of, and the potential for invasion (Davis, Grime, & Thompson 2000). Ground-based assessments are costly but provide essential information. Techniques such as remote sensing offer significant opportunities for providing information on invasions of non-native species and can be used to assess the broader vegetation characteristics. Remote sensing affords the opportunity to map an entire region and allow the delineation of species in less accessible areas. Imaging techniques can also offer financial advantages over field-based approaches, and accurately delineating the spatial distribution of invasives and combining maps with other environmental and anthropogenic data layers can provide the basis for predicting expansion into surrounding areas.

In the southern region of the United States, the counties with the highest diversity of invasive plants occur in the Southern Piedmont, Interior Low Plateau, and Southern Ridge and Valley of the Appalachian-Cumberland highlands (Miller, Lemke, & Coulston 2011). The same report showed that highest density of survey points with invasive plants was in the top half of Alabama (Figure 1). This area has had a long history of habitation and highly disturbed mining regions.

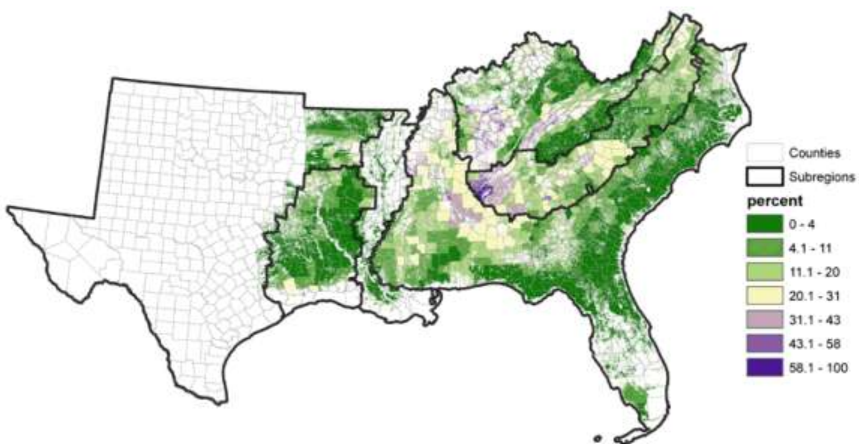

Figure 1: Percent of survey plots within a county occupied by one to four invasive plants, 2010 (Source: Forest Inventory and Analysis, Southern Research Station, U.S. Forest Service) from (Miller, Lemke, & Coulston 2011).

To assess the potential drivers of this high density we investigated the occurrence of invasive plant species in the Shale Hills Region (SHR) in mid-Alabama, quantified habitat and environmental conditions, examined the associations of invasive community and habitat and ecological characteristics, and developed predictive models for the occurrence of invasive species. We also assess the uses of remotely sensed and other geospatial datasets to develop non-field based models for assessment of non-native invasive plants on reclaimed mines. Both traditional statistics and machine- learning techniques are used to model invasive probabilities across the mined landscape of the SHR.

Executive Summary

The goals of this research program were to assess the degree of invasion by nonnative plant species on reclaimed mines, examine the relationship between landscape and habitat features with the probability of invasion by nonnative species and develop models for predicting the probability of invasive potential. These goals fall under the overarching goal to identify management strategies that will assist in minimizing the invasion of non-native species. We surveyed mined lands for all species defined by the United States Forest Service as invasive to the forest of the southern region.

During summer and fall of 2010 we sampled 372 intensive habitat plots, and in the spring of 2011 we sampled 36 groups (transects) of low intensity plots for a total of 4,644 plots for herbs, vines and forbs and 23,220 plots for trees and shrubs. In habitat analysis we focused on the six most prevalent species: shrubby lespedeza (*Lespedeza bicolor*) (found at 20 sites (n=20)), Chinese lespedeza (*Lespedeza cuneata*) (n=300), Japanese honeysuckle (*Lonicera japonica*) (n=217), Chinese privet (*Ligustrum sinense*) (n=68), autumn olive (*Elaeagnus umbellata*) (n=29), and princesstree (*Paulownia tomentosa*) (n=22). For landscape analysis we focused on species with more than 50 occurrences across the sampled landscape and included six species: Chinese lespedeza (n=2475, 53%), Japanese honeysuckle (n=1403, 30%), Chinese privet (n=238, 1%), autumn olive (n=436, 2%), princesstree (n=126 0.5%) and sawtooth oak (*Quercus accitimus*) (n=62 <0.5%).

The invasive community was most strongly associated with the habitat characteristics plant diversity, canopy cover, forest age and basal area suggesting that the long-term management of reclaimed mines may have the greatest impact on reducing preferential habitat for invasive plants. The majority of the invasive species were found in older, larger and more established forests (15 + years) that had higher tree diversity and where the invasive species would have had more time to establish. The managed monoculture pine plantations and open areas were less likely to have multiple invasive plants. Four of the invasive species were strongly associated in the community analysis, Chinese privet, autumn olive, princesstree, and Japanese honeysuckle, suggesting similar habitat preferences.

Geospatial modeling of these invasive plants is useful and offers potential for management, both in terms of identifying habitat types most at risk and areas that need management attention. Landscape analysis showed all modeled species, except princesstree and sawtooth oak, had a higher occurrence than the broader landscape, suggesting reclaimed mining areas are more vulnerable to invasive plant species.

The influence of planting non-native, invasive species in this area is likely the major driver of the high diversity of invasive plants, with four of the seven dominant species being planted. Adjusting the reclamation plantings to native species would aid in resolving this. To determine the impact that invasive species are having on reclamation areas and the impact to the productively of the land, further study needs to be undertaken. Of the three most dominant species, one is planted (Chinese lespedeza) and one is ubiquitous throughout the region at low densities (Japanese honeysuckle). The third species, privet, is of most concern. Overall, it appears that the many initial reclamation efforts, apart from the plantings, are not the major drivers impacting non-native, invasive species composition of the reclaimed, now forested mine site.

Experimental

Study Area

Our study was conducted in the Shale Hills sub-region (SHR) of the southern Cumberland Plateau of the south-eastern United States (Figure 2). The southern Cumberland Plateau has a temperate climate characterized by long, moderately hot summers and short, mild winters (Smalley 1979). The average minimum winter temperature is 1 °C, and the average summer maximum temperature is 32 °C (Smalley 1979). Annual precipitation averages about 1400 mm and is fairly well distributed throughout the year (Smalley 1979). Precipitation is greatest from January through April and least from August through November (Smalley 1979). Thunderstorms with high intensity rainfall are common in the summer (Smalley 1979). The forests of the Cumberland Plateau are among the most diverse of the world's temperate-zone forests (Ricketts *et al.* 1999). Like much of the forests in the eastern United States, the native deciduous hardwood and mixed pine hardwood ecosystems of the Cumberland Plateau have undergone a long history of land-use change (McGrath et al. 2004; Wear & Greis 2002), including surface mining, that have altered the landscape and ecosystem functions. The SHR comprises the southern extremity of the Cumberland Plateau. Topography is rugged and fairly complex. Because ridge tops are much lower than those in northern sections of the Plateau, the characteristics of the sub-region is of extensive hills, not mountains or a plateau. Strongly sloping land predominates, and the area is mostly forested. In this area, dissection has largely removed the parent soil's sandstone cap and exposed the underlying shale. Coal mining, both shaft and strip, is a major industry (Smalley 1979). Our target area included surface mines permitted after 1983, on both public and private lands. The mines were closed before 2006, thus had time to be reclaimed and for vegetation to re-establish.

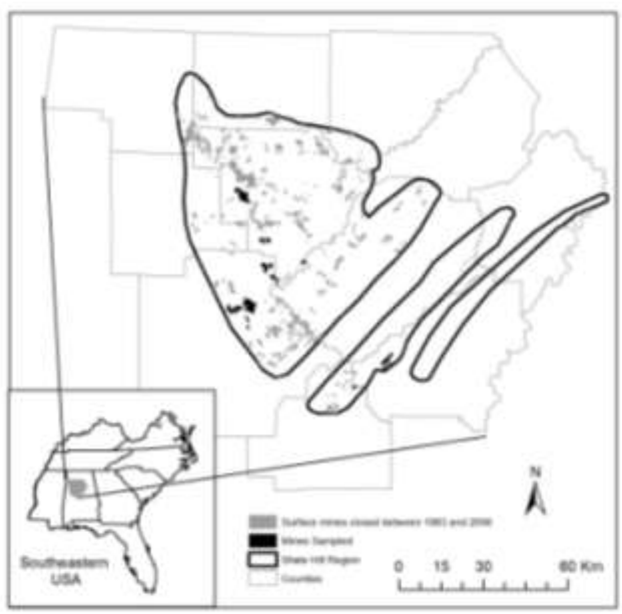

Figure 2: Shale Hills study area location map

Species of Interest

Mine lands were surveyed for all species defined by the United States Forest Service (USFS) as invasive to the forest of the southern region (Miller, Chambliss, & Loewenstein 2010). In plot analysis we focused on the six most prevalent species: shrubby lespedeza (*Lespedeza bicolor*) (found at 20 sites (n=20)), Chinese lespedeza (*Lespedeza cuneata*) (n=300), Japanese honeysuckle (*Lonicera japonica*) (n=217), *Chinese privet* (*Ligustrum sinense*) (n=68), autumn olive (*Elaeagnus umbellata)* (n=29), and princesstree (*Paulownia tomentosa*) (n=22). For landscape analysis we focuses on species with more than 50 occurrences across the sampled landscape, and included six species: Chinese lespedeza (*Lespedeza cuneata*) (n=2475, 53%), Japanese honeysuckle (*Lonicera japonica*) (n=1403, 30%), *Chinese privet* (*Ligustrum* sinense) (n=238, 1%), autumn olive (*Elaeagnus umbellata) (n=436, 2%)*, princesstree (*Paulownia tomentosa*) (n=126 0.5%) and sawtooth oak (*Quercus accitimus*) (n=62 <0.5%). Following is brief descriptions of each of these seven species.

Shrubby lespedeza (*Lespedeza bicolor*)

Shrubby lespedeza was introduced from Japan in the 1800s as an ornamental. It has been planted for wildlife habitat (Davison 1945; Haugen & Fitch 1955), and is also used in strip mine reclamation and along field borders (Graham 1941). It can reach three meters in height (Evans *et al.* 2006) and grows well in open areas, particularly on well-drained and acidic soils (Sun *et al.* 2008) The species is a serious invader in the southern region of the United States and is found in 27 states (PLANTS 2011) throughout the country. Shrubby lespedeza has been planted as part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, ASMC).

Chinese lespedeza (*Lespedeza cuneata*)

Introduced from Japan in 1899, Chinese lespedeza, also called Sericea lespedeza, is a long, slender perennial legume that can grow up to two meter tall. The species has spread quickly due to its use in pasture and erosion control (Miller et al. 2010), along roadways, on reclaimed mines, and along field borders (Graham 1941). It is flood tolerant and can survive in a wide variety of habitats, including forests, road sides, and open fields (Miller et al. 2010). Chinese lespedeza is found in 31 states in central and eastern United States (PLANTS 2011). It forms thick clusters that can spread over large areas and ultimately prevent forest regeneration, and its seed pods can stay viable for years (Miller et al. 2010). The species has been planted as a part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, ASMC).

Japanese honeysuckle (*Lonicera japonica*)

Japanese honeysuckle is native to Asia (Ohwi 1965) and was introduced to the United States in 1806 (Leatherman 1955), with the first noted escape from cultivation occurring in 1882 (U.S. National Herbarium). It was later widely planted for deer forage (Dickson, Segelquist, & Rogers 1978; Patterson 1976) and is now considered naturalized in upland and lowland forests as well as in forest-edge habitats (Patterson 1976; Yates, Levia, & Williams 2004). It has been documented in at least 42 United States, is listed as an invasive or noxious weed in several eastern states (PLANTS 2011), and is the most prevalent invasive plant in southern forest (Miller et al. 2011). The species occurs in both open and shaded areas, with annual precipitation in invaded areas averaging 1000–1200 mm and minimum temperatures as low as -15 to -8 ^{o}C (Sasek & Strain 1990). Based on the current distribution in North America, its ecology, physiology, and phenotypic plasticity, the species is expected to continue to spread in eastern North America (Schierenbeck 2004). Although it is considered a widespread, naturalized weed, as recently as 1994, it was recommended by wildlife managers for use as deer forage and cover (Dyess *et al.* 1994).

Chinese privet (*Ligustrum sinense*)

Chinese privet was introduced in the 1800s as a decorative shrub (Miller et al. 2010) and is now the most common invasive privet in the southern United States, occurring in 20 states, ranging from Texas to Massachusetts (PLANTS 2011). An evergreen thicket-forming shrub native to China and Europe, the species can grow up to ten meter tall (Miller et al. 2010). Privet is the second most abundant invasive plant in the South and is most prevalent in the understory of bottomland hardwood forests (Merriam & Feil 2002; Miller et al. 2011). The invasion by this species severely alters natural habitat and critical wetland processes, forming dense stands that exclude most native plants and preventing natural forest regeneration. The abundance of specialist birds and diversity of native plants and bees can be reduced by privet thickets (Hanula, Horn, & Taylor 2009; Wilcox & Beck 2007). Privet can survive in a variety of habitats, including wet or dry areas, but it dominates in mesic forests. Privet produces abundant seeds that are viable for about a year (Shelton & Cain 2002) and are predominately spread by birds

(Greenburg & Walter 2010). The species also increases in density by stem and root sprouts. Although controlling privet infestations costs the United States billions of dollars each year (Simberloff, Schmitz, & Brown 1997), it is still being produced, sold, and planted as an ornamental.

Autumn olive (*Elaeagnus umbellata)*

Brought to the United States in 1830 from Japan and China, autumn olive was primarily used for mine reclamation, field rows for erosion control, and wildlife habitats (Miller et al. 2010). Since then it has escaped from cultivation and is now found in 37 states including Hawaii (PLANTS 2011). Autumn olive can grow in acidic, loamy soils and produces numerous seeds (Travis & Wilterding 2005), it is a nitrogen fixer, thus can do well on poor soils (Sharp 1977). Autumn olive can aggressively colonize an area, once established, it can develop intense shade which suppresses native species, particularly those flourish on nitrogen-poor soils (Sather & Eckardt 1987). Management is required to contain the spread of this species (Travis & Wilterding 2005), but control by cutting, burning, or the combination is counter-effective and stimulates sprouting and growth (Donovan *et al.* 2007). It has been planted as part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, ASMC).

Princesstree (*Paulownia tomentosa*)

Native to East Asia, princesstree was introduced into the eastern United States in early 1800s (Miller et al. 2010), and is now found in 25 states in the east and south (PLANTS 2011). It is still widely sold and planted as an "instant" shade tree. Until recently, most research on princesstree in the United States focused on increasing growth in plantations due to the exceptional timber value in exports to Japan (Johnson, Mitchem, & Kreh 2003; Miller et al. 2010). In the northeast United States, princesstree plantations can produce valuable high quality wood, but in the southern region, due to the more favorable growing season, tree growth is too fast, producing low-density wood that is of much lower quality and value.

The presence of princesstree is associated with natural disturbance (Williams 1993) and is therefore likely to be promoted by anthropogenic disturbance. Williams (1993) classified the species as a non-aggressive species, though others (Langdon & Johnson 1994) suggested that in areas of high disturbance it shows invasive traits. Although sun-adapted and capable of extremely rapid growth in high light environments, princesstree is tolerant of a wide range of light levels (Longbrake & McCarthy 2001). Forest management practices can affect the establishment and development of this species with growth and survival on clearcuts being greater than in forest edges or in undisturbed forest (Longbrake 2001).

Sawtooth Oak (*Quercus acutissima*)

Introduced from Asia in 1862 as an ornamental, sawtooth oak is a member of the white oak family (Short 1976). Sawtooth oak is native to Japan, Korea, China, and areas of the Himalayans (Huntley 1979). Sawtooth oak has received much of its attention from the fact of rapid growth and prolific mast production at an early age and as such had been seen as useful for wildlife. The sawtooth oak may reach heights of 20 meters at maturity, and mast production may begin as early as 6 years of age, with regular annual production by 10 years of age (Huntley 1979). Although sawtooth oak possesses many favorable traits, some studies have shown that it is not as hardy as some of the native oaks and may not be as long lived (Huntley 1979). It has been planted as part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, Alabama Surface Mining Commission).

Sampling

Two different sampling strategies were used, intensive plot sampling to assess habitat characteristics and low intensity transect sampling to assess landscape characteristics. For habitat plot sampling, sampling points were selected using the stratified spatial balanced sampling design, Generalized Random Tessellation Stratified (GRTS) (Stevens & Olsen 2004). GRTS design allows flexibility in sampling; the selected sample points are spatially balanced, so that if a point is inaccessible (land access permit and difficult physical conditions), the next point in the sample-list can be selected while maintaining spatial balance. Sampling was also allowed to be extended beyond the initial plan if time permitted while maintaining spatial balance. Two hundred sites were located across the study area with the goal of surveying at least one hundred sites. Site selection was stratified by years since reclamation: >20 years, 10 to 20 years, <10 years. At each sample site, an adaptive cluster sampling design was used to assess the magnitude of invasive plants and habitat and environmental conditions which might encourage introduction and spread of invasive plant species. Adaptive sampling was employed when individuals of invasive species were found on the main survey plot; four additional sampling plots were used gain more information about the species preferences. As invasive plants are often a rare or clustered event, this approach allows for greater efficiency of research resources by ensuring effort is targeted to where the plants are located (Brown 2003; Kriticos *et al.* 2003).

Habitat plot field sampling occurred from June through October 2010. We sampled 112, 405 m^2 (1/10-acre) circular plots. GPS coordinates (Trimble Nomad with ArcPad), date, time, forest type (pine, mixed or hardwood), regeneration type (natural or planted), distance to established forest, and forest ages were recorded on each plot. All trees with $\geq$ 25 mm diameter at breast height (DBH, ca 1.4 meter above ground level) were recorded for species and categorical DBH (25 to 75 mm, 75 to 150 mm, 150 to 225 mm, 225 to 375 mm, or greater than 375 mm) to assess habitat structural diversity. These categorical groupings were latter reduced to three, small (DBH 25 to 75 mm), medium (75 to 225mm) and large (greater than 225mm). An increment borer was used to obtain a tree core from the largest accessible tree in each plot. Two circular subplots of 1.8 meter radius were established 3.7 meter north and south of the main plot centre for assessing percentages of overstory, midstory, and understory cover (0 to 1m) (USDA Forest Service 1998) and the dominant species in each stratum. Ground variables were recorded at each subplot as percent cover of rock, bare soil, litter (tree and grass litter were estimated separately), non-vascular plants and fungi, and downed woody debris. A hand-held spherical densiometer was used to determine the cover of the forest canopy within each of these subplot, two readings were taken at each subplot to give four readings per plot. Leaf litter and humus depth were measured to the nearest mm at north and south edge of each subplot (four readings per plot). After removing the litters from the soil surface, soil samples were taken with a hand-held-probe (10 cm i.d.) from 0 to 10 cm depth at the centre of each subplot (two soil samples per plot). The soil samples were air-dried, ground, and sieved using a two mm stainless-steel sieve into plastic bags and stored until soil analysis was undertaken. If any invasive plant species was detected, an additional four neighboring sampling plots, referred to as adaptive plots, were measured, using the same sampling techniques as for the main plot, with the plot centre 33.5 meter in each cardinal direction from the main plot centre. In few cases, it was not possible to reach the additional plot due to water or topography (cliffs); in such cases no data was recorded for that additional plot.

For landscape sampling, sites were also selected using a GRTS design (Stevens & Olsen 2004). Sixty groups of plots were located across the 1983-2005 reclaimed mined landscape of the SHR, with the goal of sampling at least 30 groups of sites. Sampling was carried out in the spring of 2011. Sampling was stratified by mine age: >20 years, 10 to 20 years and <10 years. Vegetation sampling for herbaceous, forbs and vines was carried out at 129 plots within each site, and at 645 plots for shrubs and trees. Plots were arranged in figure eight transects covering 1.6km (200m on each side of the figure eight). Occurrence of invasive plants was assessed for every 6 meter of the transect. Forbs and vines were only assessed on the main transect, while trees and shrubs were assessed on the main transect, as well as 3 to 9 and 9 to 15 meters on either side of the main transect.

Soil Analysis

For soils samples taken at the habitat plots soil pH was measured in water at a soil to solution ratio of 1:2. The pH reported was temperature compensated at 25^{o}C. Total C, N, and S in the soil were determined using the dry combustion method with a vario Max CNS analyzer (Elementar, Hanau, Germany). Cation exchange capacity (CEC) was measured using the ammonium acetate (pH 7) method. Available micronutrients (Fe, Zn, Cu and Mn) were extracted using DTPA method (Lindsay & Norvell 1978), while macronutrients (K, Ca, Mg, P, Na) were extracted using Mehlich 3 solution (Mehlich 1984) and analyzed using inductive couple plasma spectroscopy (ICP, Perkin Elmer, Massachusetts, USA). Inorganic ammonium and nitrate content in the soil were extracted with 2 *M* KCl and analyzed using ammonium-nitrate analyzer (Timberline Instrument, Model no. TL-2800). Ammonium acetate extractable bases (K, Na, Ca, and Mg) were used to determine percent base saturation of the soil. Once analysis was complete, results were combined for each main plot and used to represent the main plot and surrounding adaptive plots

Geospatial data

Mine boundaries were obtained from Alabama Surface Mining Commission and verified with aerial photos. The age since mine closure was determined by the permit release or forfeit date, and grouped into three age classes (1983 to 1990, 1991 to 2000, and 2001 to 2006). Environmental and topographic variables were represented by slope, aspect (northness), solar radiation, curvature, and distance from water. These variables were selected based on their biological significance in other studies (Bartuszevige, Gorchov, & Raab 2006; Gutirrez et al. 2005; Lemke et al. 2011; Lockwood, Hoopes, & Marchetti 2007). They were predominately derived from a 10 meter digital elevation model (DEM) (United States Geological Service 2011). The DEM was used to generate slope (degrees), aspect (degrees), solar radiation (Wh/m^2) and curvature using ArcGIS (ESRI 2009), Spatial Analyst Tools. Aspect was transformed into a linear north–south gradient (northness) by performing cosine transformation (Guisan, Weiss, & Weiss 1999). Solar radiation was calculated as the annual watt hours per square meter given no cloud cover. Curvature is a measure of shape of the landscape, whether it is flat, convex, or concave. In ArcGIS "curvature" assesses surrounding cells to calculate a curvature, with increasing positive scores representing increasing concavity (ESRI 2009).

Streams and water bodies may affect the distribution and establishment of plant species by influencing seed dispersal and moisture availability. Riparian areas have been shown to contain more non-native plant species than nearby upland areas (Stohlgren *et al.* 2002). Therefore distance from stream was included in the model. Considerable landscape alteration (due to mining activities) has occurred so this information was digitized from 2009 aerial photo and not taken from available geospatial datasets. Climate and elevation data was not integrated in to these models, because with sampling only a few counties the variation in climate and elevation is not significant enough to have any strong relevance.

A number of anthropogenic variables were integrated in to this study, and included land cover and roads. Public road files are available from the Census Bureau and Alabama Department of Transport but due to the large numbers of access roads, additional roads were digitized from 2009 aerial photos. Land cover data were taken from the National Land Cover Data (NLCD) available for 1992, 2001 and 2006 (Fry et al. 2011; Homer et al. 2004; Vogelmann et al. 2001). Land cover characteristics that were derived for the above dataset included percent forest within 100m of plot for each year. Open area was digitized from aerial photos of 2009. Distance to forest edge was also estimated using this dataset.

Color infrared imagery (CIR) of 2009, Landsat Thematic Mapper imagery of 1987, 1991, 1998, 2004 and 2011 were used to derive Normalized Difference Vegetation Index (NDVI) (Rouse et al. 1974). Images were pre-processed by absolutely calibrating the most recent image (reference image), and then normalizing the older historical images (Schroeder 2006).

Data Analysis

Habitat data were analyzed in three groups: soil characteristics, ground variables (from soil to understory), and forest structure (above understory). Soils nutrient variables were standardized to a concentration of parts per million (ppm). Ground variables included categorical ground cover recorded as percent, percent understory cover, litter depth and humus depth. Forest structure was estimated using tree measurements and included diversity indices (Shannon & Weaver 1949; Simpson 1949), basal area (of trees with a DBH > 150 mm), and tree density, percent upper and midstory cover, and overall canopy cover. These calculations were conducted for all forest types combined, and then for the pines and hardwoods, separately. Correlations among variables within each habitat groups were assessed to exclude the variables with highly correlated ($r^2 > 0.50$) variables for further analysis. The selection among highly correlated variable was based on the relative easiness for field application.

The relationship between habitat variables and the invasive community was initially assessed using Canonical Correspondence Analysis (CCA). Invasive plant species that were observed at less 5% of the sites times were excluded for CCA (Heikkinen 1996; Hill 1991). We first assessed the relationship between the invasive community and each of the three groups of habitat variables separately. An overall CCA was then conducted, using the variables that had the strongest associations ($r^2>0.3$) based on the three habitat group CCA.

Logistic regression was used to build occurrence predictive models. Logistic regression is a generalized linear model that is used to investigate the relationship between a categorical outcome and a set of explanatory variables or for predicting the probability of occurrence of an event, presence of invasive species in this study, by fitting data to a logistic curve (Hosmer & Lemeshow 2000). As with CCA, each habitat group was used first separately (soils, ground, and forest), and the variables showing significance in the separate logistic regression were used in the final overall model. Logistic regression was applied to those invasive species that in occurred ≥ 50 sampling plots to assure balance in the number of absences and presences (suggested ratio 2:8) in the data (Oommen, Baise, & Vogel 2010). Piecewise, stepwise procedure was used to build most of the parsimonious model with a p-value of 0.01 for entering or dropping out of model. A p-value of 0.01 was used for each model. With five models total, three sub-models, a combined model and a final model, the overall p-value of the analysis is limited to 0.05. For descriptive purposes percent contribution and direction of variables was tabulated. Percent contribution was determined using the Wald chi squared statistic, dropping the intercept Wald chi square and standardizing the remainder to 100. Accuracy of prediction was assessed using percentage concordance, false omission rate (FN/(FN+TN), and Type II error (FN/(FN+TP). False omission rate and Type II error were assessed based on a threshold value determined by maximizing specificity and sensitivity (Manel, Williams, & Ormerod 2002). Due to variation in species occurrence across the study area a benchmark omission rate and type II error were defined as if data were randomly assigned, and a decrease of more than 25% was considered a useful model (Hair et al. 2006).

The stability of final models for each species was assessed by re-sampling the data. One hundred observations were randomly selected by maintaining the observed occurrence:non-occurrence ratio of that species. A total of 1,000 resampling were conducted. If the mean p value of a variable from the resamples was greater than 0.15, the variable was dropped (Nilsson & Belyaev 1998). It is expected that these models have weaker relationships as the number of data points has been substantially reduced, thus a higher p-value has been used. Standard deviation and 99% confidence limits were calculated for the each variable in the final model based on resampling runs.

The first stage of the landscape analysis was to assess correlations between each of the environmental and anthropogenic variables to identify uncorrelated ($r^2 < 0.50$) variables for further analysis. Invasive species occurrence data were then assessed using logistic regression and MaxEnt. Logistic regression is generalized linear model that is used to investigate the relationship between a categorical outcome and a set of explanatory variables or for prediction of the probability of occurrence of an event by fitting data to a logistic curve. It makes use of several predictor variables that may be either numerical or categorical. Logistic regression makes no assumptions about the distribution of the independent variables. MaxEnt (Phillips, Anderson, & Schapire 2006) is based on maximum entropy probability distribution, and it is a probability distribution whose entropy is at least as great as that of all other members of a specified class of distributions. MaxEnt estimates the probability distribution that is most spread out subject to constraints such as the known locations of the species. MaxEnt only uses occurrence data. To assess models, data were split spatially with 30% assigned to a test set and 70% to a training set, with one hundred replications run for MaxEnt and logistic regression to obtain the average contribution, Area Under the Curve (AUC) and omission rate. For logistic regression, sample data were resampled to give at least a 20% occurrence for each species (Oommen et al. 2010). For descriptive purposes percent contribution and direction of variables were tabulated. Percent contribution was determined using the Wald chi squared statistic, dropping the intercept Wald chi square and standardizing the remainder to 100. Accuracy of prediction was assessed using AUC and omission rate, (FN/(FN+TP). Omission rate was assessed based on a threshold value determined by maximizing specificity and sensitivity (Manel et al. 2002). Due to variation in species occurrence across the study area, benchmark omission rates were defined as if data were randomly assigned, and decrease of more than 25% was considered a useful model (Hair et al. 2006). We used the following classes of AUC to assess model performance: 0.50 to 0.75 = fair, 0.75 to 0.92 = good, 0.92 to 0.97 = very good, and 0.97 to 1.00 = excellent (Hosmer and Lemeshow, 2000). Models were then applied to the geospatial data to give potential distribution of each species, and each model type. Maps were generated by reclassifying the continuous output to binary using the maximized specificity and sensitivity threshold. For each species the logistic and MaxEnt maps were combined to give estimates of the proportion of the landscape that had low potential (not predicted by either model), moderate potential (predicted by one model) and high potential (predicted by both models). Correlations between models for each species and between species were measured. These maps were then combined to give an estimate of invasive species diversity across the landscape.

Results and Discussion

We sampled a total 374 plots, 112 main plots and 262 adaptive plots in the habitat sampling. Average forest age was 13 ± 7 years. The ground cover was variable, though predominantly litter in composition, averaging 63 ± 32% litter coverage. The predominant herbaceous species was Chinese lespedeza. Understory cover was high at 59 ± 26%, with midstory averaging 23 ± 20%. The sites varied in forest composition from no tree cover to even-aged pine stands to mixed-species of varying ages. Basal area across all sampling plots averaged 43 ± 53 m^2/ha. Pine was the major component (95% of the total basal area), this is the species of choice when reforesting reclaimed mines in this area. The soils were mostly acidic, with pH ranging from 3.89 to 7.12. Macro and micronutrients content ranged from 1.8 (phosphorus) to 2,468 (calcium) and from 0.3 (copper) to 447 (iron) mg kg^{-1} soil, respectively. The CEC ranged from 2.5 to 21.5 cmole kg^{-1} soil, while the percent of base saturation ranged from 2 to 137% (Table 1).

The CCA of soil variables within the invasive plant community illustrated that autumn olive and princesstree are associated with sites with higher nitrogen, and lower calcium to magnesium ratio; Chinese privet and Japanese honeysuckle were associated with high manganese; whereas Chinese and shrubby lespedeza were associated with lower nitrogen, and higher calcium to magnesium ratio (a). The first two CCA axes with soil features explained 12 % variation within the invasive community (Figure 3a). The CCA of ground variables with the invasive plant community showed shrubby lespedeza and princesstree preferred sites with more bare soil, Chinese lespedeza was associated with high grass litter cover; Japanese honeysuckle and Chinese privet were strongly associated with litter depth and litter cover; and autumn olive was most strongly associated with downed woody debris (Figure 3b). The first two CCA axes of ground variables explained 9 % variation within the invasive community (Figure 3b). The CCA of forest variables with the invasive plant community showed stronger associations including hardwood basal area with princesstree and autumn olive, along with high canopy cover, high diversity and high hardwood density with Japanese honeysuckle and Chinese privet; Chinese and shrubby lespedezas were negatively associated with the forest structure variables (Figure 3c).

Table 1: Habitat variables measured at each sampling plot (X identifies variables with low correlations that are used for further analysis)

	Code	Unit	r^2<0.50	mean	SD	min	max
pH	pH		X	5.55	0.70	3.89	7.12
Phosphorus	P	ppm	X	10.2	6.5	1.8	34.9
Potassium	K	ppm	X	163	86	14	440
Sodium	Na	ppm	X	36	16	6	104
Magnesium	Mg	ppm	X	249	165	16	746
Calcium	Ca	ppm		814	630	42	2468
Iron	Fe	ppm	X	192	93	16	447
Zinc	Zn	ppm	X	5.4	4.4	0.5	23.7
Copper	Cu	ppm	X	2.8	2.1	0.3	10.7
Manganese	Mn	ppm	X	99	65	5	340
Calcium Magnesium ratio	Ca:Mg		X	4.6	7.3	0.5	54.8
Ammonium	NH4	ppm	X	11.4	5.8	2.9	43.2
Nitrate	NO3	ppm	X	6.9	7.3	0	36.4
% Carbon	%C	%		2.0	1.4	0.1	6.2
% Nitrogen	%N	%	X	0.13	0.08	0.01	0.37
% Sulfur	%S	%	X	0.06	0.09	0.00	0.48
Carbon Nitrogen ratio	C:N		X	14.7	4.7	5.8	25.7
cation exchange capacity	CEC		X	11.6	3.6	2.5	21.5
% base saturation	%sat	%		45	30	2	137
% understory	%Under	%	X	59	26	0	100
% Rock	%Rock	%	X	4	9	0	70
% Bare Soil	%BS	%	X	9	15	0	80
% non vascular plants	%NVP	%	X	3	6	0	40
DWD	%DWD	%	X	8	12	0	80
% Shale	%Shale	%	X	6	14	0	88
% Leaf Litter	%LL	%		51	39	0	100
% Grass Litter	%GL	%	X	13	21	0	95
%Total Litter	%Litter	%	X	63	32	0	115
Litter Depth	Litter	cm	X	1.8	1.3	0	8.0
Humus depth	Humus	cm	X	0.8	1.0	0	5.6
Richness	Rich			5	4	0	23
Shannon	Shan			0.76	0.67	0	2.61

Table 1continued: Habitat variables measured at each sampling plot

	Code	Unit	r^2<0.50	mean	SD	min	max
Simpson's Evenness	Simp		X	0.39	0.30	0	1
Hardwood Richness	Hrich			3	4	0	21
Oak Richness	Orich			0.4	1.2	0	7
Densiometre	Canopy cover	%	X	49	33	0	96
% Overstory	%upper	%		26	29	0	95
% Midstory	%mid	%	X	23	26	0	100
Number of stems per plot	DEN		X	52	54	0	388
# of small stems 25 to 75 mm	SDEN			28	36	0	242
# of medium stems 75 to 225 mm	MDEN			20	24	0	167
# of large stems greater than 225 mm	LDEN			3	5	0	34
Basal area of trees greater than 150 mm	BA	m2/ha	X	43	53	0	303
# of pine stems	PDEN			36	45	0	388
# of small pine stems 25 to 75 mm	SPDEN		X	17	29	0	235
# of medium pine stems 75 to 225 mm	MPDEN		X	17	23	0	167
# of large pine stems > 225 mm	LPDEN			3	5	0	34
Basal area of pine trees > 150 mm	PBA	m2/ha		39	52	0	301
# of hardwood stems	HDEN		X	15	32	0	230
# of hardwood stems 25 to 75 mm	SHDEN			11	23	0	203
# of hardwood stems 75 to 225 mm	MHDEN			4	9	0	65
# of hardwood stems > 225 mm	LHDEN			0.4	1.3	0	10
Basal area of hardwood trees > 150 mm	HBA	m2/ha	X	5	15	0	104
Number of heavy seeding hardwood stems	HHSDEN			3	11	0	131
Basal area of heavy seeding hardwood trees > 150 mm	HHSBA			2	9	0	86
Forest Age	Age	years	X	13	7	0	50

The first two CCA axes of forest structure variables explained 18% variation within the invasive community (Figure 3c). The first two axes of CCA with selected variables combined from three habitat variable sets explained 13% and 6% of the invasive community variation, respectively (Figure 3d). Overall, forest structure variables had the only strong correlations with the invasive plant community; and followed the same pattern as with the forest CCA.

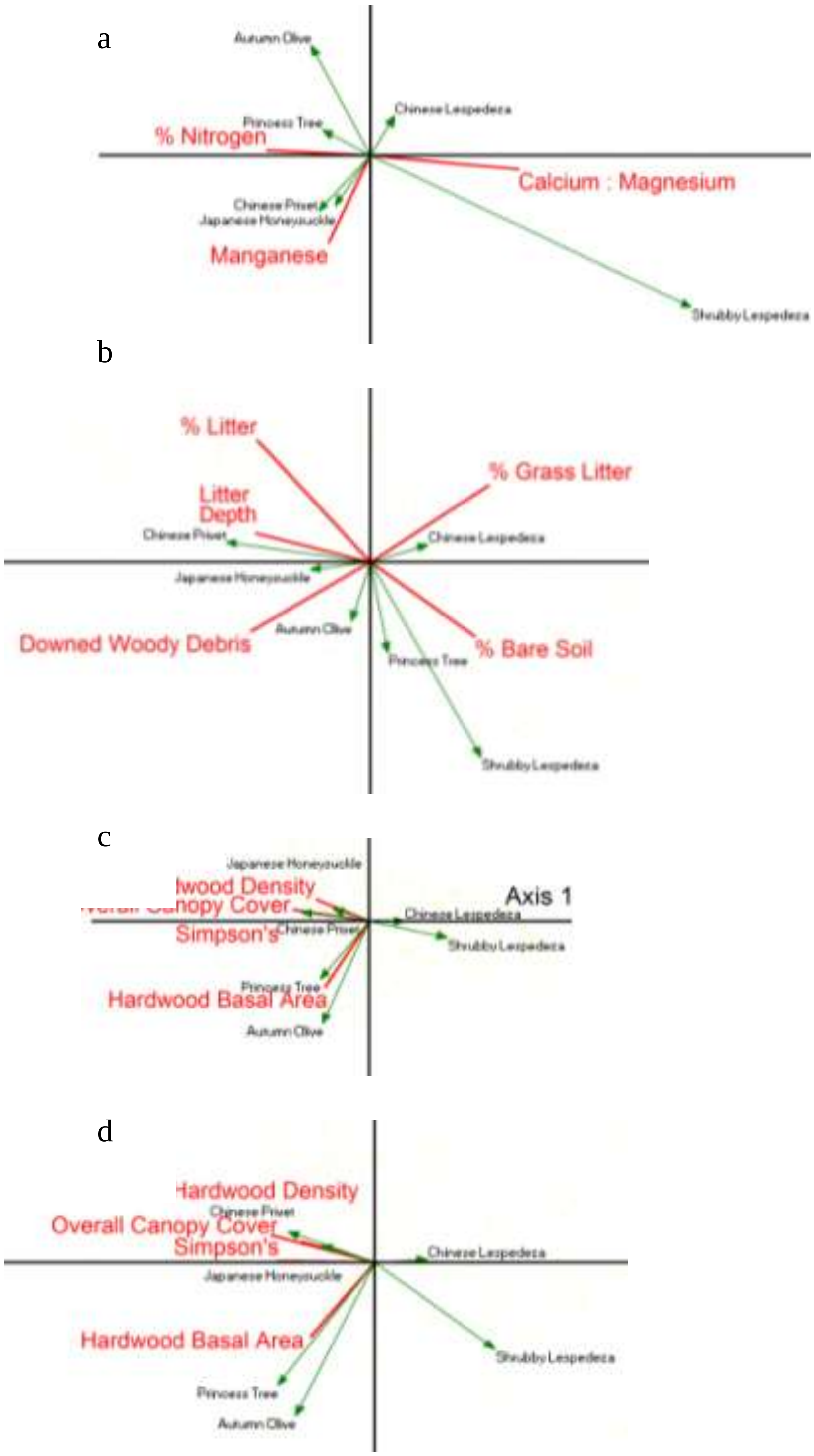

Figure 3: Relationship between habitat variables and the invasive community as assessed through Canonical Correspondence Analysis (CCA), a – soil features (axis 1 = 7, axis 2 =5), b – ground (axis 1 =7, axis 2 =2), c – forest structure (axis 1 = 12, axis 2 = 4)), d – all habitat variables combined (axis 1 =13, axis 2 =6), variables r^2>0.30 are displayed.

The invasive community was most strongly associated with vegetation characteristics such as plant diversity, canopy cover, forest age, and basal area, suggesting that the long-term management of these areas may have the greatest impact on reducing preferential habitat for invasive plants. The majority of the invasive species were in the older, larger, more established forests (15 + years) that had higher tree diversity and where the invasive species would have had more time to establish. The managed monoculture pine plantations and open areas were less likely to have multiple invasive plants. Four of the species are strongly associated in the community analysis, Chinese privet, autumn olive, princesstree, and Japanese honeysuckle, suggesting similar habitat preferences.

We applied logistic regression to three invasive species that occurred at 50 or more habitat plots*:* Chinese lespedeza, Japanese honeysuckle, and Chinese privet, using habitat variables selected with limited correlation (Tables 2 to 4). Regressions using the soils data included seven soils variables (Table 4), with no variable dominating all models. Regression models for the ground component used six of the eight variables, with percent grass litter having the highest overall contribution to all models at 31% (Table 4). Of the ten forest composition variables, four were used for the logistic regression, with canopy cover dominating models (Table 4).

The regressions with combined variables all had reasonable concordance (>75) and over 25% decrease in false omission rate and type II error from random, suggesting useful models for predicting occurrence of the three invasive species (Table 2). Re-sampling assessment suggested that relative contribution of habitat variables, accuracy for prediction, and p-value were stable for most variables, and models remained significant. There were two variables that were not stable ($p>0.15$): ammonium for privet and hardwood density for Japanese honeysuckle (Table 3). These variables were dropped and the models were rerun.

Chinese lespedeza had a positive relationship with soil magnesium and negative relationships with downed woody debris, midstory cover, and hardwood density. This suggests that Chinese lespedeza is more likely to be found in open or pine areas with higher magnesium levels in the soil and little or no midstory and downed woody debris. There was more than a 50% decrease in error from random, suggesting this model is useful in assessing habitat characteristics that are influencing the occurrence of Chinese lespedeza. Chinese lespedeza has been planted since 1970 as part of reclamations; this still continues today (pers com Dr. Randall Johnson, Director, Alabama Surface Mining Commission). It is very prevalent throughout the SHR, having been widely planted and then dispersed. Its high tolerance for a wide variety of habitats (Miller et al. 2010) has made it a pervasive invader in the area. It forms thick clusters that have spread over large areas and may ultimately prevent forest regeneration (Miller et al. 2010). In this study, Chinese lespedeza was more likely to be found in open or pine areas with higher magnesium levels in the soil and little or no midstory and downed woody debris. The model had a high false omission rate, suggesting there are other reasons for Chinese lespedeza occurrence than the attributes measured. One of the potential confounding factors is the active planting of this species. For the management of this species, increased canopy cover with a diverse forest structure seems to be the best long term approach, but the biggest contribution to management of this species would be elimination from seeding material.

Japanese honeysuckle was found in high canopy cover with little midstory and in areas of high soil magnesium and higher diversity. Canopy cover was the most important variable, contributed 41% of the predictive power. There was more than a 60% decrease in error from random, suggesting this model is useful in assessing habitat characteristics that are influencing occurrence of Japanese honeysuckle. Japanese honeysuckle has been widely planted for deer and cattle forage (Dickson et al. 1978; Patterson 1976) and is now considered naturalized in upland and lowland forests as well as in forest-edge habitats (Patterson 1976; Yates et al. 2004). It is not as detrimental as some of the other non-native species, but it has been shown to impact even-aged pine regeneration when at very high densities.

Chinese privet had one variable, a positive relationship with canopy cover. There was more than a 50% decrease in error from random, suggesting this model is useful in assessing habitat characteristics that are influencing occurrence of privet, even with only one variable. Of the three species considered, Chinese privet might be the most detrimental. It is considered the second most abundant invasive plant in the South and is most prevalent in the understory of bottomland hardwood forests (Merriam & Feil 2002). It can form dense stands to the exclusion of most native plants and replacement regeneration, impacting the abundance of specialist birds and diversity of native plants and bees (Wilcox & Beck 2007).

Table 2: Summary statistics of three invasive species from three logistic regression submodels (soil, ground and forest), combined models and final model (variables that remain stable). MaxSS is the threshold where sensitive plus specificity is maximized, false omission rate is FN / (FN + TN), type II error is FN / (FN + TP).

		Soil	Ground	Forest	Combined	Final
Chinese Lespedeza	% Concordance	83	75	78	89	89
	Max SS Threshold	0.86	0.68	0.8	0.78	0.78
	Max SS false Omission Rate	68	47	46	34	34
	Max SS Type II	36	11	14	9	9
	Decrease in false omission rate from random (80)	15	41	43	58	58
	Decrease in type II from random (20)	-80	45	30	55	55
Japanese Honeysuckle	% Concordance	76	78	83	85	87
	Max SS Threshold	0.5	0.52	0.56	0.56	0.46
	Max SS false Omission Rate	25	33	30	28	17
	Max SS Type II	14	22	21	20	15
	Decrease in false omission rate from random (58)	57	43	48	52	71
	Decrease in type II from random (42)	67	48	50	52	64
Chinese Privet	% Concordance	64	55	73	77	76
	Threshold	0.14	0.16	0.2	0.2	0.2
	Max SS false Omission Rate	5	6	7	8	8
	Max SS Type II	7	13	22	25	31
	Decrease in false omission rate from random (19)	74	68	63	58	58
	Decrease in type II from random (81)	91	84	73	69	62

Table 3: Summary of resampling of final logistic model for 100 observations run 1000 time. Variable contribution, direction, 99% confidences limit, standard deviations and mean p value of the 1000 resampled models is given.

		Contribution			P value
		Mean of resamples	99% confidence limit	SD	Mean of resamples
Chinese Lespedeza	Hardwood Density	-13	1.7	6.5	0.11
	Magnesium	38	2.5	9.5	<0.01
	Midstory	-17	2.1	8.2	0.08
	Manganese	-32	2.4	9.0	0.02
Chinese Privet	Canopy cover	81	3.8	14.6	0.04
	Ammonium	-19	3.9	14.9	0.32
Japanese Honeysuckle	Canopy cover	41	2.8	10.6	<0.01
	Hardwood Density	-8	1.6	6.0	0.28
	Magnesium	14	2.6	9.8	0.12
	Midstory	-11	1.9	7.4	0.15
	Simpson's	26	2.3	8.7	0.02

Table 4: Summary of significant variables for three invasive species from three logistic regression submodels (soil, ground and forest), combined models and final model with only variables that remain stables over resampling. Percent contribution to the model and direction of relationship are given along with the average contribution of each variable to all species.

		Chinese Lespedeza	Japanese Honeysuckle	Chinese Privet	Average Contribution
Soil features	CEC		21	66	29
	Magnesium	47	12		20
	Manganese	-25	12		12
	Ammonium	-16	-21	-34	24
	Zinc		-17		6
	% Nitrate		16		5
	Sodium	-12			4
Ground	Downed Woody Debris	-27			9
	Grass Litter		-37	-56	31
	Humus Depth	-31	-8		13
	Shale			-44	15
	Total % Litter		44		15
	Understory	41	12		18
Forest structure	Canopy cover		53	100	51
	Hardwood Density	-77	-9		29
	Midstory	-23	-14		12
	Simpson's		24		8
Variables combined	Canopy cover		45	85	43
	Hardwood Density	-16	-6		7
	Magnesium	32	13		15
	Midstory	-19	-11		10
	Manganese	-33			11
	Ammonium			-15	5
	Simpson's		25		8
Resample assessment based on variables combined	Canopy cover		44	100	48
	Hardwood Density	-16			5
	Magnesium	32	17		16
	Midstory	-19	-18		12
	Manganese	-33			11
	Simpson's		21		7

For landscape analysis 36 groups of sites were visited to give 4,644 plots assessed for herbs, vines and forbs and 23,220 plots for trees and shrubs. Overall elevation ranged from 103m to 230m, with the average slope of 9%. The average distance to a road (including service roads) was 92m and was 240m to water, and the surrounding area (100m radius) of any point was on average 42% forested (Table 5). Pearson's correlation was used to remove the highly correlated variables of open area, proportion of forest within a 100m area, NDVI 1991 and NDVI 2004 resulting in 13 non correlated variables for model use.

Table 5: Summary of geospatial variables measured at each sampling site.

code	Variable	mean	SD	Range	Unit	Source
age	Years since bond release or forfeit	1995	6	1983 to 2006	years	Alabama Surface Mining Commission
northness	transformed into a linear north–south gradient, cosine transformation	0.03	0.71	-1 to 1		10m DEM (USGS 2011)
curvature	shape of the landscape, whether it is flat, convex, or concave (increasing positive scores representing increasing concavity)	0.04	1.46	-14.8 to 16.5		10m DEM (USGS 2011)
slope	Slope	8.9	6.6	0.3 to 55	degrees	10m DEM (USGS 2011)
solar	Solar radiation given no cloud cover	246	10	131 to 254	Wh/m2	10m DEM (USGS 2011)
dist_riv	Distance to water	240	132	0 to 660	m	2009 aerial photo
open_100	Proportion of open land within 100m of plot	0.39	0.34	0 to 1		2009 aerial photo
dist_for	Distance to forest	73.3	88.1	0 to 360	m	2009 aerial photo
dist_roads	Distance to roads	92.2	71.6	0 to 352	m	2009 aerial photo
ff06	Proportion of forest within 100m of plot in 2006	0.42	0.36	0 to 1		USGS LULC
ff00	Proportion of forest within 100m of plot in 2001	0.46	0.36	0 to 1		USGS LULC
ff90	Proportion of forest within 100m of plot in 1992	0.78	0.29	0 to 1		USGS LULC
NDVI2009	NDVI in 2009	0.19	0.05	-0.05 to 0.31		2009 aerial photo
NDVI1987	NDVI in 1987	0.43	0.25	-0.12 to 0.74		Landsat
NDVI1991	NDVI in 1991	0.34	0.21	-0.05 to 0.71		Landsat
NDVI1998	NDVI in 1998	0.23	0.11	-0.05 to 0.6		Landsat
NDVI2004	NDVI in 2004	0.41	0.17	-0.03 to 0.73		Landsat
NDVI2011	NDVI in 2011	0.43	0.11	0 to 0.66		Landsat

A good model was defined as one that had an AUC of greater than 0.75 and decrease omission rate of more than 0.25, and nine good models were developed. Lespedeza was the only species that did not have any good models. Each of the other five species had similar AUC and decrease in omission rates between MaxEnt and logistic regression models. The greatest discrepancy was for sawtooth oak where MaxEnt had an AUC 0.09 higher than logistic and 0.15 greater decrease in omission rate than logistic. The distance to forest had the highest overall contribution (19%) to the models, with distance to roads, NDVI in 1987 and 2011 each contributing over 10%. All dominate variables (over 10% individual contribution to any model) had the same direction of relationship. This suggests that landscape disturbance and habitat characteristics (amount of forest) are greatly influencing the distribution of invasive species in the area.

At an individual species level, Chinese lespedeza had the weakest models with logistic test AUC of only 0.70 and MaxEnt test AUC of 0.69 and decrease in omission rates were 0.45 and 0.36, respectively. This suggests that the model may be useful but not strong. Chinese lespedeza has been planted since 1970 as part of reclamation plans; this still continues today (pers com Dr. Randall Johnson, Director, Alabama Surface Mining Commission), and the effect of this heightened and supplemental introduction may be what we are failing to capture. Overall, Chinese lespedeza was found in newer mines that had a greater distance from an established forest, was closer to roads, had less forest in 1990, and a lower NDVI in 1998 and 2011. This suggested that it was the most recently disturbed areas that are dominated by Chinese lespedeza and that this species may be competitively excluded as forest re-establishes. In our study of habitat characteristics we found Chinese lespedeza was more likely to be found in open or pine areas with higher magnesium levels in the soil and little or no midstory and downed woody debris. For the management of this species, increased canopy cover with a diverse forest structure seems to be the best long-term approach, but the best management practice to assist in eliminating this species from the reclaimed sites would be to ban it from allowable seeding mixtures.

Chinese lespedeza and Japanese honeysuckle utilized opposing habitats in the landscape but were ubiquitous across the area. Japanese honeysuckle had the second weakest model with AUC of 0.75 and 0.73, however these models had a high decrease in omission rates at 0.70 and 0.66. Japanese honeysuckle was more likely to be found close to or within forest and with a higher

NDVI in 2011, suggesting it is primarily in forested environments. This agrees with other studies that have found Japanese honeysuckle to have high shade tolerance and lower competitive abilities in open/high light environments (Miller et al. 2010). Japanese honeysuckle has been widely planted for deer and cattle forage (Dickson et al. 1978; Patterson 1976) and is now considered naturalized in upland and lowland forests as well as in forest-edge habitats (Patterson 1976; Yates et al. 2004). It is not as detrimental as some of the other non-native species, but it has been shown to impact even-aged pine regeneration when established at very high densities. Given Japanese honeysuckle prevalence throughout the southern states, there may be scant management efficacy for its removal from the SHR.

Privet had reasonable models, with AUC of 0.79 and 0.83, and decreases in omission rates of 0.54 and 0.58. The models agreed that a higher chance of privet occurrence was associated with proximity to forest(s), proximity of river(s) and age (older mines as having higher chance of occurrence). Privet was more likely to be found close to, or within forest, close to water and on older reclaimed mines. Habitat analysis found that privet was associated with areas with high canopy cover. Privet is considered the second most abundant invasive plant in the south and is most prevalent in the understory of bottomland hardwood forests (Merriam & Feil 2002). It can form dense stands to the exclusion of most native plants and replacement regeneration, impacting the abundance of specialist birds and diversity of native plants and bees (Wilcox & Beck 2007). We would suggest this is of management concern in the SHR and that as forest regenerates that it would be advisable to manage for privet, particularly in the depressions and low lying areas that are more moist and closer to water.

Autumn olive had strong models with AUC of 0.82 and 0.88 and change in omission rates of 0.85 and 0.78. Autumn olive was more likely to occur within or closer to forest, closer to rivers but further from roads and with low NDVI in 1987, suggesting it is more prevalent in areas that have had fewer disturbances in recent years. Once established, it can develop intense shade which suppresses native species and can cause serious problems for native species that flourish on nitrogen poor soils (Sather & Eckardt 1987). It has been planted as part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, Alabama Surface Mining Commission). As this species is not at high densities throughout the larger region (Miller et al.

2011), it is very likely that removal of current infestation and no future planting activity would be very worthwhile and possible lead to eradication in the area.

Princesstree had strong models with AUC of 0.97 and 0.96 and change in omission rates of 0.91 and 0.93. Princesstree occurrence was more likely to be found with low NDVI in 1987 and high NDVI in 2009, or historical disturbance but no recent disturbance. Princess tree occurrence was as approximately even throughout the broader region and although some active management would be useful as this species is not of major concern.

Sawtooth oak had reasonably good models with AUC's of 0.83 and 0.92 and change in omission rates of 0.60 and 0.75. Sawtooth oak was more likely to be found within or closer to forests and further from roads. It has been planted as part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, Alabama Surface Mining Commission) as a wildlife species. Although sawtooth oak possesses many favorable traits, some studies have shown that it is not as hardy as some of the native oaks and may not be as long lived (Huntley 1979) , with maturing no earlier than some of the native oaks. During the course of the field work we did not notice volunteer species, however any non-native species is a risk and we would suggest that it be removed from the list of plants appropriate for reclamation.

Species models were then applied to the mined landscape of the SHR to assess the probable prevalence of each species across the landscape. Japanese honeysuckle had the highest probable prevalence at 48% (73% moderate potential), with princesstree having the lowest at less than 1% (3% moderate potential)(Table 7). Overall 33% of the landscape was predicted to have no invasive plants, with 47% predicted to have one, 17% to have two and 3% to have three or more. An example of the mapped output is given in Figure 4. Chinese lespedeza and Japanese honeysuckle had the highest correlation (r= -0.43), and were found on opposing areas (example Figure 4 B and C). Of interest is how the invasive plant distribution in this landscape compares to the rest of the region. Overall, all invasive species apart from Japanese honeysuckle (5%) and privet (2%) have an average of less than 1% current coverage across the southern region (Miller et al. 2011). All species we modeled, apart from princesstree and sawtooth oak, showed much high occurrence, suggesting that there is something of concern in this area. When assessed as probable proportion of the mined landscape this discrepancy increased even further.

Table 6: Summary statistics of six invasive species from logistic regression and MaxEnt models from 100 resample's, variable contribution and direction of relationship are given along with the average contribution of each variables to all species.

	Autumn Olive		Chinese Lespedeza		Privet		Japanese Honeysuckle		Princesstree		Sawtooth Oak	
	M	L	M	L	M	L	M	L	M	L	M	L
Test AUC	0.88	0.82	0.69	0.70	0.83	0.79	0.73	0.75	0.96	0.97	0.92	0.83
MaxSS Threshold	0.31	-1.83	0.46	-0.09	0.32	-1.26	0.42	-0.88	0.11	-1.30	0.27	-1.06
MaxSSTrOm.1	0.17	0.12	0.30	0.26	0.34	0.37	0.24	0.21	0.06	0.07	0.20	0.32
Random omission rate	0.80	0.80	0.47	0.47	0.80	0.80	0.70	0.70	0.80	0.80	0.80	0.80
Decrease in omission rate from random	0.78	0.85	0.36	0.45	0.58	0.54	0.66	0.70	0.93	0.91	0.75	0.60
Mine age	2	6	5	22	-10	-15	-2	-2	17	12	n0	1
Curvature	1	1	u0	0	n1	0	n1	0	n0	1	n1	0
Distance to forest	-17	-13	7	12	-21	-41	-27	-42	-2	2	-15	-34
Distance to water	-12	0	8	4	-21	-24	u3	-3	n0	-1	n1	3
Distance to roads	7	14	u4	-5	-8	-1	u5	-5	n2	7	30	36
Forest in 2006	13	0	-5	2	n9	-8	3	-11	4	1	n18	-4
Forest in 1992	-3	-9	-10	-11	-1	0	2	3	-1	0	-1	-9
NDVI1987	-35	-51	u2	-7	n10	1	4	0	-60	-45	n25	-3
NDVI1998	n2	2	n26	-28	n1	0	11	7	n0	-3	0	-1
NDVI2009	1	1	n2	-1	-1	1	1	4	9	12	3	4
NDVI2011	5	2	n29	-7	n12	-3	38	22	3	7	u1	-3
Northness	1	1	1	0	-3	-5	1	0	u1	-5	n2	1
Slope	1	0	1	1	n2	1	n2	-1	n1	-4	-3	1

Table 7: Probable proportion of mined landscape invaded

	Autumn Olive	Chinese Lespedeza	Privet	Japanese Honeysuckle	Princesstree	Sawtooth Oak
Low Occurrence	0.62	0.62	0.70	0.27	0.97	0.74
Moderate Occurrence	0.25	0.21	0.23	0.25	0.03	0.21
High Occurrence	0.14	0.17	0.07	0.48	<0.01	0.05
Correlation	0.41	0.51	0.28	0.52	0.30	0.17

Table 8: Correlation between species

	Chinese Lespedeza	Privet	Japanese Honeysuckle	Princesstree	Sawtooth Oak
Autumn Olive	-0.05	0.05	0.16	0.18	0.39
Chinese Lespedeza		-0.15	-0.42	0.13	-0.15
Privet			0.13	-0.03	0.01
Japanese Honeysuckle				-0.04	0.05
Princesstree					0.05

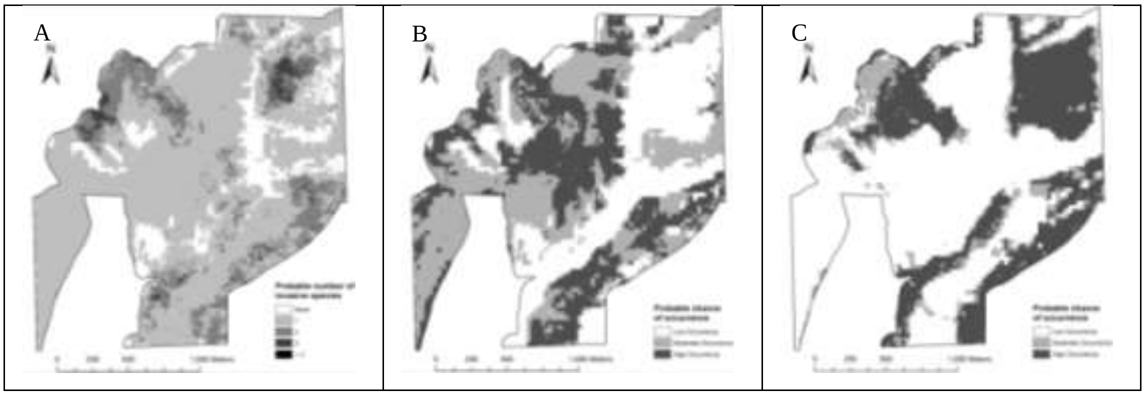

Figure 4: Example of mapping distribution (A – Combined geospatial species models to give probable number of invasive species, B – Probable distribution of Chinese lespedeza, C – Probable distribution of Japanese honeysuckle).

Conclusion

SMCRA mandates that mined land be reclaimed and restored to its original use or a use of higher value. This includes full restoration of ecosystem functions and services, of which the distribution and diversity of the plant species are an integral part. Restoration assessment often focuses on the more easily measurable restoration of edaphic and hydrological systems. These may not reflect the recovery of the pre-mining biological communities or mitigate landscape, structural and ecological changes (Burger et al. 2002). Most legislation mandates the evaluation of land reclamation success using readily quantifiable metrics with land assessed after a relatively short time period (Holl & Cairns 2002). This encourages reclamation approaches that address the short-term goals of providing erosion control and minimizing acid mine drainage but not necessarily the longer-term and more difficult to quantify objective of restoration of ecosystem services. It has been suggested that goals for short-term and long-term recovery of highly disturbed sites may be in conflict (Holl 2002). Many mine reclamation efforts focus on establishing rapid-growing non-native species that control erosion but may slow or prevent the establishment of later-successional, native species (Holl 2002). For example, a general restoration practice creates piles of soil that are then graded to a smooth condition to stabilize the surface and prevent erosion (Zipper *et al.* 2011). These sites then are revegetated by hydroseeding with a mixture of herbaceous seeds (mix of grasses and legumes) and fertilizer. This method can encourage dense herbaceous vegetation that in turn can negatively affect establishment of native trees and success of planted seedlings (Chaney, Pope, & Byrnes 1995).

Geospatial modeling of these invasive plants, at this scale, is useful and offers potential for management, both in terms of identifying habitat types most at risk and areas that need management attention. The landscape analysis indicated that all modeled species, apart from princesstree and sawtooth oak, showed much high occurrence than in the broader landscape, suggesting that this area is of concern.

The influence of planting non-native, invasive species in this area is likely the major driver of the high diversity of invasive plants, as four of the seven dominant species were planted. Adjusting the reclamation plantings to native species would aid in resolving this issue In terms of the impact invasive species are having on reclamation efforts and land productivity, further study needs to be undertaken. Of the three most dominant species, one is planted (Chinese lespedeza) and one is ubiquitous throughout the region at low densities (Japanese honeysuckle).The third species, privet, is of most concern. In general it appears that the use of deliberate planting of invasive species as a reclamation effort has in turn become a hindrance to restoring full ecosystem, however other reclamation practices are not the major drivers impacting the non-native, invasive species composition of the reclaimed, now forested mine sites.

References

Alpert, P., Bone, E. & Holzapfel, C. (2000) Invasiveness, invasibility and the role of environmental stress in the spread of non-native plants. Perspectives in Plant Ecology, Evolution and Systematics, 3, 52-66.

Bartuszevige, A., Gorchov, D. & Raab, L. (2006) The relative importance of landscape and community features in the invasion of an exotic shrub in a fragmented landscape. Ecography, 29, 213-222.

Bradshaw, A.D. (1984) Technology lecture: Land restoration: Now and in the future. Proceedings of the Royal Society of London. Series B, Biological Sciences (1934-1990), 223, 1-23.

Brown, J.A. (2003) Designing an efficient adaptive cluster sample. Environmental and Ecological Statistics, 10, 95-105.

Burger, J.A., Scott, D. & Mitchem, D. (2002) Field assessment of mine soil quality for establishing hardwoods in the Appalachians. Reclamation with a Purpose. (eds R. Barnhisel & M. Collins), pp. 226-240. 19th Annual Meeting, American Society of Mining and Reclamation, Lexington, KY.

Chaney, W.R., Pope, P.E. & Byrnes, W.R. (1995) Tree survival and growth on land reclaimed in accord with Public Law 95-87. Journal of Environmental Quality, 24, 630-634.

Davis, M. a., Grime, J.P. & Thompson, K. (2000) Fluctuating resources in plant communities: a general theory of invasibility. Journal of Ecology, 88, 528-534.

Davison, V.E. (1945) Wildlife values of the lespedezas. Journal of Wildlife Management, 9, 1-9.

Dickson, J.G., Segelquist, C.A. & Rogers, M.J. (1978) Establishment of Japanese honeysuckle in the Ozark Mountains. Proceedings of the Southeastern Association of Fish and Wildlife Agencies. pp. 242-245.

Donovan, M.L., Mayhew, S.L., Warren, B.E. & Stephens, V.E. (2007) An evaluation of chemical treatment and burning on the control of autumn olive (Elaeagnus umbellate). Michigan Department of Natural Resources, Wildlife Division Report 3471.

Dyess, J.G., Causey, M.K., Striblin, H.L. & Lockaby, B.G. (1994) Effects of fertilization on production and quality of Japanese honeysuckle. Southern Journal of Applied Forestry, 18, 68-71.

D’Antonio, C.M., Dudley, T.L. & Mack, M.C. (1999) Disturbance and biological invasions: Direct effects and feedbacks. Ecosystems of Disturbed Ground. (ed L. Walker), pp. 413-452. Elsevier, Amsterdam.

ESRI (2009) ArcGIS, Redlands, CA, USA, Environmental Systems Research Institute.

Evans, C.W., Moorhead, D.J., Bargeron, C.T. & Douce, G.K. (2006) Invasive plant responses to silvicultural practices in the south. Tifton, GA. BW-2006-03.

Fry, J., Xian, G., Jin, S., Dewitz, J., Homer, C., Yang, L., Barnes, C., Herold, N. & Wickham, J. (2011) Completion of the 2006 National Land Cover Database for the Conterminous United States. PE&RS, 77, 858-864.

Graham, E.H. (1941) Legumes for erosion control and wildlife. Misc. Pub. 412. Washington, DC: USDA. 153 p

Greenburg, C.H. & Walter, S.T. (2010) Fleshy fruit removal and nutritional composition of winter-fruiting plants: a comparison of non-native invasive and native species. Natural Areas Journal, 30, 312-321.

Guisan, A., Weiss, S.B. & Weiss, A.D. (1999) GLM versus CCA spatial modeling of plant species distribution. Plant Ecology, 143, 107-122.

Gutirrez, D., Fernndez, P., Seymour, A.S. & Jordano, D. (2005) Habitat Distribution Models: Are Mutualist Distributions Good Predictors of Their Associates? Ecological Applications, 15, 3-18.

Hair, J.F., Black, W.C., Babin, B.J., Anderson, R.E. & Tatham, R.L. (2006) Multivariate Data Analysis Sixth Edition Pearson Education. New Jersey.

Hanula, J.L., Horn, S. & Taylor, J.W. (2009) Chinese privet ({Ligustrum sinense}) removal and its effect on native plant communities of riparian forests. Invasive Plant Science and Management, 2, 292-300.

Hardt, R.A. & Forman, I.I. (1989) Boundary form effects on woody colonization of reclaimed surface mines. Ecology, 70, 1252-1260.

Haugen, A.O. & Fitch, F.W.J. (1955) Seasonal availability of certain bush lespedeza and partridge pea seed as determined from ground samples. Journal of Wildlife Management, 19, 297-301.

Heikkinen, R.K. (1996) Predicting patterns of vascular plant species richness with composite variables: A meso-scale study in Finnish Lapland. Plant Ecology, 126, 151-165.

Hill, M.O. (1991) Patterns of species distribution in Britain elucidated by canonical correspondence analysis. Journal of Biogeography, 18, 247-255.

Holl, K.D. (2002) Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. Applied Ecology, 39, 960-970.

Holl, K.D. & Cairns, J.J. (2002) Monitoring ecological restoration. Handbook of Ecological Restoration. pp. 411-432. Cambridge University Press, Cambridge, UK.

Homer, C., Huang, C., Yang, L., Wylie, B. & Coan, M. (2004) Development of a 2001 National Land-Cover Database for the United States. Photogrammetric Engineering Remote Sensing, 70, 829-840.

Hosmer, D.W. & Lemeshow, S. (2000) Applied Logistic Regression. Wiley-Interscience, New York.

Huntley, C.R.H. and J.C. (1979) Establishment of Sawtooth Oak as a Mast Source for Wildlife. Wildlife Society Bulletin, 7, 253-258.

Johnson, J.E., Mitchem, D.O. & Kreh, R.E. (2003) Establishing royal paulownia on the Virginia Piedmont. New Forests, 25, 11–23.

Kriticos, D.J., Sutherst, R.W., Brown, J.R., Adkins, S.W. & Maywald, G.F. (2003) Climate change and the potential distribution of an invasive alien plant: Acacia nilotica ssp. indica in Australia. Journal of Applied Ecology, 40, 111-124.

Langdon, K.R. & Johnson, K.D. (1994) Additional notes on invasiveness of Paulownia tomentosa in natural areas. Natural Areas Journal, 14, 139-140.

Leatherman, A.D. 1955. Ecological life-history of Lonicera japonica Thunb. Unpublished PhD thesis, University of Tennessee 97 p.

Lemke, D., Hulme, P.E., Brown, J.A. & Tadesse, W. (2011) Distribution modelling of Japanese honeysuckle (Lonicera japonica) invasion in the Cumberland Plateau and Mountain Region, USA. Forest Ecology and Management, 262, 139-149.

Lindsay, W.L. & Norvell. (1978) Development of a DTPA soil test for zinc, iron, manganese, and copper. Soil Sci.Soc. Am. J., 42, 421-428.

Lockwood, J.L., Hoopes, M.F. & Marchetti, M.P. (2007) Invasion Ecology. Blackwell Publishing, Malden, Massachusetts.

Lodge, D.M. (1993) Biological invasions: Lessons from ecology. Trends in Ecology & Evolution, 8, 133-136.

Longbrake, A.C.W. (2001) Ecology and invasive potential of Paulownia tomentosa (Scrophulariaceae) in hardwood forest landscape. , PhD, 174.

Longbrake, A.C. & McCarthy, B.C. (2001) Biomass allocation and resprouting ability of princess tree (Paulownia tomentosa: Scrophulariaceae) across a light gradient. The American Midland Naturalist, 146, 388-403.

Manel, S., Williams, H.C. & Ormerod, S.J. (2002) Evaluating presence-absence models in ecology: the need to account for prevalence. Journal of Applied Ecology, 38, 921-931.

McElfish, J.M. & Beier, A.E. (1990) Environmental regulation of coal mining: SMCRA's second decade. Environmental Law Institute, Washington, DC, USA.

McGrath, D. a., Evans, J.P., Smith, C.K., Haskell, D.G., Pelkey, N.W., Gottfried, R.R., Brockett, C.D., Lane, M.D. & Williams, E.D. (2004) Mapping land-use change and monitoring the impacts of hardwood-to-pine conversion on the Southern Cumberland Plateau in Tennessee. Earth Interactions, 8, 1.

McSweeney, K. & Jansen, I.L. (1984) Soil structure and associated rooting behavior in mine soils. Soil Science Society of America Journal, 48, 607-612.

Mehlich, A. (1984) Mehlich 3 soil test extractant: A modification of the Mehlich 2 Extractant. Commun. Soil Sci. Plant Anal., 15, 1409-1416.

Merriam, R.W. & Feil, E. (2002) The potential impact of an introduced shrub on native plant diversity and forest regeneration. Biological Invasions, 4, 369-373.

Miller, J.H., Chambliss, E.B. & Loewenstein, N.J. (2010) A field guide for the identification of invasive plants in southern forests. , 126.

Miller, J.H., Lemke, D. & Coulston, J. (2011) The Southern Forest Futures Project, Chapter 15. The Invasion of Southern Forests by Nonnative Plants: Current and Future Occupation with Impacts, Management Strategies, and Mitigation Approaches.

Negley, T.L. & Eshleman, K.N. (2006) Comparison of stormflow responses of surface-mined and forested watersheds in the Appalachian Mountains, USA. Hydrological Processes, 20, 3467-3483.

Nilsson, L. & Belyaev, Y.K. (1998) Application of resampling to exponential and logistic regression. Sweden.

Ohwi, J. (1965) Flora of Japan (FG Meyer and EH Walker, Eds.). Smithsonian Institution, Washington, DC, USA.

Oommen, T., Baise, L.G. & Vogel, R.M. (2010) Sampling Bias and Class Imbalance in Maximum-likelihood Logistic Regression. Mathematical Geosciences, 43, 99-120.

PLANTS. (2011) National Plants Database. United States Department of Agriculture, Natural Resources Conservation Services, National Plant Data Center.

Patterson, D. (1976) The history and distribution of five exotic weeds in North Carolina. Southern Appalachian Botanical Society, 41, 177-180.

Phillips, S., Anderson, R. & Schapire, R. (2006) Maximum entropy modeling of species geographic distributions. Ecological Modelling, 190, 231-259.

Ricciardi, A. (2007) Are modern biological invasions an unprecedented form of global change? Conservation Biology, 21, 329-36.

Ricketts, T.H., Dinerstein, E., Olson, D.M., Loucks, C.J. & al., et. (1999) Terrestrial Ecoregions of North America: A Conservation Assessment. Island Press, Washington, D.C.

Ripley, E.A., Redman, R.E. & Crowder, A.A. (1996) Environmental effects of mining. St. Lucie Press, Boca Raton, Florida, USA.

Rouse, J.W., Haas, R.H., Schell, J.A., Deering, D.W. & Harlan, J.C. (1974) Monitoring the vernal advancement and retrogradation (greenwave effect) of natural vegetation. Security, 371.

Sasek, T.W. & Strain, B.R. (1990) Implications of atmospheric CO2 enrichment and climatic change for the geographical distribution of two introduced vines in the U.S.A. Climatic Change, 16, 31-51.

Sather, N. & Eckardt, N. (1987) Element Stewardship Abstract for {Elaeagnus umbellata} (Autumn olive). Arlington, VA.

Schierenbeck, K. (2004) Japanese Honeysuckle (Lonicera japonica) as an Invasive Species; History, Ecology, and Context. Critical Reviews in Plant Sciences, 23, 391-400.

Schroeder, T.A. (2006) Understanding changes in forest cover and carbon storage in early successional forests of the Pacific Northwest Using USDA Forest Service FIA and multi-temporal Landsat data. , Doctor of.

Shannon, C.E. & Weaver, W. (1949) The mathematical theory of communication. Univ. Illinois Press, Urbana.

Sharp, W.C. (1977) Conservation plants for the Northeast. USDA Soil Conservation Service Program Aid 1154. 40p

Shelton, M.G. & Cain, M.D. (2002) Potential carry-over of seeds from 11 common shrub and vine competitors of loblolly and shortleaf pines. Canadian Journal of Forest Research, 32, 412-419.

Short, H.L. (1976) Composition and squirrel use of acorns of black and white oak groups. Journal of Wildlife Management, 40, 479-483.

Shukla, M.K., Lal, R. & Ebinger, M.H. (2005) Physical and chemical properties of a minespoil eight years after reclamation in northeastern Ohio. Soil Science Society of America Journal, 69, 1288-1297.

Simberloff, D., Schmitz, D. & Brown, T. (1997) Strangers in paradise: impact and management of nonindigenous species in Florida. Island Press, Washington, DC, USA.

Simpson, E.H. (1949) Measurement of diversity. Nature, 163, 688.

Skousen, J.G., Johnson, C.D. & Garbutt, K. (1994) Natural revegetation of 15 abandoned mine land sites in West Virginia. Journal of Environmental Quality, 23, 1224-1230.

Smalley, G.W. (1979) Classification and evaluation for forest sites on the Southern Cumberland Plateau. U.S. Department of Agriculture, Forest Service, Southeastern Forest Experiment Station.

Stevens, D.L. & Olsen, A.R. (2004) Spatially balanced sampling of natural resources. Journal of the American Statistical Association, 99, 262-278.

Stohlgren, T.J., Chong, G.W., Schell, L.D., Rimar, K. a., Otsuki, Y., Lee, M., Kalkhan, M. a. & Villa, C. a. (2002) Assessing vulnerability to invasion by nonnative plant species at multiple spatial scales. Environmental Management, 29, 566-577.

Sun, Q.B., Shen, R.F., Zhao, X.Q., Chen, R.F. & Dong, X.Y. (2008) Phosphorus enhances Al resistance in Al-resistant {Lespedeza bicolor} but not in Al-sensitive {L. cuneata} under relatively high Al stress. Annals of Botany, 102, 795-804.

Travis, J. & Wilterding, J. (2005) Assessment of autumn olive ({Elaeagnus umbellate}) population at Pierce Cedar Creek.

USDA Forest Service (1998) Field Instructions for Southern Forest Inventory.

Vitousek, P.M., D'Antonio, C.M., Loope, L.L., Rejmanek, M. & Westbrooks, R. (1997) Introduced species: A significant component of human-caused global change. New Zealand Journal of Ecology, 21, 1-16.

Vogelmann, J.E., Howard, S.M., Yang, L., Larson, C.R., Wylie, B.K. & Driel, J.N. Van. (2001) Completion of the 1990's National Land Cover Data Set for the conterminous United States. Photogrammetric Engineering and Remote Sensing, 67, 650-662.

Wear, D.N. & Greis, J.G. (2002) The Southern Forest Resource Assessment–Summary Report. USDA Forest Service, Southern Research Station. Gen Tech. Report SRS-53.

Webster, C.R., Jenkins, M.A.M.A. & Jose, S. (2006) Woody invaders and the challenges they pose to forest ecosystems in the eastern United States. Journal of Forestry, 104, 366-379.

Wilcox, J. & Beck, C.W. (2007) Effects of {Ligustrum sinense} lour. (Chinese privet) on abundance and diversity of songbirds and native plants in a southeastern nature preserve. Southeastern Naturalist, 6, 535-550.

Williams, C.E. (1993) The exotic empress tree, Paulownia tomentosa: an invasive pest of forests? Natural Areas Journal, 13, 221-222.

Yates, E.D., Levia, D.F. & Williams, C.L. (2004) Recruitment of three non-native invasive plants into a fragmented forest in southern Illinois. Forest Ecology and Management, 190, 119-130.

Zeleznik, J.D. & Skousen, J.G. (1996) Survival of three tree species on old reclaimed surface mines in Ohio. Journal of Environmental Quality, 25, 1429-1435.

Zipper, C.E., Burger, J.A., McGrath, J.., Rodrigue, J.A. & Holtzman, G.I. (2011) Forest Restoration Potential of Coal-Mined Lands in the Eastern United States. Journal of Environmental Quality.