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RESEARCH ARTICLE

Mine Reclamation Practices Influence Emerging Plant Communities in Appalachia, USA

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Abstract
Coal surface mines disturb forests in Appalachia, USA, and many miners use reclamation practices intended to reestablish native forests. We investigated influences of soil construction, soil grading, and herbaceous seeding on postmining plant communities on an Appalachian mine. Two soil grading (smooth and loose) and three herbaceous seeding (conventional, tree compatible, and low competition annual ryegrass) treatments were established in all combinations on two reclamation areas with differing soil types, and all areas were planted with native trees in 2008. Living trees and understory plants were tallied and measured in 2014. Larger native trees were found on annual ryegrass- and tree-compatible-seeded areas. Native understory plant cover was greater on the annual ryegrass-seeded areas than on conventionally seeded areas. Where soils were of weathered sandstone origin and mildly acidic, such as native soils, native trees grew larger and established greater canopy than in areas with alkaline-siltstone-origin soils, where primary plant community components were exotic. Soil material selection and reclamation seeding influenced the developing plant communities, as expected. Native trees established and grew where these practices were applied in accord with restoration best practices. Native understory plants established and proliferated in association with the native tree canopy. Exotic plants with potential to persist were prominent in all reclamation areas, regardless of reclamation practices; hence, plant community restoration may also require exotic plant control.

Keywords Coal · Exotic plants · Native plants · Reforestation · Restoration

Reclamation Highlights:
- Native tree survival and growth are improved with less competitive groundcover.
- Of the variables evaluated in this study, selection of favorable mine soil materials was most influential in improving native tree survival and growth.
- Invasive species are less abundant where native trees are well established.

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Introduction

Forest ecosystems are being transformed by anthropogenic activity to other land cover types worldwide (Hansen et al. 2013). Transitions of forest to nonforest land cover affect climate, alter numerous ecosystem services provided by forest plant communities (Shvidenko and Gonzalez 2005), and result in the proliferation of exotic plants (Vilà et al. 2011). The Appalachian region of the eastern United States hosts some of the most diverse nontropical and extensive temperate forests in the world (Ricketts et al. 1999; Riitters et al. 2000). Throughout the Appalachians, extensive forest losses have occurred as a result of coal surface mining (Drummond and Loveland 2010), a dramatic form of landscape disturbance that often does not enable forest recovery via natural processes within decadal or longer time scales (Zipper et al. 2011a). Mining-related forest losses are directly linked with alterations of ecosystem structure and function in Appalachian areas (Simmons et al. 2008; Wickham et al. 2013).

In Appalachia, as elsewhere, new mine reclamation methods based on rigorous scientific studies are being developed to restore native plant communities following surface coal mining (Burger et al. 2005; Zipper et al. 2011b). These methods seek to establish soil conditions favorable to native vegetation and natural successional processes and to establish tree species that are prominent in native forests including those that are slow to disperse across landscapes naturally. Methods for reestablishing native plant communities on mine sites in other world regions rely on similar approaches (Bell 2001; Bradshaw 1983; Cooke and Johnson 2002; Grant et al. 2007; Parrotta 2001, Macdonald et al. 2015).

Use of native soil, "weathered" geologic materials, with physical and chemical properties similar to native soils, or both are prescribed for soil construction to improve native plant establishment (Zipper et al. 2013). Excessive grading compacts soils, hindering water infiltration, soil-air exchange, and root growth (Andrews et al. 1998; Torbert and Burger 2000; Torbert et al. 1988); thus, grading procedures that minimize soil compaction are recommended (Burger et al. 2005). Fast-growing herbaceous plants compete with tree seedlings for resources; seeding herbaceous species that will provide adequate groundcover to control erosion, but not compete with establishing trees, is prescribed (Franklin et al. 2012). Underlying these methods is the expectation that native flora will volunteer in the reclamation area via natural processes such as seed transport by wildlife and wind; however, it is also possible that exotic and invasive plant species will invade and establish (Brenner et al. 1984). Franke et al. (2019) have shown that soils and seed mixes that are preferable for native tree establishment may also favor establishment of the Eurasian invasive species autumn olive (*Elaeagnus umbellata*). Although soil selection, grading, and seeding practices are known to influence reforestation success, few studies have assessed their influences on postmining plant communities when applied operationally by mining firms.

We assessed reclamation areas on an Appalachian mine site that were planted with native trees following differing combinations of soil selection, grading, and seeding to determine how these practices influenced postmining plant communities over 6 years.

Methods

Reclamation Treatments

An experiment was established on a surface mine in Wise County, VA, in late 2007 and early 2008 (Fields-Johnson et al. 2012). Two grading and three seeding treatments were established in all combinations on two reclamation areas (Blocks 1 and 2; Figure 1). Block 1 was at approximately N 37.006047, W 82.713649. Block 2 was at approximately N 37.008729, W 82.699239. The treatments were also applied to a third reclamation area that was destroyed by subsequent mining. However, in large-scale reclamation, valuable information can still be attained even if experimental replication is limited (Hurlburt 1984). Treatment plots averaged 0.4 ha in size. Mine soil construction was conducted while endeavoring to maintain constant slope, aspect, and soil conditions within each block. Mine soils in Block 1 were constructed from a mixture of weathered and unweathered sandstones on generally south-facing slopes that averaged 58%, whereas those of Block 2 were constructed with a spoil mixture comprised predominantly of unweathered siltstones on east-facing slopes that averaged 48% (Fields-Johnson et al. 2012). Initial soil data from the 12 treatment plots are presented in Table 1.

The two grading treatments were (1) smooth grading with backblading (dragging the dozer blade across the soil to create a smooth surface; SG) and (2) loose grading with a single dozer pass while leaving a rougher surface (LG). Each grading treatment was applied over three adjacent treatment plots within each block (Figure 1). Three seeding treatments were applied within each grading area of each block: (1) conventional herbaceous species intended to create dense groundcover rapidly (CON), (2) species intended to create a moderate level of initial groundcover (tree compatible [TC]), and (3) annual ryegrass (*Lolium perenne* subsp. *multiflorum* L.) intended to create the lowest level of groundcover by seeded species (AR; Table 2). The CON seeding is commonly applied on coal mining sites in the area. Seed was applied by a commercial hydraulic seeding contractor under supervision by the mining firm. These

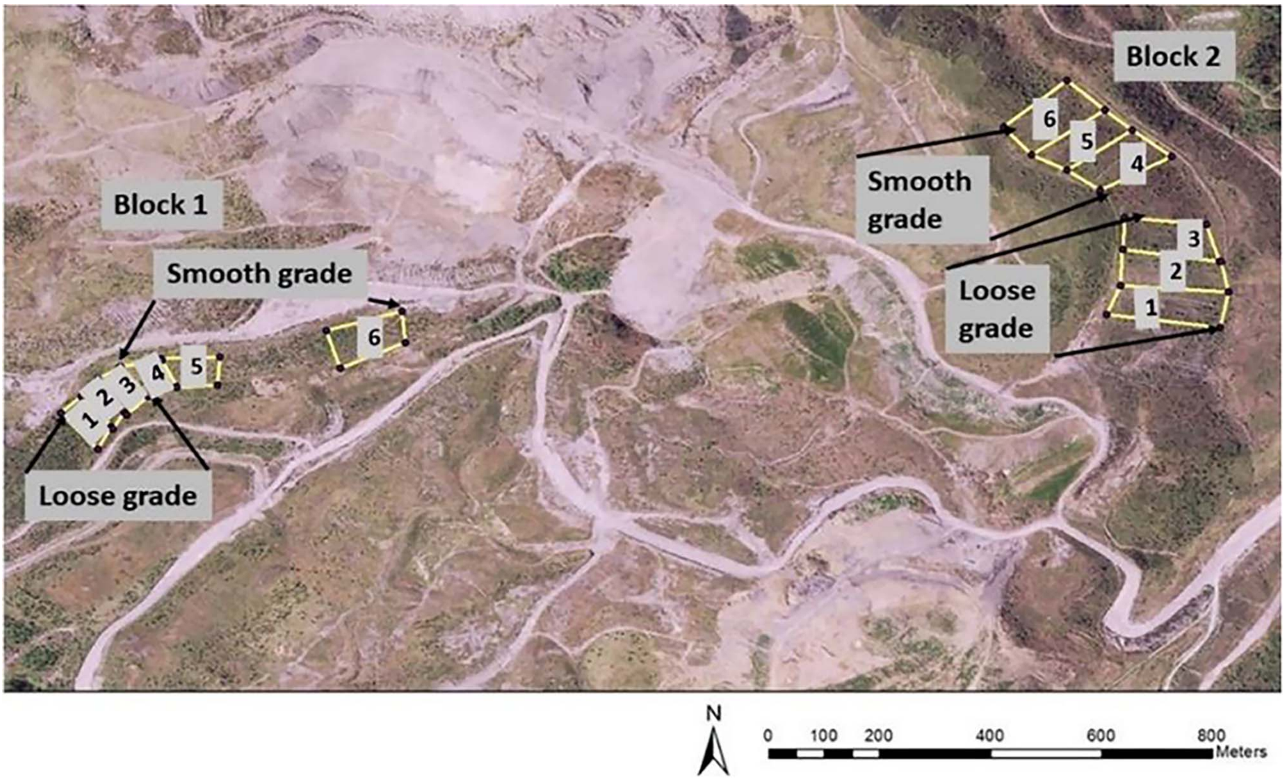

Fig. 1 Blocks 1 and 2 grading and seeding treatment layout. Plots 1 and 6 = AR; Plots 2 and 4 = CON; Plots 3 and 5 = TC. Seeding prescriptions described in Table 2

grading and seeding combinations produced six treatment plots within each block and 12 treatment plots in total.

Following seeding, the areas were hand planted with native trees with 1-year-old, bare-root seedlings in early 2008 (Table 3) and partially replanted in early 2009 (Fields-Johnson et al. 2012). All areas were planted with species intended to grow into timber-producing trees (crop trees) and with other species intended to establish rapid canopy cover and attract wildlife prescribed at lower rates.

Table 1 Soil data from spring 2008 in all 12 treatment plots

Block	Grading	Seeding	pH	Soluble salts (mg/kg; calculated from SC)	Organic matter (%; loss on ignition)
1	LG	AR	5.96	269	1.3
		CON	5.52	602	1.2
		TF	5.51	909	1.2
	SG	CON	4.59	563	1.2
		TF	5.8	973	1.2
		AR	6.93	115	1.5
2	LG	AR	7.93	218	1.6
		CON	8.1	218	1.4
		TF	7.46	230	2
	SG	CON	7.21	627	1.8
		TF	7.2	218	2.3
		AR	6.76	384	1.8

Vegetation Sampling

Five 0.02-ha, circular tree sampling plots (tree plots) were established on each of the 12 treatment plots after the initial tree planting and were the same plots sampled in this study (Fields-Johnson et al. 2011). Tree sampling plots were placed such that they would not cross treatment plot boundaries, would not intersect each other, and would be representative of top-to-bottom slope positions. Plots centers were selected and marked before vegetation establishment so that plot placement remained unbiased. Within each tree sampling plot, height (h) was measured for each tree as vertical distance from soil surface to highest live bud; and basal diameter (d) was measured. For each tree, volume index (VI) was calculated as $h*d^2$.

Tree community metrics evaluated included numbers of living trees (density); size metrics (mean basal diameter, height, and VI); and the VI sum for all recorded trees, calculated as a cumulative biomass indicator. Tree community metrics were calculated for each tree plot and for each treatment plot, overall and by taxon. Density and VI sum were normalized to ha^{-1} bases. Ash (*Fraxinus* spp.) trees were classified above the species level, considering

Table 2 Prescribed species and application rates for seeding treatments and soil amendments

	Seeding treatment and species and amendments	Rate (kg ha^{-1})
CON	Cereal rye (*Secale cereale* L.)	34
	Orchardgrass (*Dactylis glomerata* L.)	22
	Perennial ryegrass (*Lolium perenne* L.)	11
	Korean lespedeza (*Kummerowia stipulacea* (Maxim.) Makino)	6
	Birdsfoot trefoil (*Lotus corniculatus* L.)	6
	Ladino clover (*Trifolium repens* L.)	6
	Redtop (*Agrostis gigantean* Roth)	3
	Weeping lovegrass (*Eragrostis curvula* (Schrad.) Nees)	2
TC	Annual ryegrass (*Lolium perenne* L. subsp. *multiflorum* (Lam.) Husnot.)	22
	Perennial ryegrass (*Lolium perenne* L.)	11
	Timothy (*Phleum pretense* L.)	6
	Birdsfoot trefoil (*Lotus corniculatus* L.)	6
	Ladino clover (*Trifolium repens* L.)	3
	Weeping lovegrass (*Eragrostis curvula* (Schrad.) Nees)	2
AR	Annual ryegrass (*Lolium multiflorum* Lam.)	22
Soil amendments[a]	Fertilizer nitrogen	22
	Fertilizer phosphorus	68
	Fertilizer potassium	18
	Wood cellulose fiber mulch	1,680

[a] Amendments applied by hydraulic seeding with the seed for all treatments.

that both planted and non-planted species of this taxa occur and disperse in the study area. Each taxon was classified as native or exotic following USDA (2024), with *Fraxinus* classified as native. Although classified by US Department of Agriculture (USDA 2024) as a shrub, autumn olive was tallied with trees because individuals of this species were among the largest plants recorded. Stem counts were used to calculate Shannon-Wiener diversity (H′) for all tree plots.

Table 3 Tree planting prescriptions for Blocks 1 and 2

Tree type and species	Planting density (trees ha^{-1})
Crop trees	
White ash (*Fraxinus americana* L.)	205
White oak (*Quercus alba* L.)	205
Sugar maple (*Acer saccharum* Marsh.)	205
Black cherry (*Prunus serotina* Ehrh.)	205
Red oak (*Quercus rubra* L.)	205
Chestnut oak (*Quercus prinus* L.)	205
Black oak (*Quercus velutina* Lam.)	205
Yellow-poplar (*Liriodendron tulipifera* L.)	124
Wildlife and nurse trees	
Gray dogwood (*Cornus racemosa* Lam.)	54
Red mulberry (*Morus rubra* L.)	25
Redbud (*Cercis canadensis* L.)	54
White pine (*Pinus strobus* L.)	91
Shagbark hickory (*Carya ovata* (Mill.) K. Koch)	62

Four 1-m^2 understory vegetation sampling plots were nested within each tree plot as in the initial study (Fields-Johnson et al. 2011). All vegetation not tallied as trees was considered as understory. During August 2014, each observed plant type within each understory plot was identified to the lowest possible taxonomic level. Percent cover of species was used to calculate the H′ value for all herbaceous plots. Ocular estimates of total living understory groundcover, expressed as a percentage of each understory plot's ground area, were made for the full understory community and for each taxon. For the individual taxon estimates, overlapping taxa were tallied separately such that the sum of individual estimates may exceed the total living understory groundcover estimate. Canopy cover by trees was also estimated for each understory plot by using a spherical densiometer.

Understory cover by taxon was summed for each treatment plot and divided by the number of understory sampling areas to obtain treatment-plot means. Each taxon was assigned to several classes. Species included in any seeding mix (Table 2) were classified as "seeded" for all measurement areas. Crown vetch (*Securigera varia* (L.) Lassen) was seeded inadvertently as a tank contaminant in Block 1, tallied only in Block 1, and considered as seeded in data analysis. Taxa were classified as "native" or "exotic" (USDA 2024). Plant species identified as invasive by

Virginia Department of Conservation and Recreation (Virginia Invasive Plant Species list; Virginia DCR 2024) were defined as invasive for this study.

In total, 60 tree plots (five per treatment plot) were located and sampled in April 2014. When possible, the locations sampled by Fields-Johnson et al. (2012) were used. In August 2014, one tree plot in the densely vegetated Block 1 could not be located, so 59 tree plots were sampled for understory vegetation and for soils.

Soil Sampling and Testing

Soils were sampled in August 2014. Four sampling points were located within each tree plot, each approximately 10 m from the center point: upslope, downslope, and on either side. At each sampling point, surface (upper 5 cm) and subsurface (approx. 10–20 cm) subsamples were obtained, and the four subsamples were composited for each depth. Fines (<0.2 mm) were analyzed by the Virginia Tech Soil Testing laboratory (Maguire and Heckendorn 2009). Organic matter was measured as loss on ignition; thus, the measured quantities likely reflect combustion of coal-like mineral particles as well as conventional soil organic matter. Specific conductance (SC) was measured on the supernatant of a 1:2 soil:water solution (vol/vol). Soluble salts were calculated from the following formula: ppm soluble salts = SC $\times$ 6.4 $\times$ 2 (Maguire and Heckendorn 2009). Surface and subsurface parameters were averaged for data analysis.

Statistical Analysis

Vegetation and soil data were analyzed using SigmaPlot 14.5 (Systat Software, Inc. 2020). Grading treatment, seeding treatment, and block effects were compared using three-way analysis of variance (ANOVA). All ANOVA results were inspected for normality using the Shapiro-Wilk test and for equality of variance by using the Brown-Forsythe test and transformed as needed. Where block, grading, or seeding effects were found to be significant, comparisons were performed using the Holm-Sidak test. Some parameters (seeded herbaceous cover, exotic tree density, and exotic tree volume) could not be normalized with any transformation. In these cases, data were analyzed for main treatment effects with a Kruskal-Wallis one-way ANOVA run on ranks. Where seeding treatment effects were found to be significant, comparisons were performed using Tukey's honestly significant difference test or Dunn's method, as appropriate.

Table 4 Soil pH, soluble salts, and organic matter content

Assessment area	Soil pH (mean ± SE)	Soluble salts (mean ± SE; mg kg^{-1})	Organic matter (%)
Treatment effects[a]			
LG	7.0 ± 0.1a	93 ± 5	2.9 ± 0.2b
SG	6.7 ± 0.1b	90 ± 5	3.5 ± 0.2a
AR	7.0 ± 0.2	90 ± 7	2.8 ± 0.2b
TC	6.8 ± 0.1	104 ± 7	3.8 ± 0.2a
CON	6.7 ± 0.2	81 ± 7	2.9 ± 0.2b
Block effects			
Block 1	6.2 ± 0.1b	98 ± 5	3.3 ± 0.2
Block 2	7.5 ± 0.1a	86 ± 5	3.1 ± 0.2

[a] Different lowercase letters indicate significant difference between groups ($p < 0.10$).

Results

Soils

Mean soil pH was lower in Block 1 than in Block 2 and lower within SG plots than LG plots. Soil pH did not differ among seeding treatments, but did differ between grading treatments ($p < 0.001$) and blocks ($p < 0.001$; Table 4). Soluble salts did not differ among blocks or treatments (Table 4). Mean soluble salt concentrations in 2014 (Table 4) were <140 ppm for all treatment plots and analysis areas and nominally less than levels recorded in 2008 for Blocks 1 and 2. Percent organic matter did not differ among blocks, but did differ among grading ($p = 0.009$) and seeding ($p = 0.001$) treatments (Table 4). Organic matter was higher in SG plots and highest in the TC treatment. Mean organic matter content has nominally increased by at least 100% since 2008.

Understory

In total, 73 understory taxa were recorded including the 10 planted species (Table 2) and the inadvertently seeded *S. varia*. Of these taxa, 44 were classified as native and 29 as exotic; 6 exotics were classified as invasive.

Understory richness ranged from 5 to 18 for treatment plots (Table 5). Seeded taxa (1.7 per treatment plot) constituted approximately 14% of total recorded taxa (12.8 per treatment plot), whereas native taxa (6.9 per treatment plot), on average, constituted approximately half of recorded taxa, but <30% of groundcover. Within treatment plots, invasive taxa richness ranged from 0 to 4 and averaged 1.4. Invasive-taxa richness was higher in Block 1 than in Block 2 ($p <$ 0.001). There were no main effects on richness from grading treatments.

Table 5 Main effects for mean understory groundcover and richness by taxonomic classification and total and for groundcover by tree canopies

	Understory cover (%)				Understory richness (no.)			Understory diversity (H′)	Canopy cover (%)
	Total	Seeded[b]	Native	Invasive	Total	Native	Invasive		
Treatment effects[a]									
LG	68.4	21.2	20.8	58.1	12.9	7.4	1.4	1.7	23.8
SG	75.5	13.7	19.8	68.9	12.5	6.4	1.5	1.6	22.2
AR	65.5	17.7	23.2	54.9b	12.0	7.1	1.5	1.6	26.6ab
TC	72.8	16.9	22.1	60.6b	13.8	7.4	1.6	1.8	28.4a
CON	77.5	17.8	15.6	75.0a	12.3	6.2	1.3	1.6	13.9b
Block effects									
Block 1	75.2	28.8a	27.8a	58.8	12.2	7.2	1.6a	1.6	42.1a
Block 2	68.6	6.1b	12.8b	68.2	13.2	6.6	1.2b	1.7	3.9b

[a] Different lowercase letters indicate significant differences between groups ($p < 0.05$).
[b] Intentionally seeded, including the contaminant *S. varia*, which appeared only in Block 1.

Treatment-plot mean total cover (Table 5) was 72% and ranged from 9 to 95.5%; no significant main effects were detected. On average across all reclamation areas, cumulative cover of seeded taxa and native taxa totaled 17.4 and 20.1%, respectively, whereas cover of invasive taxa totaled 63.8%. Native-taxa cover was nominally greater for AR and TC than for CON seeding, and invasive-taxa cover was greater for CON than for AR and TC seeding ($p = 0.004$). Canopy cover was highest in the TC treatment and lowest in the CON treatment ($p = 0.028$). Native taxa provided greater groundcover in Block 1 ($p < 0.001$), and invasive-taxa cover was nominally higher in Block 2. Seeded taxa provided greater cover for Block 1 ($p < 0.001$), although that is likely driven by the abundant presence of *S. varia*, which was seeded inadvertently and only appeared in Block 1. There were no main effects on cover from grading treatments.

Both total cover ($p = 0.002$) and invasive cover ($p < 0.001$) were affected by an interaction between block and grading treatment; responses of these parameters to blocks differed between grading treatment levels, and responses to grading treatments differed between blocks (Figure 2). Within Block 1, total cover and invasive cover did not differ between LG and SG, but were higher in SG plots within Block 2. Total cover did not differ between Blocks 1 and 2 within SG plots, but was higher in Block 1 within LG plots. Invasive cover did not differ between Blocks 1 and 2 within LG plots, but was higher in Block 2 within SG plots.

Total cover ($p = 0.046$), invasive groundcover ($p = 0.05$), total herbaceous richness ($p < 0.001$), seeded richness ($p = 0.002$), and herbaceous diversity ($p = 0.019$) were also affected by an interaction between block and seeding treatment (Table 6). Total cover was higher in Block 1 than in Block 2 within CON plots and higher in the TC than in the AR seeding plots within Block 2. Invasive cover was higher in Block 2 within TC plots and within Block 1 was highest in CON plots. Total richness was higher in Block 2 within TC and AR plots and within Block 2 was lowest in CON plots, likely due to the observed higher invasive cover in these plots. Seeded richness was higher in Block 1 within AR plots and higher in Block 2 within CON plots; seeded richness was lowest in AR plots in both blocks, but richness did not differ between TC and CON within Block 2. Because AR treatments consisted of one species (two in Block 1, due to *S. varia*), fewer species in the AR treatment is expected. Overall Shannon-Wiener diversity of the understory was

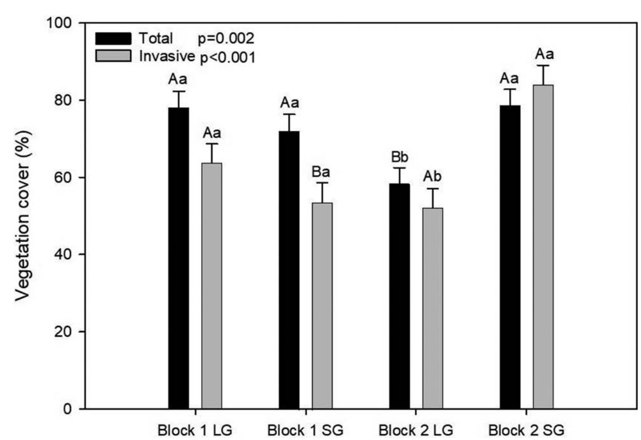

Fig. 2 Mean total and invasive cover for block and grading treatment combinations. Error bars represent SE. Different uppercase letters indicate significant differences between blocks within grading treatments. Different lowercase letters indicate significant differences between grading treatments within blocks ($p < 0.05$)

Table 6 Mean total and invasive herbaceous cover, total and native herbaceous richness, and herbaceous Shannon-Wiener diversity within block and seeding treatment combinations

Block	Seed	Cover (%)		Richness (no.)		Diversity (H′)
		Total	Invasive	Total	Native	
1	AR	72Aa[a]	53.2Ab	4.4Ba	3.0Bb	1.4Aa
	CON	85Aa	75.7Aa	5.5Aa	4.8Aa	1.8Ab
	TC	68.8Aa	47.4Bb	5.1Ba	4.2Bb	1.7Aa
2	AR	59.1Bc	56.7Aa	6.8Aa	6.0Aa	1.8Ba
	CON	70Bab	74.3Aa	5.4Ab	4.5Ab	1.4Bb
	TC	76.9Aa	73.8Aa	6.4Aa	5.5Ab	1.8Ba

[a] Different uppercase letters indicate significant differences between blocks within seeding treatments. Different lowercase letters indicate significant differences between seeding treatments within blocks ($p <$ 0.05).

similar across all treatments, but within the AR treatment, was higher in Block 2.

Trees

In total, 23 taxa (20 natives and 3 exotics) were recorded. Native-tree richness ranged from 13 to 17 species per treatment plot (Table 7) and averaged 13.9. Mean native tree richness was greater in Block 1 than in Block 2 ($p < 0.001$). Native tree density within treatment plots ranged from 150 to 4450 stems ha^{-1} and averaged 1,464 stems ha^{-1}. As with richness, native tree density was greater in Block 1 than in Block 2 ($p < 0.001$). Conversely, exotic tree density within treatment plots ranged from 0 to 300 stems ha^{-1} and averaged 39 stems ha^{-1}. Exotic tree density was lower in Block 1 ($p = 0.011$) and in LG plots ($p = 0.025$). Exotic tree density did not differ among seeding plots ($p = 0.096$); however, the TC plots had only 39% of the stem density of CON plots and 32% of the stem density of AR plots, suggesting TC seeding may somewhat restrict exotic tree establishment.

Native tree heights within treatment plots averaged 1.4 m and ranged from 0.6 to 2.5 m; mean basal diameter was 2.3 cm and ranged from 0.2 to 11.5 cm. Mean cumulative VI was 3.7 m^3 ha^{-1} and ranged from 0.01 to 17.4 m^3 ha^{-1}. Mean heights ($p < 0.001$) and basal diameters ($p <$ 0.001) were greater in Block 1 than in Block 2 (Table 7). Mean heights and diameters of native trees were greater in the AR- and TC- than in CON-seeded plots. Similarly, native tree cumulative VI were greater in Block 1 than in Block 2 ($p < 0.001$) and greater in TC- than in CON-seeded plots ($p = 0.032$).

Statistical comparisons were not performed among tree species; hence, all species-level effects discussed below may not be significant. Species-level height data are presented by block because other analyses showed block effects to be significant for many measured variables. Average heights and densities were greater for all planted species in Block 1 than in Block 2. Establishment rates for several planted species (most notably *C. racemosa*, 1,059%, but also red mulberry (*Morus rubra* L.) *Fraxinus* spp., and yellow poplar (*Liriodendron tulipifera* L.)) were >100% (meaning that tallied densities exceeded planting rates) in Block 1, whereas the establishment rate for *C. racemosa* in Block 2 (87%) was higher than for other planted species (Figure 3). All tree species were at least 50% taller in Block 1 than in Block 2, and for most species, trees were at least 100% taller in Block 1 (Figure 4).

Table 7 Treatment-plot mean richness, height, and diameter for native trees and mean density and cumulative VI for both native and exotic trees

	Native trees					Exotic trees	
	Richness	Density (trees ha^{-1})	Height (m)	Basal diameter (cm)	VI (m^{3}ha^{-1})	Density (trees ha^{-1})	VI (m^{3}ha^{-1})
Treatment effects[a]							
LG	9.6	1,525	1.5	2.0	3.8	23	0.6a
SG	8.7	1,403	1.5	2.1	3.5	55	0.9b
AR	13.5	1,330	1.6a	2.3a	3.9ab	55	1.3
TC	13.5	1,503	1.6a	2.3a	5.2a	18	0.1
CON	13.5	1,438	1.1b	1.6b	1.9b	45	0.8
Block effects							
Block 1	11.2a	2195a	2.0 a	2.6a	6.8a	22b	0.3b
Block 2	7.1b	733b	1.1 b	1.5b	0.6b	57a	1.2a

[a] Different lowercase letters indicate significant differences among treatment groups ($p < 0.10$).

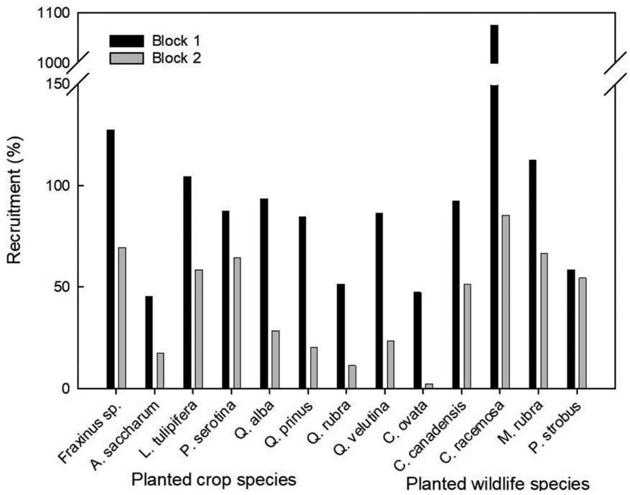

Fig. 3 Mean recruitment percentage of planted species within the two treatment blocks

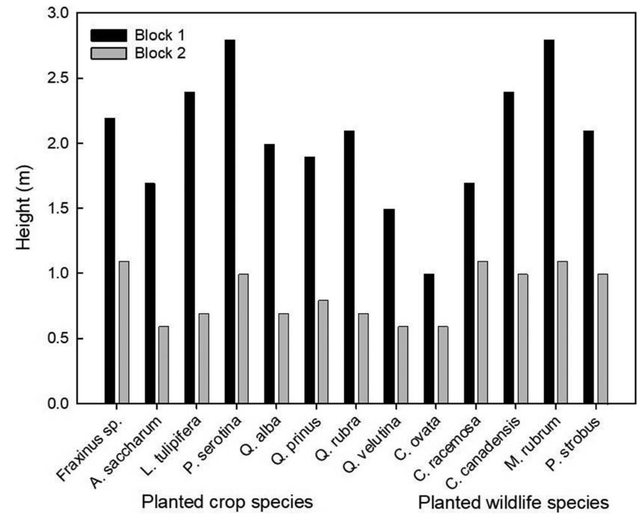

Fig. 4 Mean height of planted trees by taxon within Blocks 1 and 2

Discussion

Experimental Treatment Effects

Clear plant community differences among study areas were observed: AR-seeded areas have greater native understory groundcover and less invasive groundcover and TC-seeded areas have greater tree-canopy groundcover than CON-seeded areas. Native tree densities did not differ among seeding treatments, but native trees were larger in AR and TC than in CON seedings for most species present (Table 7). This suggests that reduced competition with understory vegetation in AR- and TC-seeded areas has enabled more rapid tree growth, consistent with prior study in the Appalachians and elsewhere (Franklin et al. 2012). Other studies have also found that the initial vegetation regime established on disturbed soils can influence the composition of developing plant communities (Körner et al. 2008; Parrotta et al. 1997).

We found few plant community differences among grading treatments, similar to Fields-Johnson et al. (2012) for this same experiment after 2 years. Previous studies have demonstrated that high soil bulk density can cause reduced survival and growth of planted trees on Appalachian mines (Andrews et al. 1998; Ashby 1997; Skousen et al. 2009; Torbert et al. 1988). Avoidance or mitigation of soil compaction has been recognized as essential to plant community restoration on mines in other world regions (Cooke and Johnson 2002; Ghose 2001; Haigh and Sansom 1999; Parrotta 2001). Our results indicate that smooth grading on these steep slopes failed to compact the soil sufficiently to cause such effects.

Understory native-taxa richness in LG areas nominally exceeded that of SG areas, suggesting that greater surface roughness may have somewhat aided native plant recruitment on LG areas. These results are consistent with other studies' findings that surface microtopography influences plant recruitment and that enhanced surface roughness can aid recruitment of volunteer plants that enter rehabilitation sites as viable seed (Bell 2001; Gilland and McCarthy 2014; Simmons et al. 2011) as live seed.

Block Effects

Sharp differences among plant community metrics were detected between the two reclamation areas. Block 1 had greater native understory groundcover, less groundcover by noxious taxa, greater native tree density, and larger native trees than Block 2.

We do recognize that there may be some effects due to aspect. However, if aspect were the sole driver of block differences, we would have expected east-facing Block 2 to outperform the south-facing Block 1 (Davis et al. 2012). The observed patterns are consistent with expectations of differential soil effects based on previous studies on Appalachian mines that have found mine soils constructed from weathered rock to be more favorable to native plants than soils constructed from unweathered geologic materials, mildly acidic mine soils to be more favorable than alkaline soils, and soils constructed from sandstones to be generally more favorable than those constructed from siltstones and shales (Emerson et al. 2009; Sena et al. 2015; Zipper et al. 2013). Research in other regions has found soil materials exert strong influence on plant community development and that use of salvaged natural soil to construct mine soils can aid restoration of native plants (Cooke and Johnson 2002; Grant et al. 2007; Parrotta 2001; Topp et al. 2010). Salvaged soils are recommended for use in mine soil construction in Appalachia (Zipper et al. 2013), but were not available for this experiment's installation.

Recruitment

Recruitment influenced plant communities in all reclamation areas. Most understory groundcover throughout was provided by nonseeded taxa, and trees of unplanted species were found throughout all experimental areas.

In Block 1, nonprescribed species were recorded at densities equivalent to 93 trees ha^{-1}; most were natives of the species sourwood (*Oxydendrum arboretum* (L.) D.C), an early successional species found as a prolific invader on other mines close to the study area (Evans et al. 2013). Similarly, other planted species were recorded with greater frequencies in Block 1 than would be expected based on prescribed planting rates. Gray dogwood (*Cornus racemosa* Lam.) was found at densities far exceeding planting rates and with many young stems (suckers) growing in association with well-established and older trees. *Fraxinus* spp., *M. rubra*, and *L. tulipifera* also were found at densities exceeding prescribed planting rates, suggesting in situ generation, off-site recruitment, or both. *Fraxinus* spp. and *L. tulipifera* occur commonly in the area's forests, including some located <1 km from the study sites, and disperse via wind-blown seed, whereas *M. rubra* can regenerate from live seed and may have reproduced from seed of trees growing in the reclamation areas.

All native understory groundcover was recruited and that recruitment was influenced by seeding treatment and varied with soil type (as block effects). Understory groundcover was found to vary with tree-canopy groundcover, with native tree VI (Figure 2), and with other native tree density and size metrics (data not shown), suggesting that conditions favorable for native tree growth are also favorable for native understory species, that tree-canopy shading aids native understory recruitment, or both. Working in eastern Kentucky, Sena et al. (2015) also found greater understory groundcover by native taxa on mine spoils that were more favorable to native trees.

Across all reclamation areas, approximately half (~46%) of cumulative understory groundcover was provided by exotic invasive taxa that were not seeded. The two most common exotic taxa on this mine site, sericea (*Lespedeza cuneata* (Dum. Cours.)) and tall fescue (*Schedonorus arundinaceus* (Schreb.) Darbysh.), are both light-demanding species (National Park Service [NPS] 2006a, b), suggesting that the canopy cover provided by trees may have aided their suppression. Others have observed negative associations of light-demanding exotics with forest tree-canopy cover in both natural and reconstructed ecosystems (DeFerrari and Naiman 1994; Iannone et al. 2016; Lemke et al. 2013; Meiners et al. 2002; Parendes and Jones 2000). The plentiful existence of propagules from surrounding areas, which are mostly reclaimed mines with relatively open canopy and where exotic plants have also become established, is another likely influence on plant communities in these reclamation areas (Charbonneau and Fahrig 2004).

Community Development

Native species richness is increasing within developing forest communities on these mine sites. We found more native herbaceous (35) and nonplanted native tree taxa (7) than were found by Fields-Johnson et al. (2012) on the same areas after 2 years (12 and 2, respectively). This pattern is consistent with observations by Holl (2002), who surveyed plant communities on mine sites ranging from 12 to 37 years of age within a few kilometers of our experimental areas and found native herbaceous and woody plant richness to have increased with mine site age.

The reclamation strategy used—selective reestablishment of species known to be important to ecosystem structure or function, but leaving many other plant species to recolonize—is used throughout Appalachia (Zipper et al. 2011b) and for ecosystem restoration more widely (Dobson et al. 1997). This strategy has been more successful in Block 1, where native trees characteristic of the local ecosystem are the dominant site occupants, than in Block 2. Although most understory vegetation remains exotic, the growth and increasing canopy of native trees observed in Block 1 suggest that such development and associated native invasion and light-demanding exotics' suppression will continue.

By contrast, the Block 2 area appears to be developing as a novel ecosystem (Hobbs et al. 2006), with nonnative plants being dominant in the sparse upper canopy and in the understory. The dominant woody plant autumn olive can form dense and persistent canopies that inhibit establishment of native trees on Appalachian mine sites (Evans et al. 2013; Oliphant et al. 2016). Although this species' short stature makes it potentially subject to replacement by taller native trees, we are not aware of such replacement being documented, much less the time that would be required. The other exotic tree present in Block 2, tree of heaven (*Ailanthus altissima* (P. Miller) Swingle), has also been observed on other Appalachian mine sites (Evans et al. 2013), and mature heights may be equivalent to native forest canopies (Knapp and Canham 2000; Webster et al. 2006), whereas the understory's most prevalent species *L. cuneata* and *S. arundinaceus* are widespread as herbaceous species, and often dominant, on mine sites that lack tree canopy cover in the Appalachian coalfield (Zipper et al. 2011a); both species are capable of suppressing invasion by native trees (Brandon et al. 2004; Rudgers et al. 2007). Multiple studies have found that exotic plant communities, once established, can persist over multiple decades without native plant community reestablishment (Hobbs et al. 2006; Kulmatiski 2006; Stylinski and Allen 1999). Mechanisms that enable exotics to suppress native community reestablishment on such areas have been described previously (Cramer et al. 2008). These include an altered physical environment that creates habitat conditions less favorable for native plants than for the exotic invaders (Didham et al. 2007) as appeared to have occurred on our experimental site, especially in Block 2. Although mine soil weathering can transform mine soils such that they come to resemble more closely native soils with time (Burger et al. 2007), soils derived from mine rocks may retain highly altered characteristics multiple years after mining has been completed (Nash et al. 2016; Simmons et al. 2008; Wilson-Kokes et al. 2013). Also, some exotic communities, once established, can transform ecosystem properties and functioning in a manner that makes the invaded site more hospitable to the invaders and less so for the natives (Coykendall and Houseman 2014; Cramer et al. 2008; Hobbs et al. 2006).

Even in Block 1 where plant communities are developing with significant native components, these data give no indication that plant communities are being fully restored. Working in this same area, Holl (2002) recorded 92 native herbs and 39 native woody species on mines and in adjacent forests, including 88 native herbs and woody species in the surveyed unmined forests, and noted that certain plant species typical of adjacent forests had not recolonized any mine sites. In addition, studies in other regions have found that unmanaged primary succession often leads to plant community outcomes that differ from those preceding disturbance (Turner et al. 1998; Walker and del Moral 2003). The exotic plants on these areas also suggest potential deviation from native-forest plant communities over extended times, as at least two of the prominent exotic species present (*E. umbellata* and *L. cuneata*), although thriving in open sun, are sufficiently shade-tolerant to enable persistence in open-forest understory (Miller 2003).

These results reinforce the importance of soil material selection and seeding practices for successful reestablishment of native forest trees on Appalachian mine sites, as has been found by previous research. Where both seeding practices and soil materials (i.e., weathered and partially weathered sandstone) were applied in accord with results of previous study, native trees established and grew well and understory communities are developing with increasing native components. However, planted trees established and grew poorly and exotic taxa dominate where unweathered alkaline spoils were used for soil construction.

Soil organic matter has been accumulating rapidly across all plots, especially in the TC and SG plots, approximately doubling to >3% overall in the 6 years from 2008 to 2014. Soluble salts have been reduced to <100 mg kg^{-1} in all plots, whereas pH has increased from 7.4 to 7.5 in Block 1 and from 5.7 to 6.2 in Block 2. The rapidity of soil genesis on these sites, which started from mine spoil parent materials devoid of actual soil and organic matter, has potentially strong implications for not only forest development but also other environmental concerns such as water quality and carbon sequestration and for the study of soil genesis in other situations beginning with bare parent material.

Plant communities in all experimental areas were characterized by significant exotic components and included species with potential to persist, leading to questions about long-term trajectories. Reestablishment of forest trees on Appalachian mines was aided by appropriate soil construction and herbaceous seeding practices, but plant community restoration would likely require additional measures, including that exotic plant controls be applied.

Future study of these sites is required to track their long-term trajectories of site productivity, species composition, succession, and recruitment of natives. It is notable that these plots were all generally well stocked with planted native trees in 2014 following poor initial survival in 2008 and supplemental replanting in 2009 with better survival that year. Initial tree planting with survival evaluation at the end of the first growing season, followed by planned supplemental planting in the next year, may improve stocking success compared with single planting events.

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References

Andrews JA, Johnson JE, Torbert JL, Burger JA, Kelting DL (1998) Minesoil properties associated with early height growth of eastern white pine. *Journal of Environmental Quality* 27:192–198

Ashby WC (1997) Soil ripping and herbicides enhance tree and shrub restoration on stripmines. *Restoration Ecology* 5:169–177

Bell LC (2001) Establishment of native ecosystems after mining—Australian experience across diverse biogeographic zones. *Ecological Engineering* 17:179–186

Bradshaw AD (1983) The reconstruction of ecosystems: Presidential address to the British Ecological Society. *Journal of Applied Ecology* 20:1–17

Brandon AL, Gibson DJ, Middleton BA (2004) Mechanisms for dominance in an early successional old field by the invasive non-native *Lespedeza cuneata* (Dum. Cours.) G. Don. *Biological Invasions* 6:483–493

Brenner FJ, Werner M, Pike J (1984) Ecosystem development and natural succession in surface coal mine reclamation. *Environmental Geochemistry and Health* 6:10–22

Burger JA, Graves D, Angel P, Davis V, Zipper C (2005) The forestry reclamation approach. Office of Surface Mining, US Department of Interior, Washington, DC. Forest Reclamation Advisory 2

Burger JA, Mitchem D, Daniels WL (2007) Red oak seedling response to different topsoil substitutes after five years. In: Barnhisel RI (ed) *Proceedings of the American Society of Mining Reclamation*, Gillette, WY, 2–7 June 2007. American Society of Mining and Reclamation, Lexington, KY

Charbonneau NC, Fahrig L (2004) Influence of canopy cover and amount of open habitat in the surrounding landscape on proportion of alien plant species in forest sites. *Ecoscience* 11:278–281

Cooke JA, Johnson MS (2002) Ecological restoration of land with particular reference to the mining of metals and industrial minerals: A review of theory and practice. *Environmental Reviews* 10:41–71

Coykendall KE, Houseman GR (2014) *Lespedeza cuneata* invasion alters soils facilitating its own growth. *Biological Invasions* 16:1735–1742

Cramer VA, Hobbs RJ, Standish RJ (2008) What's new about old fields? Land abandonment and ecosystem assembly. *Trends in Ecology & Evolution* 29:104–112

Davis V, Burger JA, Rathfon R, Zipper CE, Miller CR (2012) Selecting tree species for reforestation of Appalachian mined land. Appalachian Regional Reforestation Initiative Forest Reclamation Advisory No. 9. https://www.osmre.gov/programs/arri/publications

DeFerrari CM, Naiman RJ (1994) A multi-scale assessment of the occurrence of exotic plants on the Olympic Peninsula, Washington. *Journal of Vegetation Science* 5:247–258

Didham RK, Tylianakis JM, Gemmell NJ, Rand TA, Ewers RM (2007) Interactive effects of habitat modification and species invasion on native species decline. *Trends in Ecology & Evolution* 22:489–496

Dobson AP, Bradshaw AD, Baker AJM (1997) Hopes for the future: Restoration ecology and conservation biology. *Science* 277:515–522

Drummond M, Loveland T (2010) Land-use pressure and a transition to forest-cover loss in the eastern United States. *Bioscience* 60:286–298

Emerson P, Skousen J, Ziemkiewicz P (2009) Survival and growth of hardwoods in brown versus gray sandstone on a surface mine in West Virginia. *Journal of Environmental Quality* 38:1821–1829

Evans DM, Zipper CE, Burger JA, Strahm B, Villamagna A (2013) Reforestation practice for enhancement of ecosystem services on a compacted surface mine: Path toward ecosystem recovery. *Ecological Engineering* 51:16–23

Fields-Johnson CW, Zipper CE, Burger JA, Daniels WL (2011) Appalachian Surface Mine Reforestation Techniques: Effects of Grading, Cultural Treatments and Species Selection. Thesis submitted to Virginia Polytechnic Institute and State University.

Fields-Johnson CW, Zipper CE, Burger J, Evans D (2012) Forest restoration on steep slopes after coal surface mining in Appalachian USA: Soil grading and seeding effects. *Forest Ecology and Management* 270:126–134

Franke ME, Zipper C, Barney J (2019) Invasive autumn olive performance varies in different reclamation conditions: Implications for restoration. *Restoration Ecology* 27(3):600–606

Franklin JA, Zipper CE, Burger JA, Skousen JG, Jacobs DF (2012) Influence of planted ground cover on forest restoration of eastern US coal surface mines. *New Forests* 43:905–924

Ghose MK (2001) Management of topsoil for geo-environmental reclamation of coal mining areas. *Environmental Geology* 40:1405–1410

Gilland KE, McCarthy BC (2014) Microtopography influences early successional plant communities on experimental coal surface mine land reclamation. *Restoration Ecology* 22:232–239

Grant CD, Ward SC, Morley SC (2007) Return of ecosystem function to restored bauxite mines in western Australia. *Restoration Ecology* 15:S94–S103

Haigh MJ, Sansom B (1999) Soil compaction, runoff and erosion on reclaimed coal-lands (UK). *International Journal of Surface Mining, Reclamation, and Environment* 13:135–146

Hansen MC, Potapov PV, Moore R, Hancher M, Turubanova SA, Tyukavina A, et al (2013) High-resolution global maps of 21st-century forest cover change. *Science* 342:850–853

Hobbs RJ, Arico S, Aronson J, Baron JS, Bridgewater P, Cramer VA, et al (2006) Novel ecosystems: Theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography* 15:1–7

Holl KD (2002) The effect of coal surface mine revegetation practices on long-term vegetation recovery. *Journal of Applied Ecology* 39:960–970

Hurlburt SH (1984) Pseudoreplication and the design of ecological field experiments. *Ecological Monographs* 54:187–211

Iannone BV III, Potter KM, Hamil KD, Huang W, Zhang H, Guo Q, et al (2016) Evidence of biotic resistance to invasions in forests of the eastern USA. *Landscape Ecology* 31:85–99

Knapp LB, Canham CD (2000) Invasion of an old-growth forest in New York by *Ailanthus altissima*: Sapling growth and recruitment in canopy gaps. *Journal of the Torrey Botanical Society* 127:307–315

Körner C, Stöcklin J, Reuther-Thiébaud L, Pelaez-Riedl S (2008) Small differences in arrival time influence composition and productivity of plant communities. *New Phytologist* 177:698–705

Kulmatiski A (2006) Exotic plants establish persistent communities. *Plant Ecology* 187:261–275

Lemke D, Schweitzer CJ, Tadesse W, Wang Y, Brown JA (2013) Geospatial assessment of invasive plants on reclaimed mines in Alabama. *Invasive Plant Science and Management* 6:401–410

Macdonald SE, Landhäusser SM, Skousen J, Franklin J, Frouz J, Hall S, et al (2015) Forest restoration following surface mining disturbance: Challenges and solutions. *New Forests* 46:703–732

Maguire RO, Heckendorn SE (2009) Laboratory procedures Virginia Tech Soil Testing Laboratory. Virginia Cooperative Extension Publication 452–881

Meiners SJ, Pickett ST, Cadenasso ML (2002) Exotic plant invasions over 40 years of old field successions: Community patterns and associations. *Ecography* 25:215–223

Miller JH (2003) Nonnative invasive plants of southern forests. US Forest Service, Southern Research Station General Technical Report SRS-62

Nash WL, Daniels WL, Haering KC, Burger JA, Zipper CE (2016) Long-term effects of rock type on Appalachian coal mine soil properties. *Journal of Environmental Quality* 45:1597–1606

National Park Service [NPS] (2006a) Fact sheet: Chinese *Lespedeza*. US National Park Service. http://www.nps.gov/plants/alien/fact/lecu1.htm

National Park Service [NPS] (2006b) Fact sheet: Tall fescue. US National Park Service. http://www.nps.gov/plants/alien/fact/scph1.htm

Oliphant AO, Wynne RH, Zipper CE, Ford WM, Donovan PF, Li J (2016) Autumn olive (*Elaeagnus umbellata*) presence and proliferation on former surface coal mines in eastern USA. *Biological Invasions* 19:179–195

Parendes LA, Jones JA (2000) Role of light availability and dispersal in exotic plant invasion along roads and streams in the HJ Andrews Experimental Forest, Oregon. *Conservation Biology* 14:64–75

Parrotta JA (2001) Restoring tropical forests on lands mined for bauxite: Examples from the Brazilian Amazon. *Ecological Engineering* 17:219–239
Parrotta JA, Turnbull JW, Jones N (1997) Catalyzing native forest regeneration on degraded tropical lands. *Forest Ecology and Management* 99:1–7
Ricketts TH, Dinerstein E, Olson DM, Loucks CJ, Eichbaum W, DellaSala D, et al (1999) Terrestrial ecoregions of North America, a conservation assessment. Island Press, Washington, DC
Riitters K, Wickham J, O'Neill R, Jones B, Smith E (2000) Global-scale patterns of forest fragmentation. Conservation Ecology 4(2):3
Rudgers JA, Holah J, Orr SP, Clay K (2007) Forest succession suppressed by an introduced plant–fungal symbiosis. *Ecology* 88:18–25
Sena K, Barton C, Hall S, Angel P, Agouridis C, Warner R (2015) Influence of spoil type on afforestation success and natural vegetative recolonization on a surface coal mine in Appalachia, United States. *Restoration Ecology* 23:131–138
Shvidenko A, Gonzalez P (2005) Forest and woodland systems. Chapter 21. In: Current State, Trends Assessment. Millenium Ecosystem Assessment. http://www.millenniumassessment.org/en/index.html
Simmons J, Currie W, Eshleman K, Kuers K, Monteleone S, Negley T, et al (2008) Forest to reclaimed land use change leads to altered ecosystem structure and function. *Ecological Applications* 18:104–118
Simmons ME, Wu XB, Whisenant SG (2011) Plant and soil responses to created microtopography and soil treatments in bottomland hardwood forest restoration. *Restoration Ecology* 19:136–146
Skousen J, Gorman J, Pena-Yewtukhiw E, King J, Stewart J, Emerson P, et al (2009) Hardwood tree survival in heavy ground cover on reclaimed land in West Virginia: Mowing and ripping effects. *Journal of Environmental Quality* 38:1400–1409
Stylinski CD, Allen EB (1999) Lack of native species recovery following severe exotic disturbance in southern Californian shrublands. *Journal of Applied Ecology* 36:544–554
Systat Software, Inc. (2020) SigmaPlot Version 14.5, Palo Alto, CA. https://grafiti.com/sigmaplot-detail/
Topp W, Thelen K, Kappes H (2010) Soil dumping techniques and afforestation drive ground-dwelling beetle assemblages in a 25-year-old open-cast mining reclamation area. *Ecological Engineering* 36:751–756
Torbert JL, Burger JA (2000) Forest land reclamation. Pp. 371–398. In: Barnhisel RI, Darmody RG, Daniels WL (eds) *Reclamation of drastically disturbed lands*. American Society of Agronomy
Torbert JL, Tuladhar AR, Burger JA, Bell JC (1988) Minesoil property effects on the height of ten-year-old white pine. *Journal of Environmental Quality* 17:189–192
Turner MG, Baker WL, Peterson CJ, Peet RK (1998) Factors influencing succession: Lessons from large, infrequent natural disturbances. *Ecosystems* 1:511–523
US Department of Agriculture [USDA] (2024) Plants database. http://plants.usda.gov/
Vilà M, Espinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, et al (2011) Ecological impacts of invasive alien plants: A meta-analysis of their effects on species, communities and ecosystems. *Ecology Letters* 14:702–708
Virginia Department of Conservation and Recreation [Virginia DCR] (2024) Virginia invasive plant species list 2024. http://www.dcr.virginia.gov/natural-heritage/invsppdflist
Walker LR, del Moral R (2003) *Primary succession and ecosystem rehabilitation*. Cambridge University Press
Webster CR, Jenkins MA, Jose S (2006) Woody invaders and the challenges they pose to forest ecosystems in the eastern United States. *Journal of Forestry* 104:366–374
Wickham JD, Wood PB, Nicholson MC, Jenkins W, Druckenbrod D, Suter GW, et al (2013) The overlooked terrestrial impacts of mountaintop mining. *Bioscience* 63:335–348
Wilson-Kokes L, Emerson P, DeLong C, Thomas C, Skousen J (2013) Hardwood tree growth after eight years on brown and gray mine soils in West Virginia. *Journal of Environmental Quality* 42:1353–1362
Zipper CE, Burger JA, Barton CD, Skousen JG (2013) Rebuilding soils on mined land for native forests in Appalachia. *Soil Science Society of America Journal* 77:337–349
Zipper CE, Burger JA, McGrath JM, Rodrigue JA, Holtzman GI (2011a) Forest restoration potentials of coal mined lands in the eastern United States. *Journal of Environmental Quality* 40:1567–1577
Zipper CE, Burger JA, Skousen JG, Angel PN, Barton CD, Davis V, Franklin JA (2011b) Restoring forests and associated ecosystem services on Appalachian coal surface mines. *Environmental Management* 47:751–765