International Journal of Mining, Reclamation and Environment
2012, 1–19, i*First* article

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Invasion of a mined landscape: what habitat characteristics are influencing the occurrence of invasive plants?

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(*Received 15 May 2012; final version received 29 May 2012*)

Throughout the world, the invasion of alien plants is an increasing threat to native biodiversity. Invasion is especially prevalent in areas affected by land transformation and anthropogenic disturbance. Surface mines are a major disturbance, and thus may promote the establishment and expansion of invasive plant communities. Environmental and habitat factors that may contribute to favourable conditions for heightened plant invasion were examined using the Shale Hills region (SHR) of Alabama as a case study. Overall the invasive community was predominantly associated with forest structure and composition. At an individual species level, forest structure and composition also dominated models; however, soil characteristics were also integrated. The influence of planting alien, invasive species in this area is likely the major driver of the high diversity of invasive plants, with three of the six dominant species being planted. Adjusting the reclamation plantings to native species would aid in reducing the number and diversity of invasive plants. Overall, it appears that the initial reclamation efforts, apart from the planting of alien species, are not the major driver impacting the invasive plant composition of the reclaimed, now forested mine sites.

Keywords: invasive plants; surface mining; restoration

1. Introduction

Land transformation and anthropogenic disturbance often facilitate the establishment and development of invasive plant communities. Surface mining is one of the major forms of disturbance in the United States and has changed over 2.4 million hectares of terrestrial habitat since 1930 [1]. The changes include land transformation, alteration in ecosystems and geophysical characteristics [2–5]. Consequential impacts include interruption and change of energy flow, food webs, biodiversity, successional patterns and biogeochemical cycling [6]. 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 transformed land, caused by surface coal mining in

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ISSN 1748-0930 print/ISSN 1748-0949 online

http://dx.doi.org/10.1080/17480930.2012.699215
http://www.tandfonline.com

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 [7]. Reclamation starts before the mining operation, with each new mine requiring an approved reclamation plan before commencement of operation. The first stage in the mining operation is normally the removal of the top stratum. Often this is a shallow layer of unconsolidated rubble and soil that is retained for use in the reclamation. In most cases, top-soils are not present and a heterogeneous mixture of suitable overburden materials from this top stratum is used as the final growth medium in the reclamation. On completion of the coal removal, the surface mine site is re-contoured and stabilised, covered with 'top-soil' and then vegetated. Surface mine reclamation efforts rarely result in ecosystems that mimic pre-mined characteristics; the focus is generally on short-term measurable matrices including land stability and hydrological function. However, in recent years, in the eastern United States there has been growing interest in the restoration of forest community, structure and function [8].

Throughout the world, alien plants are becoming an increasing threat to native biodiversity and ecosystem functions [9,10]. Historically and still to some extent today, alien plants are used in reclamation, to stabilise 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 [11]. 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 alien plants [12]. The characteristics of plants that assist in some of the short-term goals of restoration, including land stabilisation and nitrogen fixing are often the same traits that are associated with invasive plants. Some of the traits reclamationists favour 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 [13] and high environmental stress [12,14,15]. Mined sites often display these attributes and thus may have a high probability of being invaded by unwanted species.

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, with the highest density of invasive plants in the top half of Alabama [16]. These areas have had a long-history of human habitation and highly disturbed mining regions.

The occurrence of invasive plants was investigated in the Shale Hills Region (SHR) in mid-Alabama and assessed as to the quantification of habitat and environmental conditions, the examination of the associations of invasive community and habitat and ecological characteristics, and the development of predictive models for the occurrence of invasive species.

2. Methods

2.1. Study area

This study was conducted in the SHR of the southern Cumberland Plateau of the south-eastern United States (Figure 1). The southern Cumberland Plateau has a

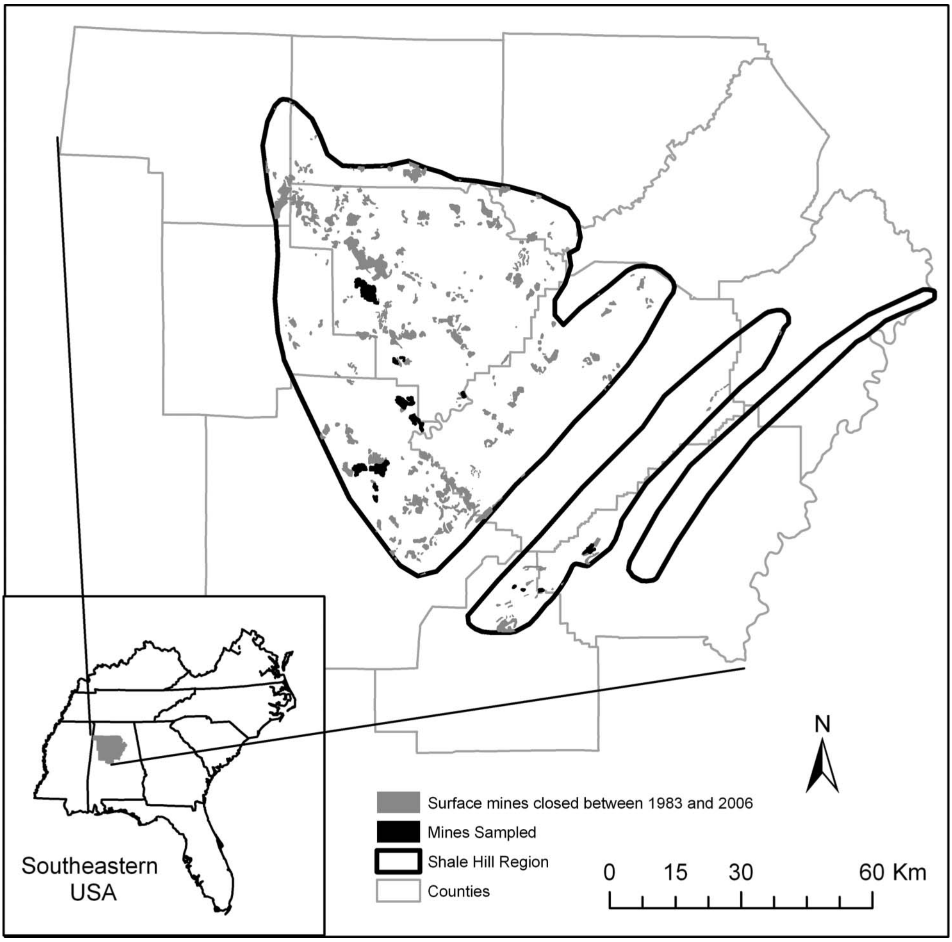

Figure 1. Study area location map, Shale Hills region, Alabama.

temperate climate characterised by long, moderately hot summers and short, mild winters [17]. The average minimum winter temperature is 1°C, and the average summer maximum temperature is 32°C [17]. Annual precipitation averages approximately 1400 mm and is fairly well-distributed throughout the year [17]. Precipitation is greatest from January through April and lowest from August through November [17]. Thunderstorms with high intensity rainfall are common in the summer [17]. The forests of the Cumberland Plateau are among the most diverse of the world's temperate-zone forests [18]. Like many 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 [19,20]. This area has undergone extensive land-use changes, 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 one 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 [17]. The target area included surface mines permitted after 1983, on both public and private lands, that were closed before 2006, thus had time to be reclaimed and for vegetation to re-establish. The final phase of restoration is the planting of the permanent vegetation, and mines considered in this study were planted at a rate of 1235–1730 pines per hectare (500–700 per acre), with 1112 pines per hectare (450 per acre) considered successful [21].

2.2. *Species of interest*

The study area has many of the 56 alien plants that are highly invasive to the forests of the south-eastern United States [22]. This study 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*) ($n = 22$). Below are brief descriptions of each of these six species.

2.2.1. *Shrubby lespedeza (Lespedeza bicolor)*

Shrubby lespedeza was introduced from Japan in the 1800s as an ornamental. It has been planted for wildlife habitat [23,24] and is also used in strip mine reclamation and along field borders [25]. It can reach 3 m in height [26] and grows well in open areas, particularly on well-drained and acidic soils [27]. Shrubby lespedeza is considered invasive in the southern region of the United States and is found in 27 states [28] throughout the country. It has been planted as part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, ASMC).

2.2.2. *Chinese lespedeza (Lespedeza cuneata)*

Introduced from Japan in 1899, Chinese lespedeza, also called Sericea lespedeza, is a long, slender perennial legume that can grow 2 m tall. The species has spread quickly due to its use in pasture and erosion control [22], along roadways, on reclaimed mines and along field borders [25]. It is flood tolerant and can survive in a wide variety of habitats, including forests, road sides and open fields [22]. Chinese lespedeza is found in 31 states in central and eastern United States [28]. It forms thick clusters that can spread over large areas and ultimately prevent forest regeneration, with seed pods which can stay viable for years [22]. It has been planted as a part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, ASMC).

2.2.3. *Japanese honeysuckle (Lonicera japonica)*

Japanese honeysuckle is native to Asia [29] and was introduced to the United States in 1806 [30], with the first noted escape from cultivation occurring in 1882 (US National Herbarium). It was later widely planted for deer forage [31,32] and is now considered naturalised in upland and lowland forests as well as in forest-edge habitats [32,33]. It has been documented in at least 42 states within the United States, is listed as an invasive in several eastern states [28] and is the most prevalent invasive plant in southern forest [16]. 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°C [34]. Based on the current distribution in the United States, its ecology, physiology and phenotypic plasticity, the species is expected to continue to spread in eastern North America [35]. Although it is considered a widespread, naturalised weed, as recently as 1994 it was recommended by wildlife managers for use as deer forage and cover [36].

2.2.4. *Chinese privet (Ligustrum sinense)*

Chinese privet was introduced in the 1800s as a decorative shrub [22] and is now the most common invasive privet in the southern United States, occurring in 20 states, ranging from Texas to Massachusetts [28]. An evergreen thicket-forming shrub native to China and Europe, the species can grow up to 10 m tall [22]. Privet is the second most abundant invasive plant in the South and is most prevalent in the understory of bottomland hardwood forests [16,37]. 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 the diversity of native plants and bees can be reduced by privet thickets [38,39]. 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 approximately a year [40] and are predominately spread by birds [41]. The species also increases in density by stem and root sprouts. Although controlling privet infestations costs the United States billions of dollars each year [42], it is still being produced, sold and planted as an ornamental.

2.2.5. *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 [22]. Since then it has escaped from cultivation and is now found in 37 states [28]. Autumn olive can grow in acidic, loamy soils and produces numerous seeds [43], it is a nitrogen fixer, thus can do well on poor soils [44]. Autumn olive can aggressively colonise an area, once established, it can develop intense shade which suppresses native species, particularly those which flourish on nitrogen-poor soils [45]. Management is required to contain the spread of this species [43], but control by cutting, burning or the combination is counter-effective and stimulates sprouting and growth [46]. It has been planted as part of reclamation in this area since the 1970s (pers com Dr. Randall Johnson, Director, ASMC).

2.2.6. *Princesstree (Paulownia tomentosa)*

Native to East Asia, princesstree was introduced into the eastern United States in the early 1800s [22] and is now found in 25 states in the east and south [28]. 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 [22,47]. In the northeast United States, princesstree plantations can produce valuable high-quality wood, but in the southern region, due to the more favourable 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 [48] and is, therefore, likely to be promoted by anthropogenic disturbance. Williams [48] classified the species as a non-aggressive species, though others [49] 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 [50]. 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 [51].

2.3. *Sampling point selection*

Sampling points were selected using the stratified spatial balanced sampling design, Generalised Random Tessellation Stratified (GRTS) [52]. Generalised Random Tessellation Stratified (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 100 sites. Site selection was stratified by years since reclamation: >20 years, 10–20 years and <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 to gain more information about the species preferences. As invasive plants are often a rare or clustered event, this approach allows for greater efficacy of research resources by ensuring effort is targeted to where the plants are located [53,54].

2.4. *Field sampling*

Field sampling occurred from June through October 2010. One hundred and twelve 405 m^2 (1/10-acre) circular plots were sampled. GPS coordinates, date, time, forest type (pine, mixed or hardwood), regeneration type (natural or planted), distance to established forest and forest age were recorded on each plot. All trees with $\geq$25 mm diameter at breast height (DBH, ca 1.4 m above ground level) were recorded for species and categorical DBH (25–75 mm, 75–150 mm, 150–225 mm, 225–375 mm or >375 mm) to assess habitat structural diversity. These categorical groupings were later reduced to three, small (DBH 25–75 mm), medium (75–225 mm) and large (>225 mm). An increment borer was used to obtain a tree core from the largest accessible tree in each plot. Two circular subplots of 1.8 m radius were established 3.7 m north and south of the main plot centre for assessing percentages of overstory, midstory and understory cover (0–1 m) [55] 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 densitometer was used to determine the cover of the forest canopy within each of these subplots, and 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 the north and south edge of each subplot (four readings per plot). After removing the leaf litter from the soil surface, soil samples were taken with a hand-held-probe 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 2 mm stainless steel sieve into plastic bags and stored until soil analysis was undertaken. If any invasive plant species was detected, an additional four neighbouring 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 m 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 were recorded for that additional plot.

2.5. Soil analysis

Soil pH was measured in water at a soil to solution ratio of 1:2. The pH reported was temperature compensated at 25°C. Total C, N and S in the soil were determined using the dry combustion method with a vario Max CNS analyser (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 [56], while macronutrients (K, Ca, Mg, P and Na) were extracted using Mehlich 3 solution [57] and analysed using inductive couple plasma spectroscopy (ICP-OES, Perkin Elmer, Massachusetts, USA). Inorganic ammonium and nitrate content in the soil were extracted with two *M* KCl and analysed using ammonium-nitrate analyser (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.

2.6. Data analysis

Habitat data were analysed in three groups: soil characteristics, ground variables (from soil to understory) and forest structure (above understory) (Table 1). Soils nutrient variables were standardised 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 [58,59], 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 group were assessed to exclude the variables with high correlation ($r^2 > 0.50$) from further analysis. The selection among highly correlated variable was based on the relative easiness for field application. All non-correlated variables were tested for any underling spatial autocorrelation in their structure that may relate more to spatial patterns than the ecological relevance of variables. Mantel test was used to measure spatial dependence among the samples [60]. If any variables had an r^2 greater than 0.1, they were explored further to assess the impact of spatial autocorrelation on the analysis.

Table 1. Habitat variables measured at each sampling plot.

	Unit	$r^2 < 0.50$	Mean	SD	Min	Max
pH		X	5.55	0.70	3.89	7.12
Phosphorus	ppm	X	10.2	6.5	1.8	34.9
Potassium	ppm	X	163	86	14	440
Sodium	ppm	X	36	16	6	104
Magnesium	ppm	X	249	165	16	746
Calcium	ppm		814	630	42	2468
Iron	ppm	X	192	93	16	447
Zinc	ppm	X	5.4	4.4	0.5	23.7
Copper	ppm	X	2.8	2.1	0.3	10.7
Manganese	ppm	X	99	65	5	340
Calcium magnesium ratio		X	4.6	7.3	0.5	54.8
Ammonium	ppm	X	11.4	5.8	2.9	43.2
Nitrate	ppm	X	6.9	7.3	0	36.4
% Carbon	%		2.0	1.4	0.1	6.2
% Nitrogen	%	X	0.13	0.08	0.01	0.37
% Sulphur	%	X	0.06	0.09	0.00	0.48
Carbon nitrogen ratio		X	14.7	4.7	5.8	25.7
Cation exchange capacity		X	11.6	3.6	2.5	21.5
% Base saturation	%		45	30	2	137
% Understory	%	X	59	26	0	100
% Rock	%	X	4	9	0	70
% Bare soil	%	X	9	15	0	80
% Non vascular plants	%	X	3	6	0	40
DWD	%	X	8	12	0	80
% Shale	%	X	6	14	0	88
% Leaf litter	%		51	39	0	100
% Grass litter	%	X	13	21	0	95
%Total litter	%	X	63	32	0	115
Litter depth	cm	X	1.8	1.3	0	8.0
Humus depth	cm	X	0.8	1.0	0	5.6
Richness			5	4	0	23
Shannon			0.76	0.67	0	2.61
Simpson's evenness		X	0.39	0.30	0	1
Hardwood richness			3	4	0	21
Oak richness			0.4	1.2	0	7
Densiometer	%	X	49	33	0	96
% Overstory	%		26	29	0	95
% Midstory	%	X	23	26	0	100
Number of stems per plot		X	52	54	0	388
Number of small stems 25–75 mm			28	36	0	242
Number of medium stems 75–225 mm			20	24	0	167
Number of large stems greater than 225 mm			3	5	0	34
Basal area of trees greater than 150 mm	m^2/ha	X	43	53	0	303
Number of pine stems			36	45	0	388
Number of small pine stems 25–75 mm		X	17	29	0	235
Number of medium pine stems 75–225 mm		X	17	23	0	167

(*continued*)

Table 1. (*Continued*).

	Unit	$r^2 < 0.50$	Mean	SD	Min	Max
Number of large pine stems >225 mm			3	5	0	34
Basal area of pine trees >150 mm	m^2/ha		39	52	0	301
Number of hardwood stems		X	15	32	0	230
Number of hardwood stems 25–75 mm			11	23	0	203
Number of hardwood stems 75–225 mm			4	9	0	65
Number of hardwood stems >225 mm			0.4	1.3	0	10
Basal area of hardwood trees >150 mm	m^2/ha	X	5	15	0	104
Number of heavy seeding hardwood stems			3	11	0	131
Basal area of heavy seeding hardwood trees >150 mm			2	9	0	86
Forest age	years	X	13	7	0	50

Note: X identifies variables with low correlations that were used for further analysis.

The relationship between habitat variables and the invasive communities were initially assessed using canonical correspondence analysis (CCA). Invasive plant species that were observed at less 5% of the sites times were excluded for CCA [61,62]. The relationship between the invasive communities was assessed with each of the three groups of habitat variables separately. An overall CCA was then conducted, using the variables that had the strongest associations ($r^2 > 0.30$) based on the three habitat group CCA.

Logistic regression was used to build occurrence predictive models. Logistic regression is a generalised 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 [63]. As with CCA, each habitat group was first analysed 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 occurred in $\geq$50 sampling plots to assure balance in the number of absences and presences (suggested ratio 2:8) in the data [64]. 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 each species. For descriptive purposes, percent contribution and direction of variables were tabulated. Percent contribution was calculated using the Wald chi-squared statistic, dropping the intercept Wald chi-square and standardising 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 maximising specificity and sensitivity [65]. 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 [66].

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 1000 re-sampling runs were conducted. If the mean p-value of a variable from the re-samples was greater than 0.15, the variable was dropped [67]. It is expected that these models have weaker relationships as the number of data points has been substantially reduced, thus a higher p-value was used. Standard deviation and 99% confidence limits were calculated for each variable in the final model based on re-sampling runs.

3. Results

A total of 374 plots were sampled, 112 main plots and 262 adaptive plots. Average forest age was 13 $\pm$ 7 years. The ground cover was variable, though predominantly leaf litter in composition, averaging 63% $\pm$ 32 leaf litter coverage. The predominant herbaceous species was Chinese lespedeza. Understory cover was high at 59% $\pm$ 26 with midstory averaging 23% $\pm$ 20. The sites varied in forest composition from no tree cover to even-aged pine stands to mixed-species of varying ages. Basal area for all stems 25 mm and greater across all sampling plots averaged 43 $\pm$ 53 m^2 ha^{-1}. 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 2468 (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 base saturation ranged from 2% to 137% (Table 1). Spatial autocorrelation as measured by Mantels test was low with all variables having an r^2 of less than 0.01.

The CCA of soil variables with the invasive plant community illustrated that autumn olive and princesstree were 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 (Figure 2a). The first two CCA axes with soil features explained 12% variation within the invasive community (Figure 2a). 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 2b). The first two CCA axes of ground variables explained 9% variation within the invasive community (Figure 2b). The CCA of forest variables with the invasive plant community showed stronger associations, which included 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 2c). The first two CCA axes of forest structure variables explained 18% variation within the invasive community (Figure 2c). The first two axes of CCA with selected variables combined from three habitat variable

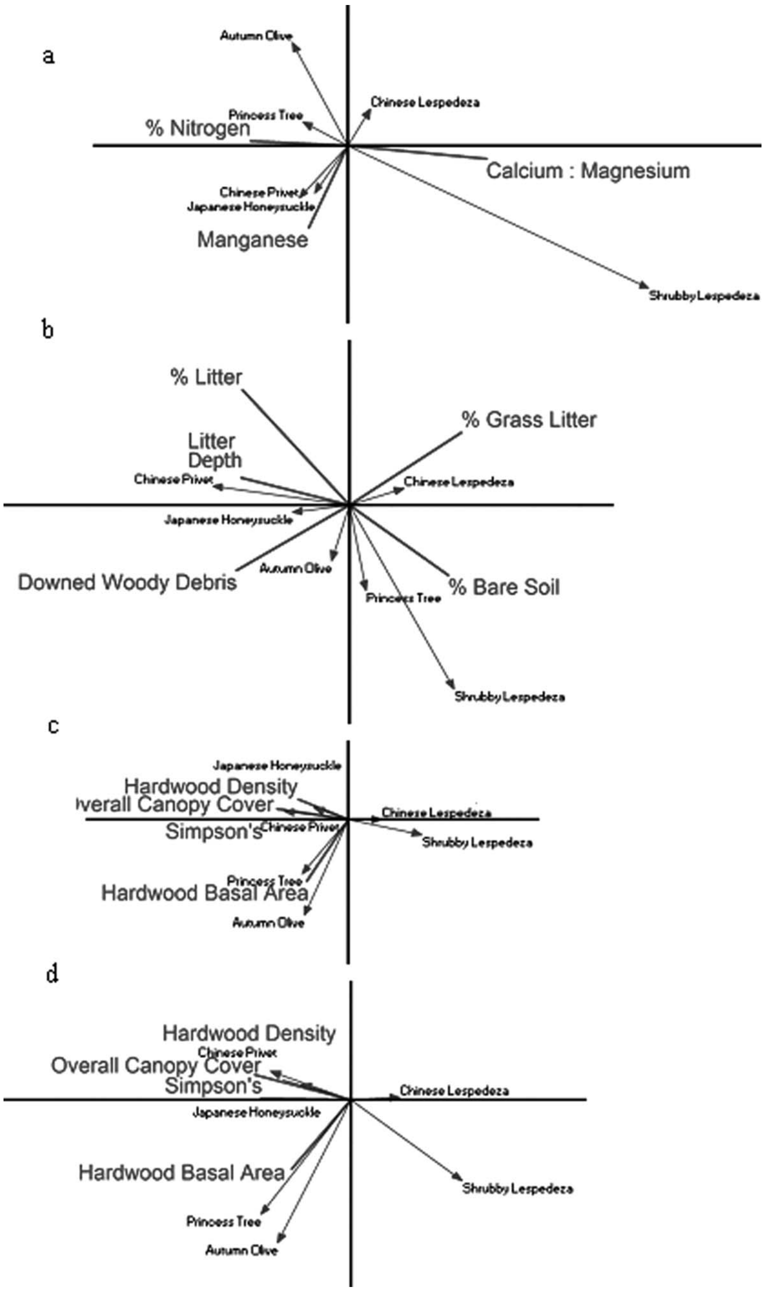

Figure 2. Relationship between habitat variables and the invasive community as assessed through canonical correspondence analysis (CCA), a – soil features (axis 1 = 7, axis 2 = 5), b – ground (axis 1 = 7, axis 2 = 2), c – forest structure (axis 1 = 12, axis 2 = 4) and d – all habitat variables combined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

sets explained 19% of the invasive community variation (Figure 2d). Overall, forest structure variables had the only strong correlations with the invasive plant community; and followed the same pattern as with the forest CCA.

Logistic regression was applied to three invasive species that occurred at 50 or more sampling sites: Chinese lespedeza, Japanese honeysuckle and Chinese privet, using habitat variables selected with limited correlation (Tables 2–4). The regressions that used soils data included seven soils variables (Table 3), with no variable dominating all the 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 3). Of the 10 forest composition variables, four were used for the logistic regression, with canopy cover dominating models (Table 3).

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 these 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 4). These variables were dropped and the models were rerun.

Table 2. Summary statistics of three invasive species from three logistic regression sub-models (soil, ground and forest), combined models and final model (variables that remained stable). Max SS is the threshold where sensitive plus specificity is maximised, false omission rate is FN/(FN + TN) and 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
	Max SS 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 significant variables for three invasive species from three logistic regression sub-models (soil, ground and forest), combined models and final model with only variables that remained stables over re-sampling. 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	Cation exchange capacity		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

Chinese lespedeza had a positive relationship with soil magnesium and negative relationships with downed woody debris, midstory cover and hardwood density. This suggested that 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. 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.

Japanese honeysuckle had a positive relationship with canopy cover, soil magnesium and Simpson's diversity index and a negative relationship with midstory; canopy cover was the most important variable and contributed about 41% of the predictive power (Table 3). Japanese honeysuckle was found in high canopy cover with little midstory and in areas of high soil magnesium and higher diversity. There was more than a 60% decrease in error from random, suggesting this model is useful

Table 4. Summary of re-sampling of final logistic model for 100 observations run 1000 times. Variable contribution, direction, 99% confidences limit, standard deviations and mean p-value of the 1000 re-sampled models are given.

		Contribution			P-value
		Mean of re-samples	99% confidence limit	SD	Mean of re-samples
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

in assessing habitat characteristics that are influencing the occurrence of Japanese honeysuckle (Table 2). Chinese privet had one prominent model variable, a positive relationship with canopy cover (Table 3). There was more than a 50% decrease in error from random, suggesting that this model is useful in assessing habitat characteristics that are influencing occurrence of privet, even with only one variable (Table 3).

4. Discussion

Surface Mining Control and Reclamation Act (SMCRA) mandates that mined land be reclaimed and restored to its original use or a use of higher value. This includes ecosystem functions and services, and an integral part of these are the distribution and diversity of the plant species. Restoration assessment often focuses more on the easily measurable restoration of edaphic and hydrological systems. However, these often do not reflect the recovery of the pre-mining biological communities or mitigate landscape, structural and ecological changes [68]. Most legislation mandates the evaluation of land reclamation success using readily quantifiable metrics with land assessed after a relatively short-time period [69]. This encourages reclamation approaches that address the short-term goals of providing erosion control and minimising 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 conflict [2]. Many mine reclamation efforts focus on establishing rapid-growing alien species that control erosion but may slow or prevent the establishment of later-successional, native species [2]. For example, a general practice creates piles of soil that are then graded to a smooth condition to stabilise the surface and prevent erosion, and these sites then are revegetated by hydroseeding with a mixture of herbaceous seeds (mix of grasses and legumes) with fertiliser [8]. This can encourage dense herbaceous vegetation that in turn can negatively affect establishment of native trees and success of planted seedlings [70].

Within the study area, the overall 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.

Forest characteristics dominated both the CCA and regression models. Canopy cover, basal area, age, Simpson's index, midstory and hardwood density were the most useful environmental variables. Four of the species were strongly associated within the community analysis, Chinese privet, autumn olive, princesstree and Japanese honeysuckle, suggesting similar habitat preferences.

Species-by-species models for the three with sufficient data revealed some differences. 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 prevalent throughout the SHR, having been widely planted and then dispersed. Its high tolerance for a wide variety of habitats [22] 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 [22]. 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 has been widely planted for deer and cattle forage [31,32] and is now considered naturalised in upland and lowland forests as well as in forest-edge habitats [32,33]. It is not as detrimental as some of the other alien species, but it has been shown to impact even-aged pine regeneration when established at high densities. In this study, Japanese honeysuckle was found in areas with high canopy cover with little midstory, low density of hardwoods and in areas of high soil magnesium and higher diversity.

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 [37]. 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 [39]. In this study, Chinese privet was found in high canopy cover areas, however the model was not strong, suggesting there are other factors influencing its distribution.

The influence of planting alien, invasive species in this area is likely the major driver of the high diversity of invasive plants, with three of the six dominant species being planted as part of the reclamation plan. Adjusting the reclamation plantings to native species would aid in resolving this. In terms of the impact, these species are having on the reclamation and productively of the land, further study needs to be undertaken. Of the three most dominant species, one is planted and another is ubiquitous throughout the region at low densities. The third species, privet, is the most concerning. Overall, it appears that the initial reclamation efforts, apart from the planting of invasive species, are not the major driver impacting the alien, invasive species composition of the reclaimed, now forested mine sites.

Acknowledgements

We wish to thank the United States Office of Surface Mining for supporting this work (OSM cooperative agreement S09AC15438). Also we would like to thank Shelly Baltar (AAMU), Matthew Carr (USFS), Ryan Sisk (USFS) and Dana Virone (AAMU) for their assistance in the field and with data processing and Clinton Patterson (AAMU) for assistance with statistical analysis.

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