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RESTORING PLANT COMMUNITIES FOR MULTIPLE ECOSYSTEM FUNCTIONS AFTER NATURAL RESOURCE DEVELOPMENT

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ABSTRACT

Natural resource development is a significant contributor to terrestrial conversion globally. The ecosystem functions and services that are degraded or lost to extraction activities must be restored after development to regain ecological integrity in the surrounding landscape. Restoration of such highly disturbed ecosystems has historically been reactive, not proactive, and limited to site stabilization measures to control soil erosion. In addition to site stabilization, restoration must also prioritize setting the successional trajectory for plant communities that will provide for multiple ecosystem functions over time.

The development of oil and gas reservoirs with low permeability and porosity, commonly referred to as unconventional production, has only become economically feasible in the last decade. Thus far, most measures to regulate and restore the terrestrial footprint of unconventional gas development in the Marcellus and Utica Shales in the northeastern United States have been reactive and limited in scope. Here, we aim to bring attention to the limited nature of the restoration approach and governing regulations, and offer alternatives from the perspective of restoration ecology. In Chapter Two, we report on the main drivers contributing to the spread of invasive plants along shale gas infrastructure, documenting the shift in plant community composition from a largely native forest to incursions of non-native plants. In Chapter Three, we provide an assessment of the establishment success of a native perennial plant mixture on compacted soils, typical conditions of shale gas well pads, as an alternative to the widespread use of non-native, cool season grasses and legumes. In Chapter Four, we report on a floristic survey of a gas pipeline corridor seeded with a similar native perennial mixture to assess the long-term assembly of these species across an environmental gradient. Finally, to address the limited scope of reclamation regulations we outline specific language and concepts in regulations pertaining to the "earth disturbance

activities" of shale gas that are barriers to shifting the norms of reclamation to ecological restoration.

This research highlights vegetation changes within the footprint of shale gas production. Non-native plants are recruited inadvertently via the high level of anthropogenic activities on drilling sites, and directly from restoration seed mixes. We demonstrate the potential for using native plants for restoration to set the successional trajectory for restoring multiple ecosystem functions beyond erosion control. However, restoration practices are unlikely to change at a scale that matters unless regulations address the lost value of terrestrial ecosystems.

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PREFACE

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Chapter 1

Prologue: Restoring plant communities for multiple ecosystem functions and advancing regulations to meet the challenge of energy development in natural landscapes

Overview

In this body of work we aim to inform the ecological strategies needed to guide monitoring and restoration of landscapes highly disturbed by the development of unconventional oil and gas. We use the Marcellus and Utica shale gas reservoirs in the northeastern United States as a case study to describe and model the inadvertent (incursion of plants with invasive growth habits) and directed (restoration plantings of desired propagule) assembly of plant communities in forested landscapes fragmented by well pads, access roads, and pipeline corridors. Through observational studies we predict the likelihood of invasive plant spread through shale gas production infrastructure (Chapter 2). And through experimental and observational work we demonstrate how restoration practices influence the likelihood of establishing native perennial grasslands (Chapters 3 and 4). Such insights can provide information to iteratively improve restoration practices and increase the successful establishment of plant communities that can provide for multiple ecosystem functions. Yet, illuminating research alone does not lead to changes in land management. Reclamation regulations are generally limited in scope to erosion control, the necessary first step, but do not hold operators accountable to restoring additional ecosystem functions. Restoration practices will likely not attain higher standards than regulations require, as has been the case with coal mining. The limited nature of regulations is certainly not due to a lack of empirical evidence from ecological restoration research. In the final chapter (Chapter 5), we describe the limited scope of reclamation regulations governing unconventional oil and gas development and outline pathways to shift the discourse from single outcomes for off-site impacts to ecological strategies for on-site ecosystem goals.

The state of unconventional oil and gas development

Shale gas is globally estimated at 7,299 trillion cubic feet (Tcf) (AEO 2014). Commercial production of shale oil and gas became viable in the 1990s with the development of “slickwater fracturing” and horizontal drilling (Montgomery and Smith 2010). Production from shale reservoirs and other low porosity, low permeability formations are often referred to as “unconventional”, compared to the more accessible oil and gas extracted with “conventional” technologies (i.e. vertical wells). Currently, unconventional production occurs only in the United States, Canada, China, and Australia (Rivard et al., 2014; Wang et al., 2015). In the United States, the recent accessibility and demand for domestic energy sources has fueled rapid extraction from the four major shale gas reservoirs, the Marcellus, Haynesville, Fayetteville, and Barnett, which together have over 30,000 wells. The United States Energy Information Administration (EIA) has projected annual production from the four shale gas reservoirs to reach 300 billion cubic meters by 2020 and remain at this level for several decades. Recent finer resolution from the University of Texas estimates are more conservative and predict a decline after 2020 and only half the production rate predicted by the EIA in 2040 (Inman 2014).

Shale gas production footprint and ecological restoration challenges

Shale gas development in the Central Appalachian region of the United States is an important case study of intense anthropogenic disturbance in which to assess current and new restoration practices for reestablishing plant communities in fragmented forests. The shale gas surface infrastructure consists of new or widened road networks, staging pads for drilling wells, compressor stations, and pipelines. Roads and pipelines make up a large proportion of the infrastructure footprint. In one county in the state of Pennsylvania pipelines and roads accounted for 67% of the footprint compared to 23% for pads, and 10% for water impoundments and other infrastructure (Langlois et al., 2017). The infrastructure for gas production does not result in extensive direct loss of vegetation as is the case with surface mining for coal. For example, in two counties within the Marcellus and Utica footprint the direct forest loss was only 1% (Donnelly et al., 2017). But, the resulting forest fragmentation creates significant loss in core forest habitat and unknown changes in plant communities due to the likely spread of invasive species and largely unregulated restoration practices. Some applied research for pipeline corridor revegetation to support rangeland habitat has been done in the western US, such Espeland (2014) on the Bakken Formation of North Dakota and Montana, and Pawelek et al. (2015) on the Eagle Ford Shale of Texas. Additional restoration approaches that address methods to regain ecological integrity from fragmented core forests in the east are needed. Here, we assess pipeline revegetation in eastern deciduous forests in Pennsylvania across heterogenous, disturbed soil environments (Chapter 4).

Restoration of well pads face similar challenges in establishing plant materials on disturbed soils, but with the additional challenge of severe soil compaction. Constructing the surface infrastructure for shale gas production results in the loss of topsoil structure and increases soil bulk density through compaction with heavy equipment (McConkey et al. 2012), particularly on well pads. To create safe operational conditions for heavy equipment topsoil is removed with bulldozers and stockpiled on the pad edge and the subsoil is compacted up to 99% of bulk density. The stockpiled topsoil is further degraded through losses of soil carbon and nitrogen from organic matter decomposition and mineralization as soils sit in large mounds for extended periods of time (Fink and Drohan 2015). When the topsoil is replaced, the resulting 'topsoil' layer is a thin, homogenous layer, low in structure, soil carbon and nitrogen and lacking a viable forest plant community seed bank. These soil conditions pose significant challenges to restoring persistent perennial native plant communities. We assess whether or not soil compaction alter assembly dynamics by reducing native perennial grass biomass resulting in greater weedy plant growth (Chapter 3).

Wachal et al. (2008) modeled sediment yields with varying best management practices for erosion control on well pads on the Barnett Shale in Texas, highlighting the need for control measures beyond vegetative ground cover. Preston (2015) found significant recruitment of non-native plants on well pads in the Williston Basin in Montana and the Dakotas. Johnston and Chapman (2014) and Johnston (2011) assessed invasion and revegetation dynamics of simulated well pads in Piceance Basin of Colorado. In British Columbia McConkey et al. (2012) tested soil amendments and tillage for reforestation on compacted gas well pads. Long term recovery of unconventional sites could be informed by the assessments of conventional gas and oil pad sites from the 1960s to the early 2000s (McConkey et al. 2012; Minnick and Alward 2015). Our research here builds on the current knowledge of plant community changes from shale gas development.

Opportunities for advancing ecological restoration practices

Restoration of ecosystems impacted by resource extraction must provide immediate site stabilization to control soil erosion, which should be followed by measures to restore a suite of ecosystem functions and services. Resetting the successional trajectory in an ecosystem is a significant challenge that requires creating the enabling conditions for dynamic plant communities to persist over time (Hobbs and Cramer 2008). Native seed mixes commonly recommended to re-establish vegetation on well pads, roads and pipelines need to be assessed for establishment success, persistence and for utility in restoring key ecosystem functions and processes. We saw this need firsthand while conducting plant surveys documenting the extent and drivers of invasive plant spread on well pads and connecting road networks. The use of low diversity mixes of non-native cool season grasses and forbs is widespread. Despite the utility of introduced species for rapid establishment and low cost (Ewel and Putz 2004; Schlaepfer et al. 2011), they have been associated with negative changes in ecosystem processes (Simberloff 2005; Ehrenfeld 2010), and can alter local food webs (Heleno et al. 2009; Burghardt et al. 2010).

Two high priority challenges for land managers will be to establish more ecologically suited plant communities on the highly disturbed soils of well pads and pipelines, and to enable sufficient ecosystem resilience that resists plant invasions that rapidly overwhelm native habitat. We addressed these two challenges for commonly recommended native perennial mixtures in Chapters 3 and 4. On well pads our main goal was to quantify establishment on compacted soils, and on pipelines we evaluated success across a soil pH gradient. The well pad study investigates the early establishment period (1-3 years) of restoration, and the pipeline surveys helps shed light on long-term assembly.

Native grassland seeding protocols for restoration in the northeast and mid-Atlantic have come far (Miller 2013), but expanding the understanding of how these plant communities establish on disturbed forest soils is necessary at a time when shale gas disturbance is likely to expand. The added insights arising from this work coupled with on-the-ground evidence of their site specific performance in situ (Török et al. 2011) will provide evidence that could support a change in practice norms.

Monitoring for and managing invasive weeds

A persistent challenge in restoration is preventing the establishment of invasive weeds (Funk et al. 2008; Rowe 2010). Propagule pressure, in terms of number and size (Simberloff 2009), and changes in the disturbance regime of an ecosystem (Moles et al. 2012), are known factors affecting invasive plant establishment and colonization, and are relevant to shale gas development as explored in Chapter 2. The rapid pace of development, particularly in remote areas, and without vigilant monitoring and management invasive weeds could become the dominant vegetation within the shale gas footprint.

In this body of work we focused on the regionally widespread non-native invasive annual C_4 grass, *Microstegium vimineum*. Not all non-native invasive plants that spread through the gas infrastructure in our focal region will result in degradation of forest habitat. *Microstegium* *vimineum* is a priority species for monitoring as it can establish as monocultures on forest road edges and forest stream banks and openings. *Microstegium vimineum* has become a competitive dominant species in eastern forests (Adams and Engelhardt 2009; Leicht et al. 2005; Warren et al. 2010) and has established on shale gas pipelines, pads and access roads (Barlow et al. 2017).

Eradication is not feasible, but new populations can be managed if caught early. In our first study we document the extent of *M. vimineum* spread through the well pad access roads and well pad edges (Chapter 2). Hypothesizing that *M. vimineum* would be a problematic invasive for the restored plant communities we test the resistance of commonly recommended native perennial grasses and forbs for restoration plantings to *M. vimineum* incursion. For well pads we expected that the severe soil compaction would lead to reduced growth of the seeded mix leaving *M. vimineum* more resources for competitive growth and propagule for further spread. We simulated the typical soil compaction of a well pad and seeded mixtures of native perennial grasses with subplots of *M. vimineum* (Chapter 3). For pipeline corridors across heterogenous environments we expected that *M. vimineum* would be more competitive with the seeded native mix in less acidic soils (Chapter 4). In both research studies our hypotheses were not supported. The native grasses were not impacted significantly by the soil compaction and did not differ in suppressing *M. vimineum*. On the pipeline *M. vimineum* density was greatest in low soil pH. The results from these two studies provide land managers with new information on establishing native plant communities with *M. vimineum* propagule pressure.

Changing restoration regulations and norms

Intentionally aligning scientific research and regulatory policy to address the environmental impacts of unconventional gas development (Souther et al., 2014) will be critical to preserving ecosystem services given the global overlap of regions with high biodiversity and fossil fuels (Butt et al. 2013). For example, an annual average of 50,000 oil and gas wells were constructed in North America from 2000-2012 which led to the conversion of 3 million ha for infrastructure and a net primary productivity loss of 4.5 Tg C (Allred et al. 2015).

During the shale gas boom period of 'land grabs' the loss of ecological integrity from removed vegetation and soil degradation moved much faster than the efforts to implement policies and regulations to manage the accompanying ecological impacts and restoration challenges. States that lie above the Marcellus and Utica shales in the northeastern United States have taken different regulatory paths and varied in the extent of development (Evensen et al. 2013). Determining environmental impacts has been reactive in most states, not proactive, only New York and Maryland continue to maintain a ban on development for research purposes. Production from neighboring Pennsylvania, Ohio, and West Virginia provided 85% of the United States shale gas production growth from 2012 to 2015 (US EIA 2016).

Regulations governing reclamation have generally been adapted from other 'earth disturbance activities', such as conventional oil and gas and coal mining. The federal Surface Mining Control and Reclamation Act (SMCRA) set the precedence for the use of non-native cool season grasses and legumes (Franklin et al. 2012; Yeiser et al. 2015). The rapid early growth and ground cover of non-native cool season grasses and legumes are often an ideal choice when immediate soil stabilization is required (Skousen and Zipper 2010), but limit further recruitment for habitat diversity over time. Reclamation norms, in terms of seed mix components, from conventional gas extraction and coal have continued with the new unconventional gas development.

One of the main restoration challenges addressed in this body of research is the feasibility of establishing native plant communities as compared to the widespread use of non-native mixes. Stakeholder decisions on restoration practices (i.e. seed mix composition and soil amendments) are influenced by cultural norms as well as by what is required by regulations. Ideally, restoration plans should align with ecosystem functions and services appropriate for the site, rather than what is the least expensive and most convenient, or "tradition" or out of "habit". We focused in on reclamation regulations to identify why restoration practices often do not align with ecological principles (Chapter 5). From our close reading of the regulations we were able to outline new language that reorients the value of the degraded components of disturbed ecosystem. We see this as one step forward of many needed to shift restoration practice norms to align with ecological science and principles.

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Chapter 2

Unconventional gas development facilitates plant invasions

Abstract

Vegetation removal and soil disturbance from natural resource development, combined with invasive plant propagule pressure, can increase vulnerability to plant invasions. Unconventional oil and gas development produces surface disturbance by way of well pad, road, and pipeline construction, and increased traffic. Little is known about the resulting impacts on plant community assembly, including the spread of invasive plants. Our work was conducted in Pennsylvania forests that overlay the Marcellus and Utica shale formations to determine if invasive plants have spread to edge habitat created by unconventional gas development and to investigate factors associated with their presence. A piecewise structural equation model was used to determine the direct and indirect factors associated with invasive plant establishment on well pads. The model included the following measured or calculated variables: current propagule pressure on local access roads, the spatial extent of the pre-development road network (potential source of invasive propagules), the number of wells per pad (indicator of traffic density), and pad age. Sixty-one percent of the 127 well pads surveyed had at least one invasive plant species present. Invasive plant presence on well pads was positively correlated with local propagule pressure on access roads and indirectly with road density pre-development, the number of wells, and age of the well pad. The vast reserves of unconventional oil and gas are in the early stages of development in the US. Continued development of this underground resource must be paired with careful monitoring and management of surface ecological impacts, including the spread of invasive plants. Prioritizing invasive plant monitoring in unconventional oil and gas development areas with existing roads and multi-well pads could improve early detection and control of invasive plants.

Introduction

The rapid development of unconventional oil and gas (UOG) resources from low-permeability rock, including shale and tight sands (limestone and sandstone), has out-paced our understanding of the ecological impacts of its extraction (Kargbo et al., 2010; Souther et al., 2014). UOG production within the continental United States (US) is projected to continue at an annual growth rate of 4% through 2020 and 1% through 2040 largely driven by shale gas production in the East (EIA, 2017). The impact of UOG development on water resources has received much attention spanning wastewater management (Rahm et al., 2013), surface water quality (Olmstead et al., 2013; Warner et al., 2013) and flow (Drohan and Brittingham, 2012), and the potential impact on US regional watersheds (Mauter et al., 2014; Medina and Suedel, 2015). Much less attention has been focused on impacts on vegetation and wildlife (Evans and Kiesecker, 2014; Kiviat, 2013; Souther et al., 2014) including the potential spread of invasive plants.

We chose Pennsylvania (PA) state forests to assess the current state of invasive plant presence and abundance on recently established unconventional gas well pads and access roads, and to assess potential drivers of invasive spread. The second-growth forests of the mid-Atlantic US, including PA, serve as an important timber resource, sink for atmospheric carbon (McGarvey et al., 2015), watershed for major northeastern rivers, key migratory pathway for birds (Brittingham and Goodrich, 2010), and recreation resource. This region has a long history of timber, coal, iron-ore, and conventional oil and gas extraction, and now the development of deep reserves of shale gas from the Marcellus and Utica formations.

Forest fragmentation is a substantial concern where UOG reserves lie beneath core forests (Drohan et al., 2012; Moran et al., 2015; Langlois et al., 2017). In PA, three-quarters of the state, and nearly 70% of PA state forests overlay shale gas reserves (Drohan et al., 2012). Unconventional gas extraction from the Marcellus Shale within the PA state forests began in 2008 and by 2012 resulted in the direct loss of 601 hectares of forest to 232 well pads, the widening or creation of 259 kilometers of roads, and 167 kilometers of widened or new pipeline corridors (DCNR, 2014). While this direct loss is less than 0.0007% of the total PA state forest system, the development has led to extensive forest fragmentation with an increase in 1762 hectares of edge forest and the loss of 3740 hectares of core forest (DCNR, 2014). Landscape-scale forest fragmentation and disturbance are known to facilitate the spread of invasive plants (With, 2004; Minor et al., 2009; Vilà and Ibáñez, 2011). Construction of well pads, transmission pipelines, and access roads create new forest edge, can alter soil chemistry and structure (Fink and Drohan, 2015), and plant community composition to non-native, disturbance-adapted species. For example, in the Williston Basin of the Northern Great Plains, US, Preston (2015) found non-native plant recruitment adjacent to unconventional oil well pads in native prairie. In addition to the vegetation and soil disturbance, UOG development results in a dramatic increase in truck and heavy equipment traffic increasing the likelihood that invasive plant seed and vegetative propagules are introduced. Increased disturbance coupled with increased propagule pressure could accelerate invasive plant spread in these previously core forest habitats (Huebner and Tobin, 2006), particularly if source populations from newly constructed roads and pipelines are not managed.

UOG is commonly produced with horizontal drilling and hydraulic fracturing staged on rectangular-shaped pads. These well pads are typically several hectares in size to accommodate drilling equipment, truck traffic and fracturing fluid and proppant storage units. Well pads are often constructed in clusters and connected by a shared access road that branches from existing roads. When development occurs near secondary or rural roads that cannot accommodate large, heavy vehicles (such as within forests), new roads are built or existing roads are widened. The expanded road network for UOG development could increase the likelihood of invasive plant spread and establishment.

Road development within a forest matrix is known to have substantial long-term impacts on ecosystem function (Forman and Alexander, 1998; Kuhman et al., 2010) and can create pathways for invasive plant establishment and spread (Birdsall et al., 2012; Hansen and Clevenger, 2005; Mortensen et al., 2009; Parendes and Jones, 2000; von der Lippe and Kowarik, 2007; Watkins et al., 2003). Roads facilitate spread when plant establishment is enhanced by increased light, and when soil physical and chemical properties are altered to favor establishment and growth of disturbance-adapted alien species over native (Johnston and Johnston, 2004; Nord et al., 2010). Material and equipment used for road construction can play an important dispersal role (Taylor et al., 2012) at these sites. Given the fact that on average 1235 one-way truck trips delivering fracturing fluid and proppant are required to complete an unconventional well (Sibrizzi and LaPuma 2016), the potential to transport invasive plant propagules is significant. Propagule transport could occur from mud on the tires and undercarriage of vehicles (Taylor et al., 2012), by road construction and maintenance, and by way of vehicle airflow (von der Lippe et al., 2013) and wind (Caplat et al., 2012). For example, Rauschert et al. (2017) attributed much of rapid advance of *Microstegium vimineum* to seed movement by road grading equipment on forested gravel roads. UOG development typically involves drilling one or more wells on a well pad and the addition of new wells can occur over a period of years (Drohan et al., 2012). As high propagule pressure is known to be a significant contributor to successful invasive plant establishment (Simberloff, 2009), we propose that the likelihood of invasive plant propagule introductions increases with accompanying high-density traffic with each additional well.

In this work, we hypothesize invasive plant presence on well pads is correlated with the length of time since pad construction, the number of wells drilled per pad, invasive plant abundance on adjacent well pad access roads, and the density of roads in the area of the pad prior to construction (Menuz and Kettenring, 2013; Watkins et al., 2003). Using field data from 127 well pads, we created an *a priori* piecewise structural equation model (SEM) to evaluate direct and indirect relationships between mechanisms and conditions that could account for invasive plant presence (Figure 2.1). We predicted that time since pad construction, wells per pad, and pre-pad road density are indirect drivers of invasive plant presence on pads mediated by the density of invasive plants on access roads. Furthermore, we predicted that wells per pad are directly correlated with time since pad construction. The surrounding plant communities were additionally surveyed on a randomly selected set of 32 well pads in the study. We hypothesized that similar to Preston (2015) non-native plant cover would be greater on the disturbed well pad edges than in the surrounding plant communities.

Methods

Study region

Invasive plant surveys were conducted between July and September in 2012 and 2013 on 127 Marcellus Shale gas well pads and adjacent access roads. Pads were located in 7 publicly managed PA State Forest Districts (n=116) and in the Allegheny National Forest (n=11) in northcentral PA (Figure 2.2). The study sites are distributed across the unglaciated Allegheny High Plateau physiographic province (Shultz, 1999), which is dominated by mixed-oak and Northern hardwood forests (Fike, 1999), and the Pittsburgh Plateau and the Ridge and Valley regions of central PA, dominated by mixed-oak forests. Loam or sandy loam soils are the most common surface and subsurface soil textures across the study area (Ciolkosz et al., 1989). The soils of the

Pittsburgh Plateau formed from acid clay shales and low fertility sandstones and siltstones while ridges of the Ridge and Valley and Allegheny High Plateau are largely comprised of sandstone and siltstone with some shale (Ciolkosz et al., 1989; Shultz, 1999). Average yearly precipitation in central and northcentral PA ranges between 95-115 cm yr^{-1}.

Invasive plant data collection

At each of the 127 well pad study sites, invasive plant data were collected by walking along a belt transect of the revegetated pad perimeter and conducting a visual scan of the pad surface. We rarely documented plants growing on the pad surface as pads are typically covered by a thick layer of limestone gravel and are not suitable for plant growth. The width of the revegetated pad perimeter varied but was approximately 10 m wide. As this study focused on well pads in forested landscapes, the disturbance from pad construction was visually distinct from the surrounding forest cover and can be seen from aerial imagery (Figure 2.3). Pad access road edges, defined as the disturbed area from gravel road edge to the forest edge, approximately 10 m wide, were also surveyed for invasive plants for a distance of 0.5 km from the pad edge. If the pad was adjacent to a main road those road edges were surveyed 0.25 km in either direction of the well pad. We surveyed for invasive non-native plants of concern in PA forests (Table 2.1). Stem counts for each species were classed in the following four categories 0, 1-10, 11-100, 101-1000, >1000.

Data on pad area, age, wells, and road density

The PA DCNR provided data on pad area, pad age (years since construction), and the number of wells drilled per pad. Pad area ranged from 0.35 to 5.60 ha, with a median 1.71 ha, pad age ranged from 0 (within first year of completed construction) to 6, with a median of 2, and the number of wells on a pad ranged from 0 to 11, with a median of 3. Road density in the area of the pad prior to construction was calculated in ArcGIS v10.1 (ESRI, 2012) from 2005 pre-shale gas extraction aerial imagery (1-foot pixel resolution) sourced from the National Agriculture Imagery Program (USDA FSA APFO, 2006). Total road length within a 200 m radius of the pad center included paved and unpaved roads. Road density ranged from 0 to 1395 m, with a median of 290 m.

Plant community surveys surrounding well pad sites

A subset (32) of well pad sites that spanned the region (6 state forest districts) was randomly selected to survey the surrounding plant communities in comparison to the reclaimed well pad edge. At each well pad 1.5 x 1.5 m quadrats were placed at 0, 25, and 100 m along a linear transect from all four sides of the well pad. Quadrat locations were determined in Google Earth (Google Inc, 2017) prior to conducting the surveys. All plants within the quadrat were identified to species and recorded for percent cover by species.

Statistical analysis

Invasive plant presence, frequency of occurrence, and density

First, we report on the presence, frequency, and density of invasive non-native plants on well pad edges, and well pad access roads. Stem densities on roads and pads were assessed for normality with the Shapiro-Wilk test and non-parametric test statistics were used when appropriate. Kendall's rank correlation tau, well suited for non-parametric data (Croux and Dehon, 2010), was used to reveal species level insights for *M. vimineum*, which is known to spread along gravel roads typical of the unconventional gas development in the PA state forest system (Mortensen et al., 2009; Rauschert et al., 2017). *M. vimineum* abundance along the well pad edges and adjoining access roads could indicate invasive populations are spreading locally. If a species is absent on the access road but present on the adjacent well pad, this could indicate invasive plant propagules are being spread by development related activities, such as gravel brought to the site and on vehicles traveling to and from the site. For the 4 most common species *M. vimineum*, *Phalaris arundinacea*, *Centaurea stoebe,* and *Cirsium arvense* we report on the frequency of populations found on the well pad and not on the access roads.

Structural Equation Modeling - Invasive plant colonization on pads

Piecewise SEM (Lefcheck, 2016) was used to assess the hypothesized *a priori* direct and indirect drivers of invasive plant colonization (presence) on pads (Figure 2.1). SEMs allow for variables to be included as both predictors and outcomes to assess the realistic complexity of their relationships. Variables within and affecting the system are referred to as endogenous, whereas exogenous variables are not driven by the system (Grace et al., 2010). Invasive species presence, not stem density, was used for the endogenous variable on pads, as density is more likely correlated to habitat suitability and abiotic factors on the pad (e.g., soil pH, soil fertility, climate) that were not measured. Because of slower reproductive rates, dispersal, establishment, and spread of woody species only herbaceous species were included in the SEM. For invasive plant abundance on access roads we did not distinguish between species, but used a sum of all invasive herbaceous plant species' abundance. We used the intraclass correlation coefficient (ICC) to assess variability at the forest district level (Nakagawa and Schielzeth, 2010) and therefore to determine if we should use multilevel modeling, i.e. observations within a forest district are more similar than between forest districts. Piecewise SEM uses the test statistic Fisher's C, derived from the *p*-values of all linear models in the SEM (basis set), and model fit is indicated by a *p*-value of 0.05 or greater (Lefcheck, 2016).

Mixed effects logistic regression (MELR) models were used for the two binary endogenous variables in the SEM; the presence or absence of invasive plants on pads, and the presence of more than one well per pad. Potential direct drivers of invasion on pads included: road density in the area of the pad pre-construction (200 m radius from pad center), invasive herbaceous plant density on access roads, presence of more than one well per pad, and pad age. Results are expressed as unstandardized correlation path coefficients, and as odds ratios by taking the exponential of the coefficients. Odds ratios (OR) describe the change in odds of an outcome for every single-unit increase in the predictor. A generalized linear mixed model (GLMM) was used for a third endogenous variable, the density of invasive plants on well pad access roads, which fit a negative binomial distribution. Results are expressed as unstandardized correlation path coefficients, and as incidence rate ratios (IRR) by taking the exponential of the coefficients. The IRR is the ratio of the rates of two outcomes. Pad area was variable and was used as an offset (Hilbe, 2014) in the GLMM for invasive presence. We report the conditional (fixed and random effects) R^2 values for the MELR model on invasive presence on well pads according to Nakagawa and Schielzeth (2013) and Johnson (2014).

The MELR model of invasive plant colonization on well pads used binary data y_i, where,

$y_i =$ 0 if the i^{th} pad had no invasive plant species present

1 if the i^{th} pad had an invasive plant species present

with the probability p that invasion occurred on the i^{th} pad, and j=8 forest regions (7 PA State Forest districts and the ANF), and colonization is dependent on the following variables, with α intercept, and β and δ path coefficients,

$$\text{logit}(p_{ij}) = \alpha + \beta_1 \times \text{ prior road density}_{ij} + \beta_2 \times \text{ invasive density on access roads}_{ij}$$
$$+ \beta_3 \times \text{ wells}_{ij} + \beta_4 \times \text{ pad age}_{ij} + \delta \times \text{ district}_j + \text{offset(pad area)}.$$

The MELR model specifications for the presence of more than one well per pad is binary data z_i, where,

$z_i =$ 0 if the i^{th} pad had 0-1 well

1   if the i^{th} pad had 2 or more wells

and dependent on the following variables, with α intercept, and β and δ path coefficients,

$$\text{logit}(p_{ij}) = \alpha + \beta_1 \times \text{pad age}_{ij} + \delta \times \text{district}_j.$$

The model for the density of invasive plants on access roads had a count data response w_i with a negative binomial distribution, and was dependent on the following variables, with α intercept, and β and δ path coefficients,

$$\log(w_{ij}) = \alpha + \beta_1 \times \text{prior road density}_{ij} + \beta_2 \times \text{number of wells}_{ij} + \beta_3 \times \text{pad age} + \delta \times \text{district}_j.$$

Prior road density was used as a surrogate for invasive plant propagule sources for the invasion of new roads, pipelines, and pads (Watkins et al., 2003). The number of wells drilled was used as a measure of the amount of traffic to that pad (Sibrizzi and LaPuma, 2016). Pad age was used as a measure of time since the original site disturbance.

Plant community change with gas development

Plant species cover from the 3 m^2 quadrats were categorized by nativity status to assess plant community composition change within a forest after the establishment of a well pad. We calculated the sum of native and non-native cover at the quadrat level, averaged cover by nativity status at each distance at the pad level, and created a ratio of non-native to native cover by distance. We used the Kruskal-Wallis test to assess our hypothesis that non-native plant cover would be greater on well pad edges compared to the surrounding plant communities.

All analyses were performed in R version 3.1.2 (R Core Team 2014), using piecewiseSEM (Lefcheck, 2016), lmerTest (Kuznetsova et al., 2014), and nlme (Pinheiro et al., 2015) libraries.

Results

Overall findings

Sixty-one percent of pads had at least one invasive non-native plant species, and 19% of those had 3 or more species. The presence of invasive herbaceous plants far outnumbered invasive woody plants; 61% of pads had herbaceous plants while only 17% of the pads had woody invasives. *Phalaris arundinacea*, *Centaurea stoebe*, *Cirsium arvense*, *M. vimineum*, and *Securigera varia* were the most common invasives and were found on 13-25% of pads, whereas other invasive species included in the survey were each found on less than 4% of the pads. We found evidence of an association between *M. vimineum* stem densities on access roads and adjacent well pads (Kendall's rank correlation tau = 0.31, z = 3.66, p < 0). *Cirsium arvense* was found on the pad and not on access roads at 21 sites, *C. stoebe* at 14, *P. arundinacea* at 13, and *M. vimineum* at 5. For example, at one survey site in the ANF the stem count of *C. stoebe* was in the 1000s of stems on the pad and absent along the 0.5 km access roadside. In such cases, the first point of introduction was likely the pad, which suggests the introduction was related to development activities, such as gravel introduction or vehicle tires.

Invasive plant colonization on unconventional gas pads via access roads

The SEM fit the data (Fisher C 1.77, df=2, *p*=0.41). The proportion of variance at the level of forest districts (ICC) was large (0.75) and therefore required it be accounted for in a mixed model analysis. The resulting MELR model with invasive plant presence on pads as the response had a conditional R^2 (fixed and random effects) of 0.42.

Of the factors we hypothesized as drivers of invasive plant spread to well pads, the density of invasive plants on pad access roads was the only directly correlated variable. Figure 2.4 provides the unstandardized path coefficients which, when taking the exponential, provides the odds ratio

(OR) for the MELR models, and the incident rate ratio (IRR) for the GLMM. Expressed as an OR, for every one-unit increase of invasive plant stems found on access road edges, the odds of an invasive plant colonizing a pad increased by 1.002. The OR is small as the stem number on access roads ranged widely from 0-2650 with a median of 110. Pre-pad development road density and sites with greater than one well per pad were significant predictors of access road invasive stem density and therefore were indirectly associated with invasive plant presence on pads. For each meter of road in the vicinity of the pad prior to gas development, the rate of colonization of invasive plant stems on access roads increased by a factor of 1.51, strong evidence that the extent of the road network is associated with invasive plant success. Well pads with more than one well had a rate of invasive plant establishment (stem density) 1.59 times greater than pads with only one well.

Pad age was not a direct causal factor of invasion on pads, but older pads were more likely to have more than one well and have invasive plants on adjacent access roads, therefore pad age was also indirectly linked to invasion on pads. Nearly 70% of pads surveyed were between 2-3 years old. Given time we would expect pad age and continued human activity to be a positive driver of invasive plant spread within regions of UOG development (Vitousek et al., 1997).

Well pads introduce non-native plants to native forest plant communities

Non-native plants were rarely found in the surrounding forests (quadrats sampled at 25 m and 100 m). In fact, across the 32 well pads we studied, non-natives were present on only 3 well pads at 100 m from the pad edge, each at less than 2% cover, and were never documented at 25 m (Figure 2.5). Yet all surveyed well pads had some nonnative plant cover on the disturbed edges with an average of 43% (s.d. 0.19). The Kruskal-Wallis test identified a difference ($\chi^2 = 85.4$, df = 2, $p < 0.001$) in the ratio of non-native to native cover across survey site distance from the well pad edge (0, 25, 100 m). Non-native presence in the disturbed areas around the pad is due in part to the seed mix composition typically used for reclamation. This is evidenced by the 5 most frequently observed non-native species on the pad edges; *Dactylis glomerata*, *Phleum pratense*, *Trifolium repens*, *Lotus corniculatus*, and *Lolium multiflorum*. But natural recruitment via wind or water, or human management activities, such as seeding materials and equipment, likely introduced species such as *Agrostis gigantea*, *Plantago lanceolata*, *Holcus lanatus,* and *Veronica officinalis*. The 5 most frequent native plants observed on the pad edges were *Dichanthelium clandestinum*, *Acer rubrum*, *Dennstaedtia punctilobula*, *Rubus sp.*, and *Rudbeckia hirta*. *D. clandestinum* and *R. hirta* are often included in reclamation seed mixes. Within the surrounding forests (25 and 100 m) the most frequently observed species were *Acer rubrum*, *Gaultheria procumbens*, *Kalmia latifolia*, *Vaccinium angustifolium*, *Gaylussacia baccata*, *Vaccinium pallidum*, *Dennstaedtia punctilobula*, *Hamamelis virginiana*, and *Amelanchier* sp., plants typical of the dry oak-health and dry oak-mixed understory of most sites in this study.

Discussion

Unconventional gas development spreads invasive plants

Non-native invasive plants are moving further into PA forests with the development of unconventional gas. In fact, non-native plants were virtually non-existent in the forested sites surveyed surrounding well pads, and yet are becoming a dominant part of the plant community around pads. We found that within less than a decade invasive non-native plants have spread to over half of the 127 well pads in our survey, and for the 85% of the pads that were less than 4 years old it occurred in a much shorter period of time. The SEM identified a positive correlation between invasive plants on pad access roads and invasive plant presence on well pads, and demonstrated that invasive plant colonization is more likely as the number of wells per pad increase. Our findings in this forested system are consistent with previous studies. Joly et al. (2011) found that paved regional roads with heavier traffic were a much better predictor of *Ambrosia artemisiifolia* L. distribution than any landscape predictor. In such cases, roads serve as a corridor of suitable habitat and for plant propagule dispersal by way of vehicles driving to pads moving propagules from nearby or distant sources on tires or the vehicle undercarriage (Taylor et al., 2012), by vehicle airflow (von der Lippe et al., 2013), by animals (Cousens et al., 2010), or on human clothes, shoes, and tools. As evidence of the impact of traffic density, we found that the rate of invasive plant establishment on access roads that led to pads with more than one well was 1.5 times that of access roads that led to single well pads. At the time of our study, 28% of well pads within the state forest system had only one well. Throughout PA the number of wells per pad ranged from 1-25, with a median of 2, where 37% had only one well and 54% had less than three wells (PA DEP, 2015). As production continues over the next several decades invasive plant spread will likely become a greater challenge. Conversely, some pad sites may remain with one well for years with little human activity. Our research suggests that limited human activity poses a lower risk of invasive plant establishment and therefore would rank single well pads at lower priority for monitoring programs.

We found that pads constructed in areas with higher road density are more likely to become invaded due to proximity to likely sources of invasive plant propagules in edge habitat within the forest matrix (Birdsall et al., 2012). Most of the road networks connecting unconventional gas infrastructure within the PA state forest system were not new (DCNR, 2014), and therefore our survey could have identified pre-existing invasive populations. The process of widening existing roads for heavy truck traffic could also have brought in invasive propagules with trucks and construction material. Widened roads and heavy traffic are common forms of disturbance with unconventional gas development in the region and are likely playing an important role in invasive plant establishment and spread (Hansen and Clevenger, 2005; Moles et al., 2012). Invasive plant populations found along existing roads could have been pre-existing populations or have established during development. From a practical perspective, the fact that invasive plant presence was associated with the pre-existing road network could be used to guide the design, frequency, and citing of invasive plant monitoring protocols. The pre-existing road-invasive plant association also raises questions about suppression strategies. For example, invasive plants in areas with high road density could be targeted for suppression prior to pad and pipeline construction.

We were surprised at the relatively small number of invasive plant species we saw on most pads and adjacent access roads. While not a part of the survey design, microsite conditions obviously varied considerably along the perimeter of the pads and along roadsides. Establishment and spread of these adapted species will be strongly influenced by context specific variation in site conditions (Cadenasso and Pickett, 2001; Minor et al., 2009; With, 2004). For example, while *P. arundinacea* and *P. australis* typically invade roadside verges and nearby wetlands (Houlahan et al., 2006; Jodion et al., 2008), *C. stoebe* and *C. arvense* invade grasslands (DiTomaso, 2000). Land managers will need to make decisions based on local site characteristics to prioritize species of concern. *Microstegium vimineum* has rapidly become a dominant species on forest roads in central PA forests (Mortensen et al., 2009). Species that are particularly problematic from a management point of view and are particularly well adapted to edges and forest interiors such as *M. vimineum*, *Elaeagnus umbellata*, and *Berberis thunbergii*, and wetland-adapted species such as *P. cuspidatum*, should be a high priority for management at pad sites and along access roads. Although woody species were infrequent in our surveys, we suspect that given time fruit-bearing shrubs favored by birds will be present on forest edges abutting pads, pipeline and access road corridors.

Conclusions

Identifying drivers of invasive plant spread within the UOG development footprint will be key to minimizing and managing further colonization. Our work indicates that this development predisposes forested landscapes to plant invasion and raises important questions about how, in the face of continued UOG production, we mitigate the success of invasive plants. Introduction, establishment, and spread of invasive plants at these disturbed sites have the potential to reduce native plant abundance and diversity in surrounding forest communities and alter ecosystem functioning (Vilà et al., 2011), as well as challenge revegetation goals (D'Antonio and Meyerson, 2002). Developing invasion resistant seed mixes for reclamation and long-term site management should be research priorities to assist in stemming establishment and further spread.

Going forward, a deeper understanding of the link between expanding road networks in forested landscapes and their role in invasive plant success is needed. Such insights should be incorporated in invasive plant management strategies so that they are designed to be context specific and informed with temporal insight. For example, expanded road networks and gas pipelines create linear corridors passing through a wide range of forested habitats some of which are highly susceptible to invasion. Further investigation into habitat susceptibility is needed to bolster resilience in forests with increasing edge habitat where newly established invasive plant populations could serve as sources of invasion in the broader landscape. Our current analysis suggests effective monitoring and rapid response weed management should be guided by preexisting knowledge of the site. Such "smart" monitoring and control programs will increase management effectiveness, reduce time and labor associated with sampling areas that are less likely to be invaded, and reduce the possibility that plant invasions are overlooked.

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Tables & Figures

Figure 2.1. Hypothesized direct and indirect causal pathways of invasive plant presence on well pads.

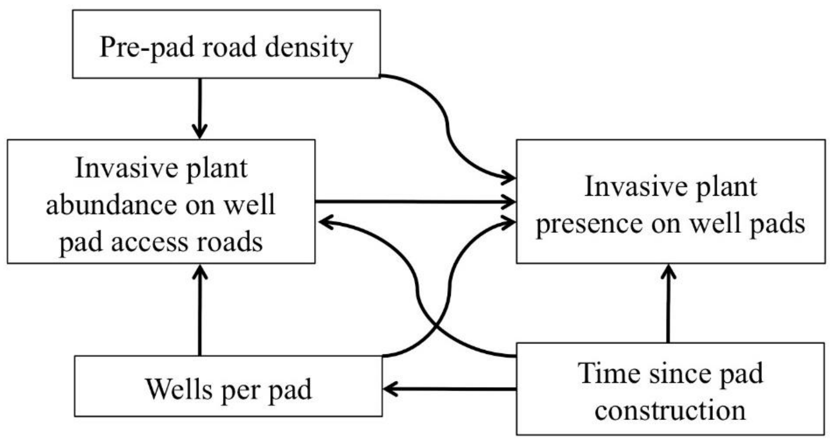


Figure 2.2. Map of the survey locations and physiographic sections. Grey spheres mark well pad sites in north central Pennsylvania, with inset map of the northeastern U.S. showing the states of Pennsylvania (PA), New York (NY), New Jersey (NJ), Maryland (MD), and West Virginia (WV).

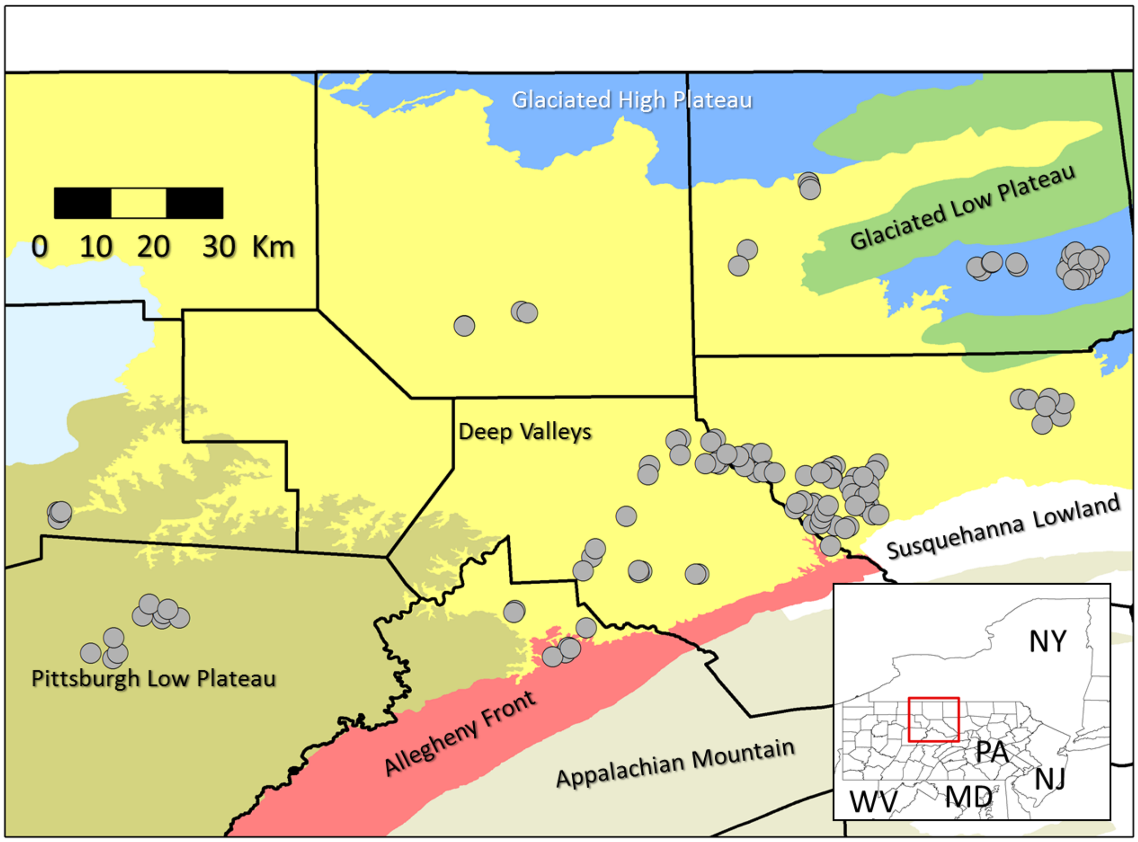
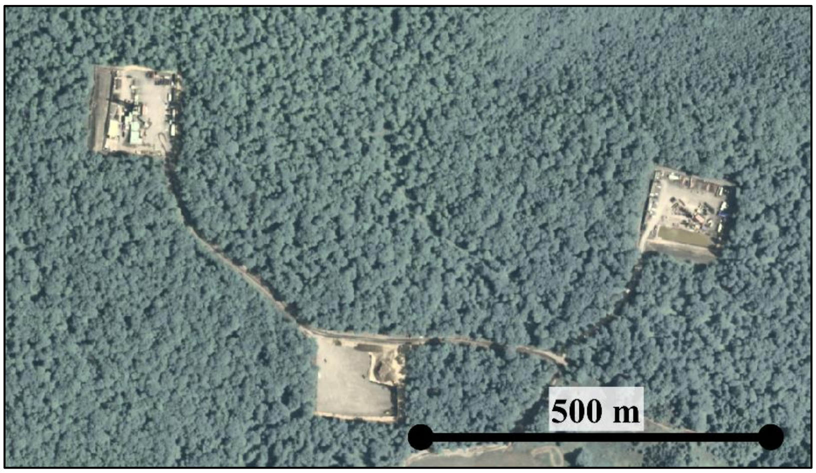


Figure 2.3. (a) Aerial photo of three unconventional gas well pads in a northcentral PA forest. A central access road connects the pads to the main road (not seen). Aerial imagery sourced from the National Agriculture Imagery Program (USDA-FSA APFO 2010). (b) Reclaimed edge of a gas well pad surveyed for invasive non-native plants, pad is out of view to the right. Photo credit: Kathryn M. Barlow



3a.

3b.

Table 2.1. Herbaceous and woody invasive non-native plants included in the survey on unconventional gas pads in PA forests. Nomenclature according to the United States Department of Agriculture plants database (USDA NRCS, 2017).

Scientific name	Common name	Plant type
Alliaria petiolata (M. Bieb.)	garlic mustard	Herbaceous
Celastrus orbiculatus Thunb.	oriental bittersweet	Herbaceous
Centaurea jacea L.	brownray knapweed	Herbaceous
Centaurea stoebe L.	spotted knapweed	Herbaceous
Cirsium arvense (L.) Scop.	Canada thistle	Herbaceous
Microstegium vimineum (Trin.) A. Camus	Japanese stiltgrass	Herbaceous
Phalaris arundinacea L.	reed canarygrass	Herbaceous
Phragmites australis (Cav.) Trin. ex Steud. subsp. *australis*	European common reed	Herbaceous
Polygonum cuspidatum Siebold & Zucc.	Japanese knotweed	Herbaceous
Polygonum perfoliatum L.	Asiatic tearthumb	Herbaceous
Securigera varia (L.) Lassen	purple crownvetch	Herbaceous
Acer platanoides L.	Norway maple	Woody
Ailanthus altissima (Mill.) Swingle	tree of heaven	Woody

Berberis thunbergii DC.	Japanese barberry	Woody
Elaeagnus umbellata Thunb.	autumn-olive	Woody
Frangula alnus Mill.	glossy buckthorn	Woody
Ligustrum vulgare L.	European privet	Woody
Lonicera maackii (Rupr.) Herder	Amur honeysuckle	Woody
Lonicera morrowii A. Gray	Morrow's honeysuckle	Woody
Rhamnus cathartica L.*Rosa multiflora*	common buckthorn	Woody
Thunb.	multiflora rose	Woody

Figure 2.4. Unstandardized path coefficients and the standard error (SE) of direct and indirect causal paths to invasive plant presence on well pads. Causal paths with solid lines are significant at the α=0.05 level and non-significant paths are indicated with dashed lines.

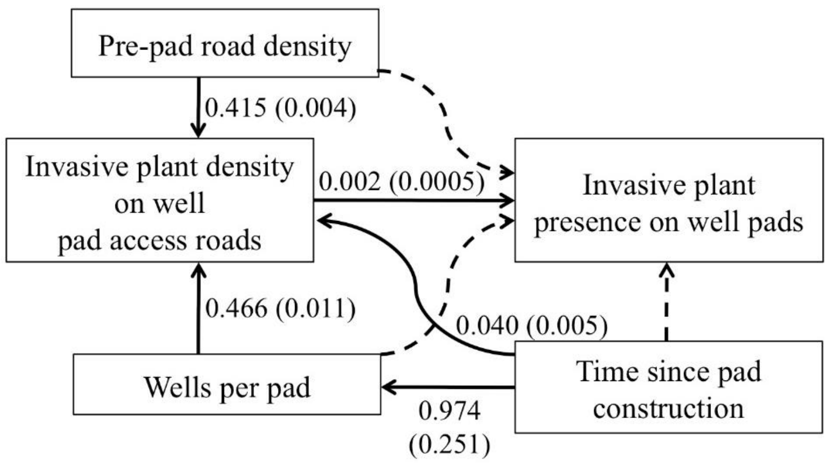
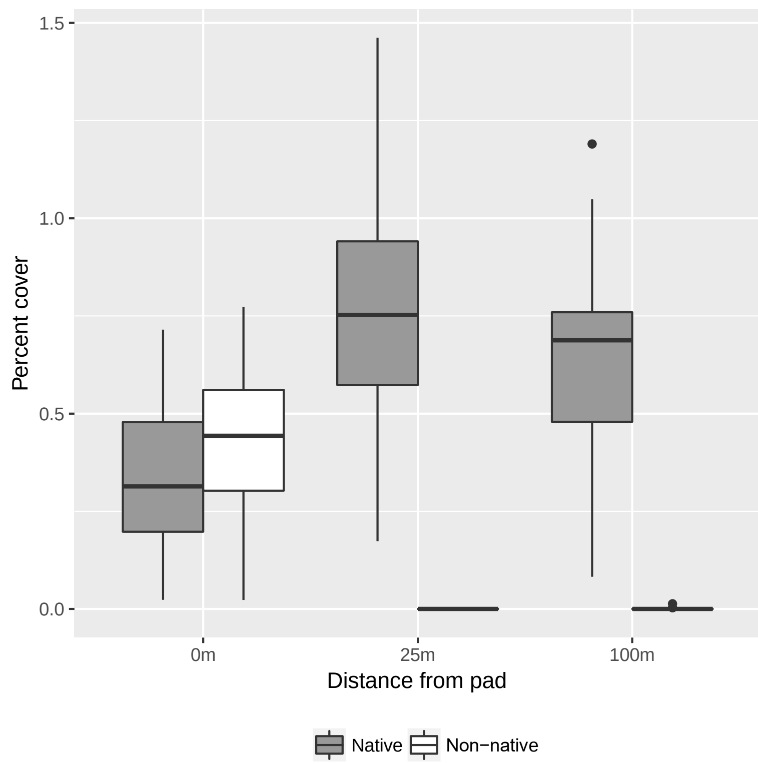


Figure 2.5. Percent cover of native and non-native plants on the well pad edge (0 m) compared with the surrounding forest at 25 m and 100 m from the edge.



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Chapter 3

Perennial native grass establishment on compacted soils

Abstract

Reclamation of unconventional oil and gas well pads is challenged by severe soil compaction. High soil bulk density limits plant root growth and could impair the successful establishment of desired plant communities. Poor establishment of the desired plant community could lend greater resources for weedy and invasive plant growth and reproduction, creating an undesirable weed seed bank on-site and a source for further spread in the landscape. Seed mixes of cool-season grasses and legumes are commonly used to attain rapid cover for site stabilization and weed exclusion, yet have limited wildlife habitat value. We assessed the initial establishment success of four commonly recommended native perennial grasses on a simulated well pad to inform the suitability of these species as an alternative for reclamation mixes with habitat benefits. The native prairie grasses, *Dichanthelium clandestinum* (L.) Gould, *Sorghastrum nutans* (L.) Nash, *Panicum virgatum* L., and *Andropogon gerardii* Vitman, were seeded in monocultures and 2-, 3-, and 4-species combinations on decompacted and compacted soils. We seeded subplots of the invasive annual *Microsteigum vimineum*, as a phytometer for invasion resistance. Over three growing seasons we collected root and shoot biomass of the native grasses along with shoot biomass of the weedy flora, including *M. vimineum*. Over the three years compaction did not result in a biomass reduction of the native grasses either above or below ground, and neither did weedy flora differ by soil treatment. The native mix species produced very little biomass in the first year, but by the third season significantly outcompeted the weedy flora, including *M. vimineum*. Given the tolerance of these native grasses to soil compaction we recommend incorporating these species in mixtures for highly disturbed soils. Further research widening the range of soil compaction treatments would aid in informing broader management prescriptions.

Introduction

Establishing plant communities and the ecosystem functions they perform is challenging in highly degraded soils (Allred et al., 2015; Angel et al., 2009; Haigh & Sansom, 1999). Soils degraded by activities associated with oil and gas extraction are typically characterized by loss of topsoil, structure, and organic matter, as well as increased bulk density resulting from compaction by heavy equipment (Cambiet al., 2015; Nawaz et al., 2013; Plass, 2000). Even with remediation efforts it can take decades to centuries to recover such severe soil degradation, with long-lasting effects on plant community structure and associated ecosystem functions. Surface and subsurface natural resource extraction often results in a legacy of degraded soils and disturbance-adapted plant communities (Nasen et al., 2011; Oliphant et al., 2017). The recent development of unconventional oil and gas will likely leave a similar legacy unless affected ecosystem functions are not addressed in restoration (Souther et al., 2014).

Unconventional oil and gas production refers to subsurface liquid and gaseous hydrocarbon reservoirs of low permeability that require advanced technologies for extraction, such as hydraulic fracturing and horizontal drilling. Well pads used to stage production are constructed by removing vegetation, stockpiling topsoil, compacting subsoil to “no movement”, and typically adding a layer of gravel. The process of restoring the site involves significant traffic from heavy equipment often resulting in highly compacted soils. Such compaction is common with natural resource extraction, as has been observed on logging landings and skid trails (Reisinger et al., 1988; Williamson and Neilsen, 2000) and well sites for conventional gas production (Fink and Drohan, 2015; McConkey et al., 2012; Nasen et al., 2011).

In a study of 5-year-old reclaimed unconventional well pads in Pennsylvania, Fink and Drohan (2015) found that bulk density values were still greater than adjacent undisturbed soils, and subsurface soils were 1.63 g cm^{-3}, a density that can adversely affect root growth. Fink and Drohan (2015) also found evidence of subsoil (20 cm) compaction on 75-year-old conventional well sites that have experienced less heavy equipment than is required for unconventional well development. Similarly, Bohrer et al. (2017)

found that the penetration resistance of reclaimed mine soils remained high 40 years after mining ceased. Contrary to what was expected due to freeze-thaw and wet-dry cycles, the soil penetration resistance remained uniformly high even in shallow soil profile layers (Bohrer et al., 2017).

Alleviating soil compaction is often necessary to establish the desired plant community during ecological restoration of disturbed sites (Randrup 1997; Bauman et al. 2015; Meyer et al. 2013; Thorne et al. 2013). Severe soil compaction poses a challenge to revegetation efforts, as plant root development is limited by high soil bulk density (Beckett et al. 2017; Daddow and Warrington 1983). Compacted soils can alter the radial expansion and elongation rates of roots (Tracy et al. 2011), and root architecture (Grzesiak et al. 2013), which in turn alters, usually reducing, biomass allocation (Bohrer et al. 2017). Masle and Passioura (1987) demonstrated that compacted soils can reduce root penetration which in turn reduces root and shoot growth and that such changes in growth are species specific. Soil treatments, such as deep-ripping, which has seen much success on mined lands (Bauman et al., 2015; Skousen et al., 2009), and seed mixes of species with deep, extensive root systems (Swab et al., 2017; Thorne et al., 2013), could be used to accelerate decompaction.

Maintenance of oil and gas well pads, pipelines and associated utility corridors requires that they remain largely free of woody species and are typically reclaimed with grasses and forbs. These interim grasslands must meet state imposed regulations that require establishment of a low maintenance herbaceous cover to minimize soil erosion and limit plant invasions (for example, PA DCNR 2013), yet soil compaction makes this a difficult goal to meet. Perennial cool season grasses tolerate infertile and compacted soils, form rapid ground cover and require little maintenance (Holl 2002). Grasses such as *Poa pratensis* L., *Bromus inermis* Leyss., *Festuca arundinacea* Schreb., *Dactylis glomerata* L., *Agrostis gigantean* Roth, *Lolium perenne* L., *Phleum pretense* L., and legumes such as *Medicago sativa* L., *Trifolium pretense* L., *T. repens* L., *T. hybridum* L., *Lotus corniculatus* L., *Vicia villosa* Roth have a long history of use on highly disturbed soils such as mining restoration sites (Skousen and Zipper 2010). *Poa pratensis* L. and *F. arundinacea* Schreb., for example, have rooting traits capable of tolerating compacted mine land soils and heavy vehicle traffic (Carrow, 1980; Crews, 1984). European *Trifolium* species are commonly included in mixes for turf and reclamation due to compaction tolerance. Given the broad success of these species for soil stabilization in former mine lands they are now commonly incorporated into the reclamation plans for the new unconventional oil and gas production related disturbances.

As has been the case on coal mining sites (Haigh and Sansom, 1999) limiting erosion on unconventional well pad construction sites is a significant concern with shale gas development (Wachal et al., 2008). However, meeting short term erosion reduction goals with these non-native species can forfeit the longer term benefits of restoration plantings that increase native plant diversity and facilitate directed forest succession (Bohrer et al., 2017; Franklin et al., 2012; García-Palacios et al., 2010). The legacy effects of these non-native mixes have been seen in the plant community composition decades after restoration, as loss of rare understory forest species and lower tree density (Holl, 2002; Skousen et al. 2009). Establishing a restoration mix that can provide soil stabilization, while also providing native habitat, can be more challenging (Zipper et al. 2011), but is a worthy goal given their capacity to increase ecosystem integrity. Native perennial grasses have been shown to facilitate tree seedling establishment (Franklin et al., 2012). Native perennial grasses also reduce plant invasions due to high levels of recalcitrant biomass (Mahaney et al., 2015), sequester carbon with extensive belowground biomass (Liebig et al., 2005), and often tolerate droughty soils, low pH, and limited P availability (Morris et al., 1982). Incorporating native warm season grasses on reclamation sites can support a broader array of invertebrate and vertebrate diversity with direct benefits to higher trophic levels than non-native cool season grasses alone (Fetcher et al., 2015).

Seed mixes with native perennial grasses, such as *Dichanthelium clandestinum* (L.) Gould, *Sorghastrum nutans* (L.) Nash, *Panicum virgatum* L., and *Andropogon gerardii* Vitman, are increasingly recommended to restore highly disturbed sites (Miller, 2013) and have been shown to be successful in restoring ecosystem functions on former mine lands and disturbed road edges (Dickerson et al., 1988; Skousen and Venable, 2008; Swab, et al., 2017). Research to inform restoring prairie soils, such as the species composition of a restoration mixture and their performance and persistence, has been well-studied

(Grman et al., 2015; Kindscher and Tieszen, 1998), but few studies exist on the establishment and competition dynamics over time on disturbed, compacted soils of Eastern forests (Swab et al., 2017).

Incorporating an understanding of belowground root growth dynamics and plant-soil feedbacks is crucial for successful restoration (Kardol and Wardle, 2010). Root competition plays an important role in the assembly and persistence of native plant communities. Wilson and Tilman (1991) found that root competition had a greater impact on plant biomass than canopy interactions of neighboring species. Root competition will likely play a larger role in shallow, compacted soils with less available suitable soil for root growth (Bohrer et al., 2017). Potentially, the species with the faster growing root system during establishment will have a competitive advantage and set the stage for community assembly.

In prairie soils tall grass species are known to root deeply; *A. gerardii* and *S. nutans* to 1.5 m minimum, and *P. virgatum* to 2.5-3 m (Weaver, 1954). All three allocate most of their root biomass vertically in the soil profile, but with 70-80% in the top 0.30 m (Nippert et al., 2012). Root branching is highly variable dependent on soil conditions (Weaver, 1954). Both *A. gerardi* and *S. nutans* can branch profusely, but less so in compacted soils (Weaver, 1954). In prairie soils *S. nutans* will not tiller in dense competition (Weaver, 1954). In pot culture monoculture stands of *A. gerardii* was found to have reduced shoot and root biomass in compacted soils (Thorne et al., 2013), and reduced height and root length (McNearney et al., 2002). *P. virgatum* does not branch out in surface soil or moist soil. Research on root traits of prairie grasses and the relationships to ecosystem processes is still in the early stages (Craine et al. 2003a; Craine et al. 2003b; Bonin et al. 2013), including, compaction tolerance, or how plant community assembly of restoration mixes could be affected by compaction.

In addition to the challenges of establishing a restoration mix on compacted soils, the resulting community diversity can also respond strongly to soil condition. Schladweiler and Vance (2005) found that increased topsoil depth on a reclaimed coal mine increased both plant cover and biomass but resulted in reduced diversity (Shannon–Wiener H’) in the initial first three years after planting. Bauman et al. (2015)

found that deliberate attempts to reduce soil decompaction by deep-ripping to 1 m increased native plant cover on reclaimed mining sites.

Inter- and intra-specific competition below and above ground resulting from mix design will also affect plant community diversity. For example, inter-specific competition altered the shoot biomass of two prairie native warm season grasses (Van Auken et al., 1994). Mix species richness, total seeding density and species proportions, and the presence of a dominant competitive will all influence competition dynamics. Greater seeding densities often result in density-dependent mortality. For example, Burton et al. (2006) found that at the end of two years, plant cover of a native grass and forb mix was not different when seeded at densities between 1,500 and 6,000 plants m^{-2}. Wilkerson et al. (2014) also found 'diminishing returns' for higher seeding rates when mixes consisted of forbs, as did Oliveira et al. (2014) for mixes of grasses and forbs. In terms of weed suppression, Nemec et al. (2013) found that greater seeding density of mix species was less important than species richness.

Often in restoration mixes, one or two species will dominate the established plant community, making it difficult to achieve the desired plant diversity at the restored site (García-Palacios et al., 2010; Sasaki and Lauenroth, 2011; Oliveira et al., 2014). *Panicum virgatum* in restored native prairie communities is a case in point (Springer et al., 2001; Baer et al., 2005; Hong et al., 2012; McCain et al., 2010). Dominance has likely evolved in *P. virgatum* because commercially available cultivars have been selected for competitive traits (Lambert et al., 2011, Mutegi et al., 2013). We aim to assess whether plant community richness and evenness can be maintained with a low seeding rate of *P. virgatum* in a restoration mix across compacted and de-compacted soils.

Finally, the restoration mix will ideally establish and persist even when challenged by high invasive plant infestations. We chose to assess the invasion resistance of native restoration mixes to Japanese stiltgrass, *Microstegium vimineum* (Trin.) A. Camus, as this invasive annual grass is widespread in Eastern forests and old fields; from the Southern Blue Ridge (Anderson et al., 2012), to New York and southern New England (Hunt & Zaremba, 1992). Large infestations dominated by this species are known to limit native plant diversity and related ecosystem functions (Adams and Engelhardt, 2009; Baiser et al., 2008; Brewer, 2011; Warren et al., 2010). Increasing the species diversity and incorporating competitive species in a restoration mix (Garnier and Navas, 2012; Simmons, 2005) could enhance tolerance of the mix to invasive plant competition. Corridors through natural landscapes, such as roads and utility right-of-ways, and the maintenance for such infrastructure facilitate invasions (Manier et al., 2014; Mortensen et al., 2009; von der Lippe and Kowarik, 2007). Therefore, when disturbed soils, such as with well pads and compressor stations, are connected by corridors, particular consideration must be given to the capacity of a seed mix to establish in the face of invasive plant pressure and once established, limit invasive plant success.

In this study we assess how shallow, compacted soil, at bulk densities typical of unconventional gas well pads, effect the early establishment success of perennial native grasses by measuring root biomass across depth, mix diversity, and weed suppression over three years. To evaluate plant community diversity over time we focused on the influences of mix species richness and the presence of a dominant competitive. The work reported herein quantifies the impact of compacted soils on community assembly of a native plant restoration mix. We focused on four perennial native grasses, *D. clandestinum*, *S. nutans*, *P. virgatum*, and *A. gerardii*, commonly recommended for disturbed sites in eastern deciduous forests typical of the Northeastern US. We hypothesized that soil compaction would reduce root biomass of the native perennial grasses, thereby reducing restoration mix cover and shoot biomass. Further, that soil compaction would reduce mix evenness due to low tolerance of some species in the restoration mix, and result in greater susceptibility to weedy flora in the restored plant communities. Lastly, we assessed the restoration mixes for susceptibility to invasion from *M. vimineum*. We hypothesized that increasing restoration mix richness with functionally competitive species would reduce *M. vimineum* shoot biomass.

Methods

Site and study design

In May 2014, we initiated the study on a 67 x 24 m plot at the Russell E. Larson Agricultural Research Center near Rock Springs, PA (40°43′N, 77°55′ W, 350 m elevation). The surface soil texture on the study site was a silt loam (Buchanan soil series, Fragiudult). Perennial cool season grasses and legumes had been maintained on the study site as it has a fragipan subsurface horizon which tends to restrict root growth in somewhat poorly to moderately well drained conditions, conditions typically be considered marginally suitable for arable agriculture. The study was arranged as a randomized complete block design. Each restoration mix seed treatment was replicated four times and distributed randomly across the soil treatments in 3 m x 3 m plots.

Prior to laying out the treatment blocks the entire research site was prepared to simulate the soil conditions typical of a natural gas extraction site. The top 0.10-0.15 m of vegetation and soil was removed and stockpiled off site with a 550 Crawler bulldozer (7636 kg) for the duration of the study. An additional 0.10-0.15 m of soil was likewise removed to be replaced as topsoil after the subsoil treatments. The two subsoil treatments, compacted and decompacted, were created on the exposed subsoil in strips lengthwise across the site. For the compacted treatment areas we used a walk-behind compactor plate (Wacker Neuson Soil Compactor Plate – WP1550), and for the decompacted treatment areas we used a subsoiling shank pulled through the plot center to a depth of 0.40 m. The subsoil bulk density was measured at the plot level, at the surface (backscatter mode) and across depth (0-0.20 m), with a nuclear density gauge (Randrup and Lichter 2001). Bulk density above 1.40 g cm^{-3} may affect root growth, and greater than 1.65 g cm^{-3} can restrict root growth. Across depth the decompacted plots were less than 1.44 g cm^{-3}, and on average 1.21 g cm^{-3}, within the ideal range for root growth in silt loam soil (Soil Quality Institute Staff 1998). Compacted plots were on average 1.53 g cm^{-3} at the surface. The five treatment plots that were below 1.40 g cm^{-3} (average of both surface and across depth) were not included in the analysis. The stockpiled topsoil was evenly replaced at ~0.05 - 0.10 m across the entire plot with the bulldozer. Soil samples, four samples per treatment replicate, were taken with a corer (0.20 m depth by 0.02 m internal diameter) for soil texture analysis.

Research on agricultural lands has shown that freeze-thaw cycles can alleviate soil compaction (Jabro et al., 2014), but on reclaimed mine lands this was not the case, even after 40 years (Bohrer et al., 2017). To determine if winter freeze-thaw cycles could have alleviated compaction we measured penetrometer resistance, a measure of the kinetic energy needed to penetrate a known depth (Herrick and Jones, 2002), at the end of the second season in September 2015. We tested soil resistance across soil treatments at 0.05, 0.10 and 0.15 m soil depth expecting to document subsoil compaction at 0.10 and 0.15 m. At this point in the trial a nuclear density gauge, a measure of weight per volume, could not be used to compare current subsoil compaction to levels prior to topsoil replacement given measurements are an average across vertical space.

Seed mix design and planting methods

To quantify intra- and inter-specific competition of the four native perennial grasses during establishment we used a replacement series design (Jolliffe 2000). Three species were warm season (C_4 photosynthetic pathway) grasses, *Sorghastrum nutans* (Indiangrass PA ecotype), *Panicum virgatum* (Switchgrass 'Shelter' WV ecotype), and *Andropogon gerardii* (Big bluestem 'Niagara' NY ecotype), and one, *Dichanthelium clandestinum* (Deertongue 'Tioga'), is a cool season grass (C_3 photosynthetic pathway). A replacement series design made it possible to study the performance of species in monoculture as well as in 2, 3, and 4 species combinations, thereby making it possible to compare performance in monoculture to performance in mixture.

Mix establishment is often not seed limited above a critical minimum seeding density (Wilkerson et al. 2014; Oliveira et al. 2014). Therefore, we held total seeding density constant in this trial, with seeding densities comparable to existing recommendations for these species. Seed density for each monoculture and mix treatment plot was held constant at 400 seeds m^{-2}, within the industry standard for mixes containing these species. For each mix, all species were present in equal proportions, except for *P. virgatum*, which was seeded at 17%, or 68 seeds m^{-2}. The recommended proportion of *P. virgatum* is 10-20% of a mix due to its competitive growth over time. An additional two treatments in which *P. virgatum* was seeded at 33%

and 50% were also included to determine the proportion at which *P. virgatum* becomes the dominant species. *Panicum virgatum* dominance was defined as >25% of the seeded proportion at the end of the 3rd season as measured by biomass and percent cover.

Native warm season grasses establish slowly, initially putting more resources into root growth (Skousen & Venable, 2008). To achieve sufficient cover for erosion control in the first year we included oats, *Avena sativa* L., with the perennial mixtures as a nurse crop at a seeding density of 300 seeds m^{-2}. Seed treatment plots were sown by hand in June 2014. After spreading the seed on the plot surface the seed was raked into the soil, mulched with straw at 50% cover, and rolled with a 45 kg drum roller to enhance seed to soil contact. In October of 2014 and 2015 *Microstegium vimineum* subplots of 40 seeds were planted in three rows of 0.25 m, each 0.10 m apart, in each seed mix treatment plot. Seed for the *M. vimineum* subplots was harvested from local populations during September of 2014 and 2015.

Emerging *Cirsium arvense* (L.) Scop. was spot sprayed with Rodeo™ (active ingredient glyphosate) at 4.40 L ha^{-1} one month after seeding. On 5 August 2015 we used 2.24 kg ha^{-1} Facet L™ (active ingredient quinclorac) with soy oil based adjuvant Destiny HC™ at 1.78 L ha^{-1} to set back competing cool season grasses (Miesel et al., 2012). We did not collect pre- and post- plant cover data to quantify the results of the herbicide treatment. Yet, we can say anecdotally that the treatments were effective in setting back or eliminating the target weedy flora.

Soil cores for root biomass across soil depth

Root biomass of species grown in monoculture was measured at the end of the second and third growing seasons. In early September of 2015 and 2016, three 0.30 m depth x 0.05 m internal diameter soil cores were collected from each monoculture plot. The soil core was split into 3 sections, 0-0.10 m, 0.11-0.20 m, and 0.21-0.30 m, and refrigerated at 6 C until processed. To loosen the soil clods each sample was soaked in 5% sodium hexametaphosphate for up to 12 hours prior removing the roots from the soil sample on a 2 mm sieve with gently running water. Root samples were dried overnight at 60 ºC and weighed for total biomass (g).

Plant data collection

To quantify the potential background plant community from the seed bank, and assess its spatial variation, we took five random soil samples per plot with a soil corer 0.20 m deep by 0.02 m wide two weeks prior to sowing the restoration seed mixes. The five soil samples were combined in a sampling bucket, placed in a 1 L sealed plastic bag and stored at 6 ºC until processed. Processing entailed sieving (2 mm sieve) the soil then splitting the sample between two greenhouse flats (0.52 m x 0.25 m x 0.05 m). The field soil was underlain by 0.02 m of vermiculite. The flats were watered twice daily. Emerging seedlings were identified to the species level and counted for density. When the first cohort of seed germination was complete the soil was dried, remixed by hand, and watered as before for an additional census. This process was repeated a total of three times. For each treatment block we report on the total weed counts, the proportion of monocots versus dicots, and the most abundant species.

At the end of the first season during September 2014, we recorded stem counts of the seeded restoration mix species and percent cover of all vegetation, including the background flora using randomly placed 1 m^{-2} quadrats. We continued to track species emergence of the seeded mix starting in late April, and percent cover of all vegetation in late July and mid-September, in both 2015 and 2016. Because a proportion of the mix species are known to be dormant at time of seeding, a final seeded restoration mix stem count was made along with percent cover in late May of 2015. The aboveground biomass, the seeded crop and the background flora were harvested at the whole plot level late September of 2014, 2015 and 2016. The *M. vimineum* subplots were harvested for biomass at seed set in the Fall of 2015 and 2016. All above-ground biomass was dried at 60 ºC prior to weighing.

Statistical analyses

All data were tested for normality and non-parametric tests were used where appropriate. All non-parametric ANOVA tests were completed within the R package WRS2 for Wilcox Robust Estimation and

Testing (Mair et al., 2017; Wilcox, 2012). To assess a difference in soil resistance to penetration across the two soil treatments we used a non-parametric ANOVA 1-way trimmed means test ('t1way'). Root biomass across species, depth, and soil treatment were assessed with a non-parametric ANOVA, 2-way median tests ('pbad2way', with post hoc 'mcp2a'), and 3-way trimmed means ('t3way'). For the plant community datasets (weed seed bank, shoot biomass of restoration mix and *M. vimineum*, and mix diversity outcome variables) we used a non-parametric ANOVA to test differences soil treatments with 1-way ('t1way') and 2-way ('t2way') trimmed means tests and followed with post hoc tests ('lincon') as appropriate. The inverse Simpson diversity metric was used to test for dominance in plant communities to assess the differences in seed mix diversity over time with varying *P. virgatum* seeding densities (equivalent to, lower, and higher than the other two tallgrass species).

Results

Soil compaction

The soil compaction treatment was not affected by freeze-thaw cycles in the second season. Soil resistance was similar at 0.05 m depth ($p = 0.594$), but greater within the compacted treatment at 0.10 and 0.15 m ($p < 0$).

Root biomass growth during establishment

Across species and soil treatments, in 2015 and continuing into 2016, root biomass was greater in the first 0-0.10 m ($p < 0$) compared to the 0.10-0.30 m depth (Figure 3.1). Root biomass more than doubled in the top 0-0.10 m from 2015 to 2016, from an average of 44.7 g m^{-3} to 155.3 g m^{-3}, and under 0.10 m root biomass remained low but quadrupled from 0.83 g m^{-3} to 3.7 g m^{-3} across species and soil treatment. Averaging across species and years we did not find evidence of greater root biomass in the decompacted soil treatment, neither in the 0-0.10 m or 0.10-0.30 m depth ($p > 0.05$) (Figure 3.1).

Root biomass data had been collected for each of the four perennial grasses in monoculture. There were significant differences in root biomass by species ($p < 0.05$) in both 2015 (Figure 3.2a) and 2016

(Figure 3.2b). In the second growing season (2015) *Dichanthelium clandestinum* root biomass in the top 0-0.10 m was 9.7 g m^{-3}, less than 20% of *Andropogon gerardii* which had the third lowest root biomass in compacted soil. By the third season (2016) *D. clandestinum* root biomass was equivalent to *Sorghastrum nutans* which had not increased much in root biomass from 2015. By 2016 *A. gerardii* had surpassed all other three species in decompacted soil. Despite *Panicum virgatum* having significantly lower shoot biomass than both *S. nutans* and *A. gerardii* in the first season, it had equivalent or greater root biomass in the second and third season. In summary, in the second year, the rank order of root biomass across the restoration mix was *S. nutans*, *P. virgatum*, *A. gerardii*, and *D. clandestinum*. By the third year the rank order shifted to *A. gerardii*, *P. virgatum*, *S. nutans*, and finally, *D. clandestinum*. *Panicum virgatum* had the largest difference in root biomass in the top 0.10 m between compacted and decompacted soils.

Plant community composition and changes during establishment

The weed seed bank growouts revealed some variability in weed propagule density across the field site ($p < 0$). Blocks 1 and 2 were weedier with total counts of 12,167 and 10,983 seedlings respectively, compared to 7,805 and 8,610 seedlings for blocks 3 and 4. Dicot weedy plant densities were greater than monocots ($p < 0$), but there was no variability in the proportion of dicot to monocot weeds across the field site ($p = 0.94$) (Figure 3.3). The top six most abundant species in each block were the same; *Carex* sp. L., *Lobelia inflata* L., *Oxalis stricta* L., *Veronica serpyllifolia* L., *Veronica peregrine* L., and *Cardamine hirsuta* L.

The expected ‘sleep, creep, leap’ pattern of perennial grass growth was evident in our study (Figure 3.4). At the end of the first growing season, the combined biomass of the four species mix was less than 1 g while the oat cover crop biomass was 82.6 g. The oats did not reseed and were not present in the second and third seasons. The average biomass of the four species mix was 30.0 g in the second season and more than quintupled to 173.4 g by the third season. The biomass of the 4 species mixture far surpassed weed biomass by the third season in both the compacted and decompacted soils. The plant biomass patterns over the three seasons are reflected similarly by the mix ground cover (%) (Figure 3.5). In the first season, across soil treatments, bare ground averaged 25%. Bare ground was marginal in the second and third seasons, averaging 6% and 4% respectively across soil treatments. Plots with compacted subsoils had greater bare ground compared to decompacted soils ($p = 0.03$) (Figure 3.6). Yet, 4% is minimal bare ground and not a concern for soil erosion. We did not collect data on soil erosion, but given slope and rainfall patterns, plant ground cover during establishment of this restoration mix was likely sufficient to meet this ecosystem function requirement (Zuazo and Pleguezuelo, 2008).

Soil compaction did not reduce mix shoot biomass (g) ($p = 0.49$) or mix vegetative ground cover ($p = 0.53$) or the background flora ($p = 0.74$) across all mix combinations by the third season. *Sorghastrum nutans* had by far the greatest shoot biomass when grown in monoculture, followed by *A. gerardii* (Figure 3.7). *Dichanthelium clandestinum* and *P. virgatum* grew comparatively poorly both in monoculture and mixtures (Figure 3.7a). Mixtures with 2 and 3 species had shoot biomass comparable to the full four species mix, except the 2-species mixture of *D. clandestinum* and *P. virgatum*. Furthermore, looking at the species individually in the 2 and 3 species mixes the total biomass is overwhelmingly comprised of *S. nutans* and *A. gerardii* (Figure 3.7b and 3.7c). The four species mix diversity was not reduced in compacted soils in either the second or third season ($p = 0.43$), and neither was there a difference in mix diversity by year and soil treatment ($p = 0.74$). The restoration mix treatment with the greatest density of *P. virgatum* seed had the greatest variability in diversity across mixes varying in *P. virgatum* seeding densities and seasons (Figure 3.8).

Microstegium vimineum biomass was significantly reduced in the third growing season compared to the second ($p < 0$) (Figure 3.9). The exponential growth of the restoration mix in the third season coincided with, and was likely a main factor in, the reduction of *M. vimineum* biomass. Given that the soil compaction did not result in lower restoration mix shoot biomass, we were not surprised that we did not observe an increase in *M. vimineum* biomass with soil compaction ($p = 0.75$). In the restoration mix monoculture plots *M. vimineum* biomass was greater when growing with *D. clandestinum* than when growing with *A. gerardii* or *S. nutans* ($p < 0.05$) (Figure 3.10a), a pattern that continued through the third growing season (Figure 3.10b). Overall, by the third growing season, the biomass of the restoration mixes with three and four species seemed to be sufficient to suppress the growth of *M. vimineum* (Figure 3.11a & 3.11b).

Discussion

We had expected to see a reduction in root and shoot biomass and ground cover of the seeded restoration mix of native perennial grasses on compacted subsoils. Over the three growing seasons we did not find evidence that soil compaction affected the below-ground or above-ground growth of these species grown in monoculture or mixture, nor did it appear to be a factor determining plant community diversity. Given the lack of soil data on unconventional well pads it is possible that the soil conditions of our treatment plots, such as the level of compaction and nutrient availability, were not as extreme as might be found on actual well pads. Our compacted treatment plots were on average 1.53 g cm^{-3}, only 0.10 g cm^{-3} less than Fink and Drohan (2015) found on nine reclaimed unconventional well pads in Pennsylvania. Further research on a wider range of compaction treatments could elucidate thresholds and therefore aid in prioritizing where deep ripping is necessary to establish native grasslands. If soil compaction is more severe we recommend decompaction treatments to escalate site recovery. But given our results, incorporating native perennial grasses can be particularly important as a restoration tool where decompaction is not feasible, such as on steep slopes, or not economical at scale.

Our observations of the changes in root biomass over the second and third growing seasons were informative of the establishment dynamics of the restoration mix. We had expected to observe a reduction in root biomass due to compaction that would impact the above ground establishment success of the mix. Instead we found evidence that the roots of these species can tolerate moderate compaction as proposed by (Thorne et al., 2013). Regardless of the soil compaction we did observe variability in root biomass among the mix species. The only cool-season species in the mix, *Dichanthelium clandestinum*, had the lowest root biomass in the second season, yet by the third season was equivalent with *Sorgastrum nutans*. Even though

D. clandestinum overall grew slower than the warm-season species in this study we expect that given a few more growing seasons it would be more competitive, particularly in soil with a lower pH. In fact, some land managers are reluctant to plant this species as it is known to dominate restoration mix composition over time (Barlow et al., *in review*). The lag in establishment of this species is likely beneficial when seeding with the native warm season grasses.

We were surprised that the root biomass of *Panicum virgatum* was comparable to the other warm season grasses given the low shoot biomass of this species in monocultures and mixtures. *Panicum virgatum* has thick crown roots that can make up half of the plant's total dry weight (Frank et al., 2004), which could be the reason for the higher root to shoot ratio compared to the other species. It is clear that the growth trajectory of these slow-to-establish perennial grasses unfolds in species specific ways. Given the disproportionately large investment in root growth by *P. virgatum*, we would expect the shoot biomass production of *P. virgatum* will continue to grow with successive seasons. The 'Shelter' northern upland cultivar of *P. virgatum* used in our study has been shown to have lower biomass compared to lowland cultivars (Fike et al., 2006). Cultivar selection, in conjunction with climate, should be a significant consideration when designing mixes. In a biofuels study in Iowa *P. virgatum* 'Shelter' had an average biomass yield among the 16 upland cultivars 'Shelter' (Lemus et al., 2002), and as opposed to our study, 'Shelter' consistently out produced two cultivars of *Andropogon gerardii* in Canada (Deen et al., 2011). If we had chosen a lowland cultivar with high biomass, we would likely have seen *P. virgatum* dominance in the plant community, as is often the result in tallgrass restoration (Baer et al. , 2005).

Across seed mix and soil treatments all four species displayed similar trends in shoot biomass, whether grown in monocultures or mixtures. When either *A. gerardii* or *S. nutans* were sown with *P. virgatum* and, or *D. clandestinum*, they had significantly greater biomass. When *A. gerardii* or *S. nutans* were sown together in a mix neither seemed to outcompete the other as they produced equivalent shoot biomass yields. Our results for *S. nutans* are contrary to what others have found in biofuel or forage studies where *S. nutans* was outcompeted by *A. gerardii* and *P. virgatum* (Hong et al., 2012; Springer et al., 2001).

Our results for *A. gerardii* are consistent with other restoration research studies on this dominant prairie species (Baer et al., 2016; McCain et al., 2010).

The typical growth rates of native perennial grasses, minimal shoot growth in the first season followed by rapid growth the second and third (Skousen & Venable, 2008), was evident in both the biomass and ground cover in our study. Following their slow growth the first field season, we observed extensive shoot growth and ground cover of restoration mix species while weed biomass remained low and weedy plant cover declined with each successive season. We used herbicide to control weedy cool-season grasses, but mowing can also be an effective weed suppression tool for establishing perennial grass mixtures (Török et al., 2012). However, mowing should be timed so that it does not coincide with weedy plant seed ripening and dispersal. Bare ground was on average nearly 30% in the first season but would have been higher, or the site would have been weedier, without a cover crop. Incorporating a fast-growing annual with a native perennial mix is often necessary for achieving ground cover in first year of establishment. Selecting species that will not be competitive with the slower establishing perennials is critical. We used *Avena sativa* as a cover crop and it did not persist into the second year when the native grasses started to fill in with ground cover. Although we did not test the direct effect of *A. sativa* on native grass perennial establishment others have seen no effect on establishment (Espeland et al., 2017), but minimal effect on weed control (Miesel et al., 2012). *Lolium* species are commonly used for immediate ground cover, but often outcompete the desired plant community (Beyers, 2004; Matesanz and Valladares, 2007). Given the restoration success we and others have seen with *A. sativa* we recommend this species as a cover crop for ground cover when seeding perennial grasses.

Reclaimed sites must have sufficient resistance to plant invasions to not create a new source of propagule pressure for the broader landscape. During this early establishment period we found that monoculture plots with *D. clandestinum* were the least invasion resistant. *Dichanthelium clandestinum* and *Panicum virgatum* were slower to establish aboveground biomass and enabled the invasive phytometer plant *M. vimineum* to expand in density and cover. If invasive propagule pressure is high at the early stages of establishing a restoration mix it will be important to include competitive perennial grass species like *S. nutans* and *A. gerardii* in the mix to limit the expansion of the invasive plant infestation and to minimize its adverse impacts on establishing the restoration mix.

Invasive plants typically have high rates of fecundity, and this is certainly the case with *M. vimineum*. Our previous research indicates that this species is spreading and becoming more and more common along disturbance corridors (Barlow etal., 2017; Rauschert et al., 2017). In a survey of plant communities on a gas pipeline we found that when *M. vimineum* was present at the time of establishing the restoration mix, it persisted and spread, particularly in low pH soils, eight years post revegetation (Barlow et al. *in review*). Complete control of weedy plants in natural systems is neither pragmatic or economical. Therefore, identifying restoration mixes that can tolerate *M. vimineum* infestations at the time of establishment and aggressively grow and suppress the invasive over time will help manage this and other invasive plants. Given these results we suggest that land managers consider prioritizing weed control measures during native plant establishment where soil environments are particularly resource-limited and exhibit extremes in soil pH.

Conclusions

Soils degraded by energy development will continue to be a significant challenge from land use conversion (Trainor et al., 2016). Site reclamation with seed mixes that can provide multiple ecosystem functions is a top priority for ecological restoration. Establishing diverse, native perennial plant communities with deep-rooted grasses can aid in soil reclamation, water retention, provide habitat and contribute to ecosystem integrity. We demonstrate that during the early establishment phase native perennial grasses can tolerate compacted soils and resist invasion from an annual grass, serving as the foundation for native wildlife habitat. Successful establishment, as was evident in our research, requires an effective cover crop in the first year to compete with weedy flora while the native perennial grasses produce little shoot biomass. The restoration mix grew exponentially over the three seasons and by the third season had outcompeted the weedy flora, including the invasive annual grass, setting the trajectory for a stable plant community.

Such success in compacted soils will vary with soil moisture availability, more severe compaction stress, additional disturbances, and greater invasive propagule pressure. As thousands of unconventional well pads will need to be reclaimed over the coming decades, continued research will be needed to guide the restoration of soils and plant communities, to knit together the ecological functions and services of fragmented forests and fields.

Tables & Figures

Figure 3.1. Root biomass (g) across soil depth averaged across the four native perennial species grown in monoculture.

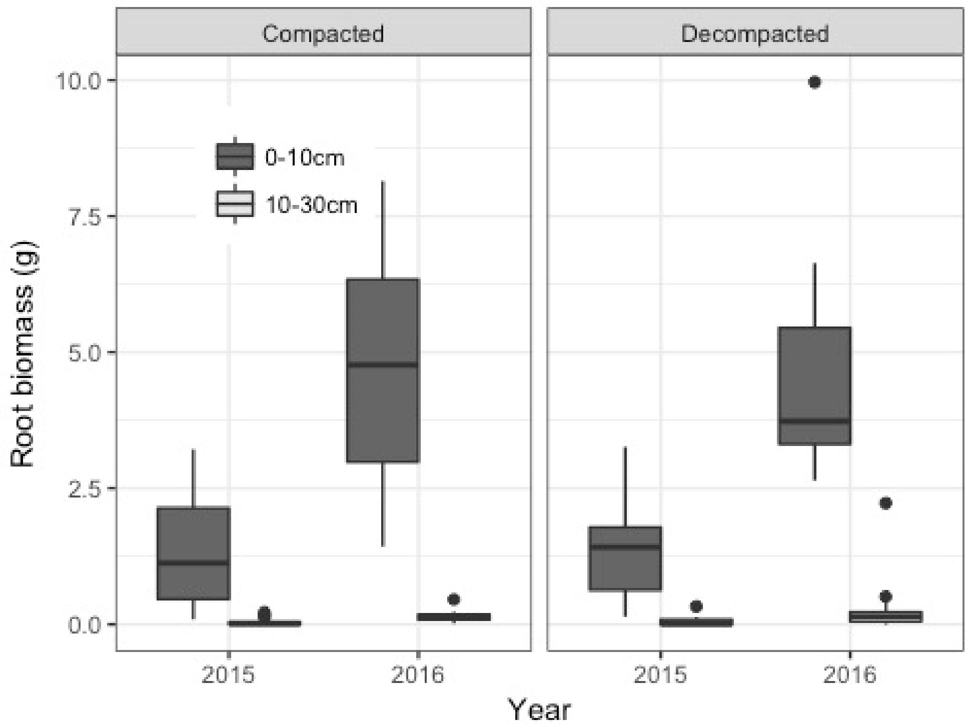
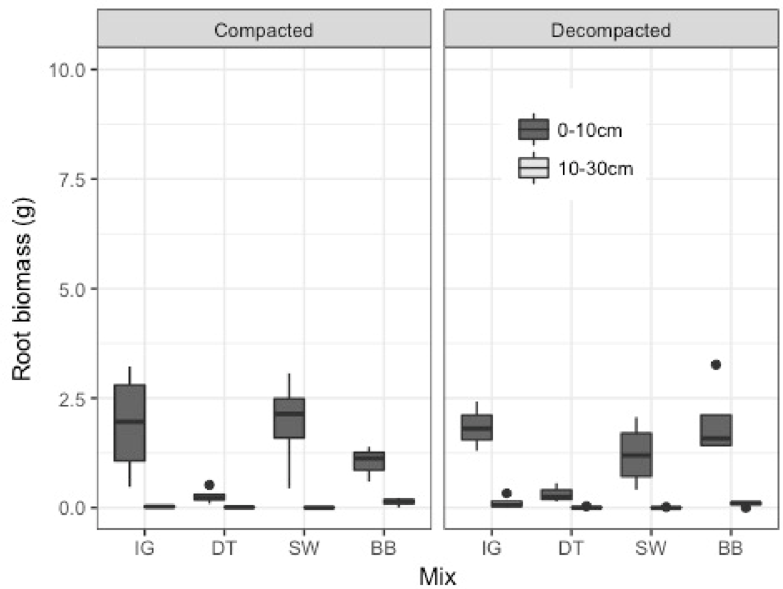


Figure 3.2. Root biomass (g) from monoculture plots across soil depth in 2015 (a) and 2016 (b). IG = Indiangrass (*S. nutans*), DT = Deertongue (*D. clandestinum*), SW = Switchgrass (*P. virgatum*), and Big bluestem (*A. gerardii*).

a.



b.

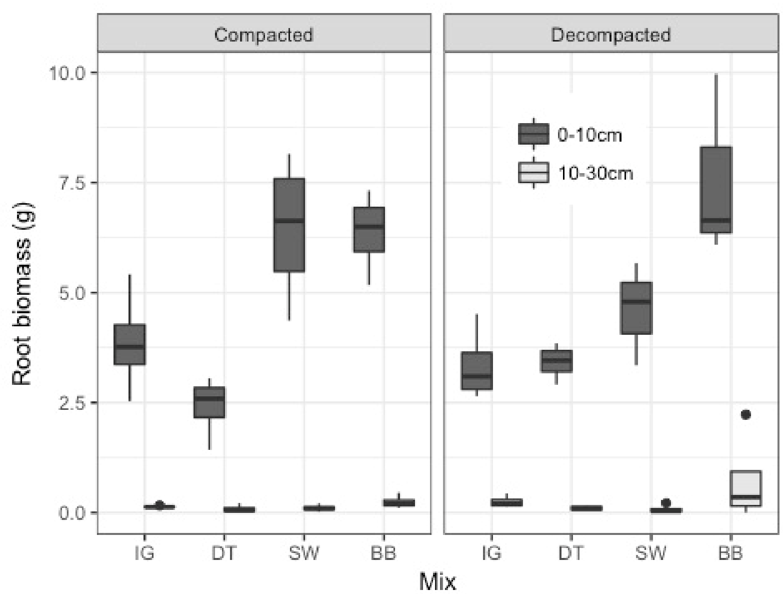


Figure 3.3. Weed seed bank counts from the replicate blocks prior to establishing the restoration seed mix.

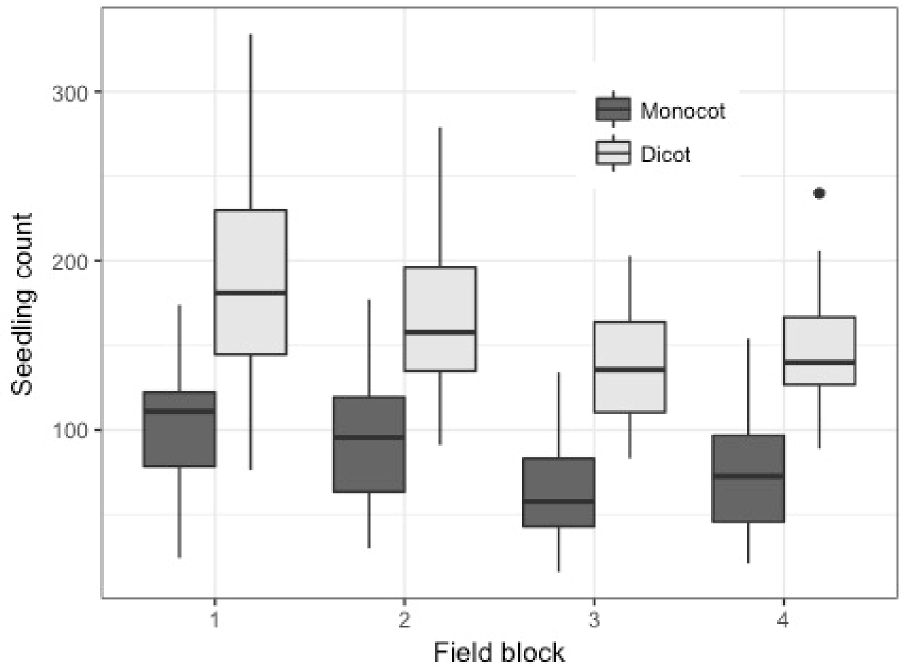


Figure 3.4. Four species mix and weed shoot biomass (g) and ground cover across time and soil treatment. The shoot biomass of the mix in 2014 is the oat cover crop.

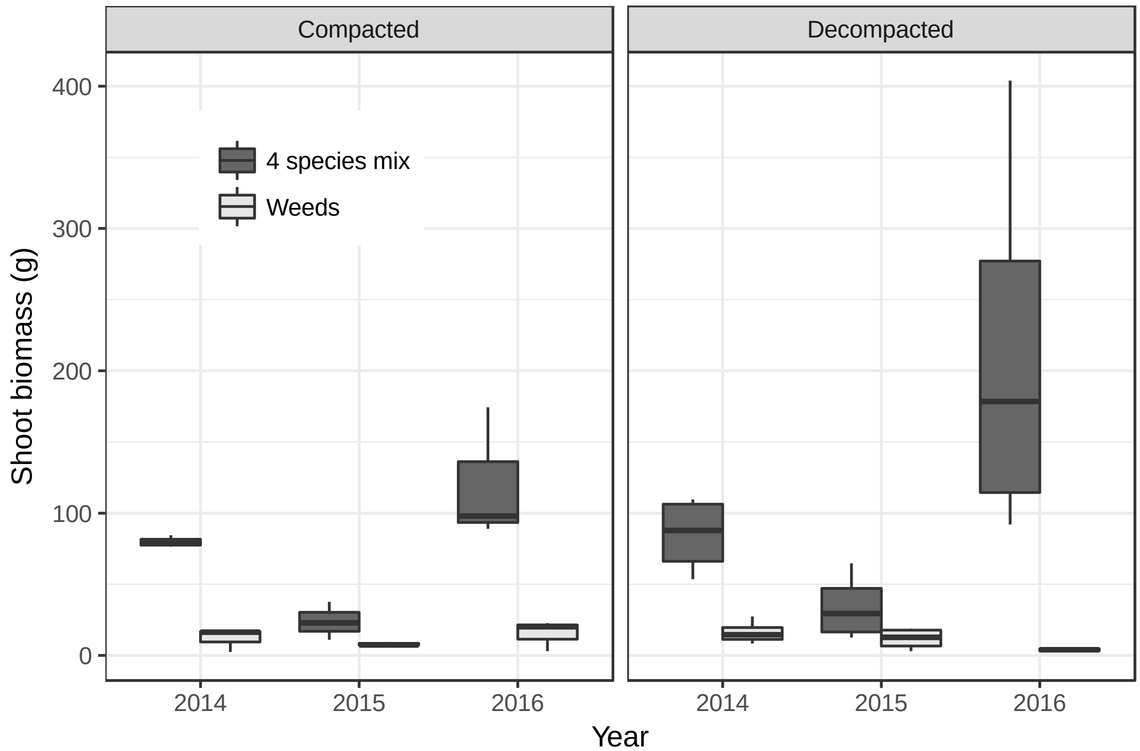


Figure 3.5. Ground cover of the four species native mix, weeds, and bare ground. The mix ground cover in 2014 is the oat cover crop.

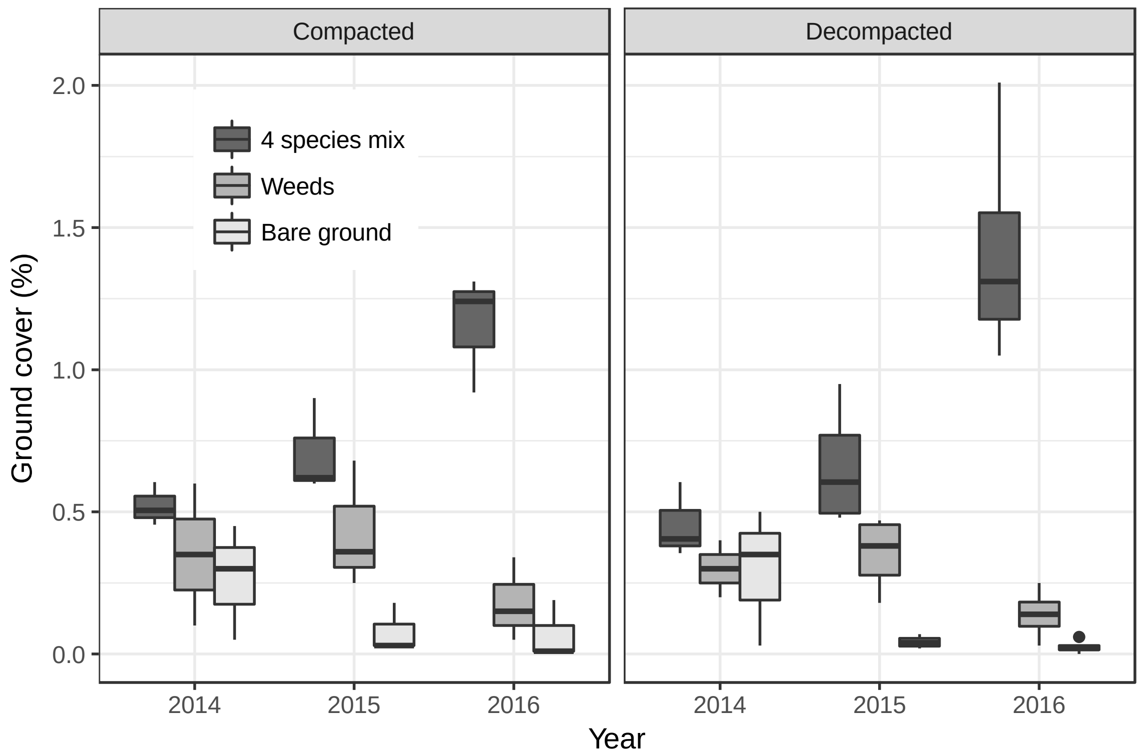


Figure 3.6. Across seed mix treatments plots with compacted subsoils had more bare ground than decompacted plots (p = 0.03) in the third season.

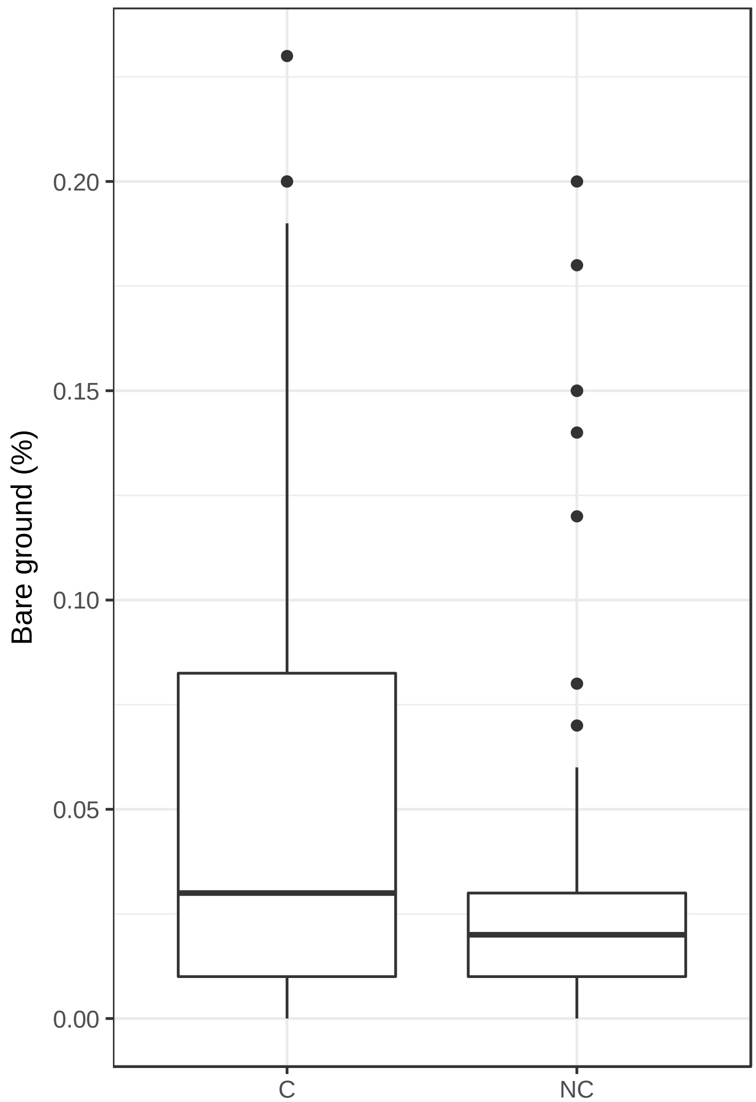
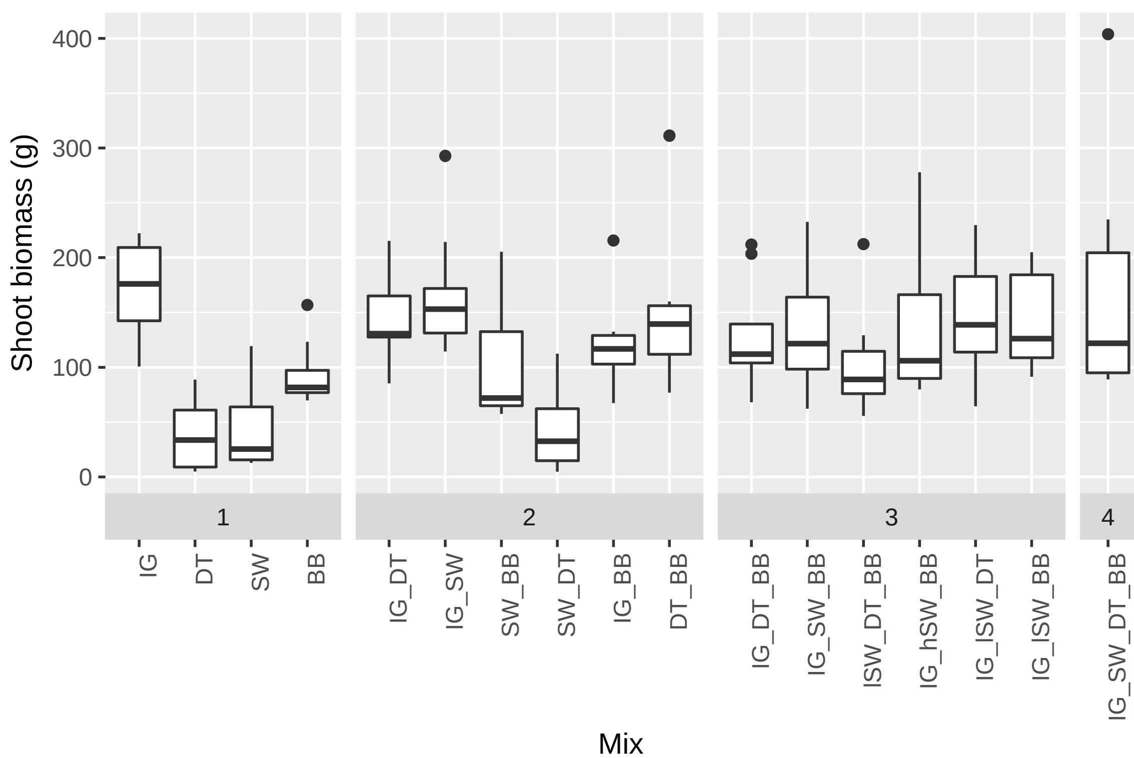
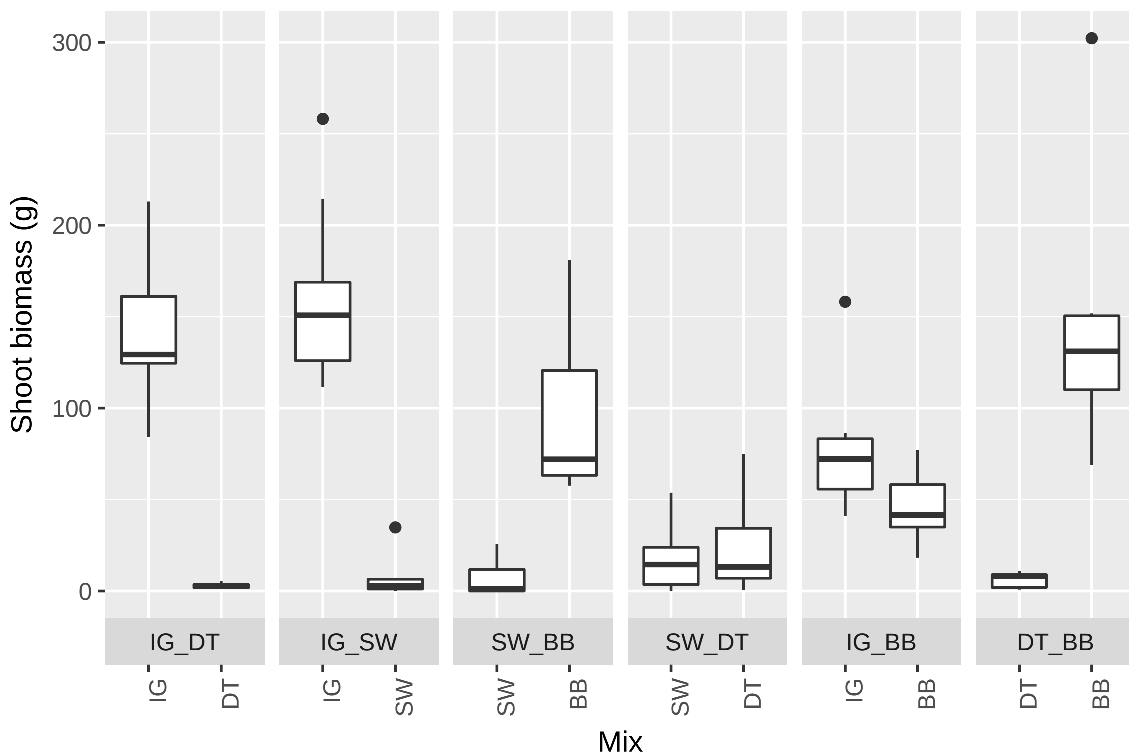


Figure 3.7. Mix shoot biomass (g) in the third growing season. (a) Mix species in monoculture and all 2, 3, and 4 species combinations. Mix components of 2 species combinations (b) and 3 species combinations (c). IG = Indiangrass (*S. nutans*), DT = Deertongue (*D. clandestinum*), SW = Switchgrass (*P. virgatum*), lSW = low seeding rate, hSW = high seeding rate, and BB = Big bluestem (*A. gerardii*)

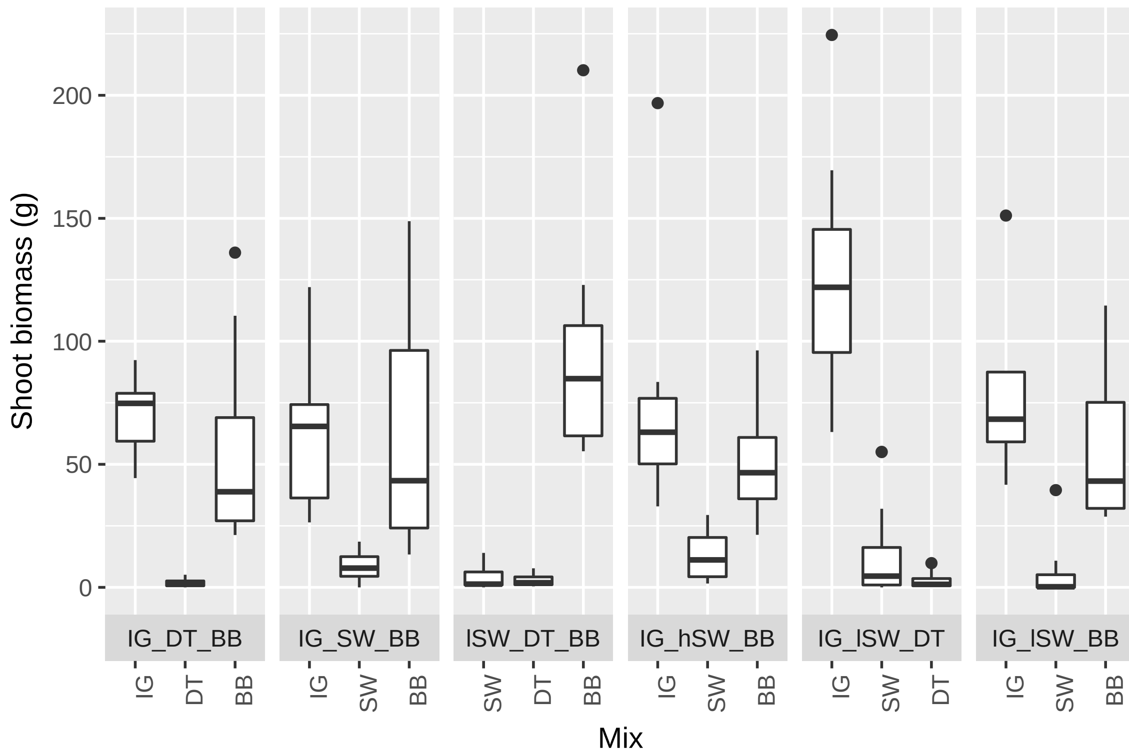
a.



b.


Shoot biomass (g)
300
200
100
0
IG_DT
IG_SW
SW_BB
SW_DT
IG_BB
DT_BB
IG
DT
IG
SW
SW
BB
SW
DT
IG
BB
DT
BB
Mix

c.



Shoot biomass (g)
200
150
100
50
0
IG_DT_BB
IG_SW_BB
lSW_DT_BB
IG_hSW_BB
IG_lSW_DT
IG_lSW_BB
IG
DT
BB
SW
Mix

Figure 3.8. Seed mixes with *Panicum virgatum* seeding densities equivalent to, higher than, and lower than the other tallgrass species, *Andropogon gerardii* and *Sorghastrum nutans*.

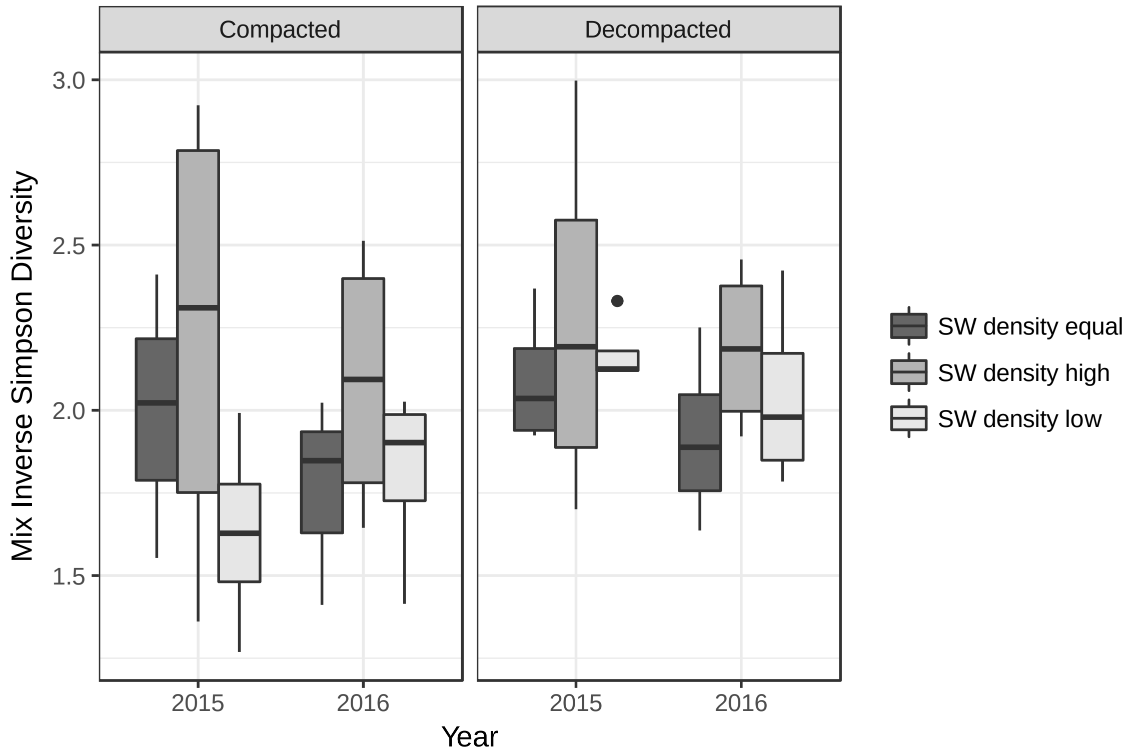


Figure 3.9. Across seed mix treatments *Microstegium vimineum* biomass (g) dropped from an average of 17.98 g in 2015 to 0.61 g in 2016.

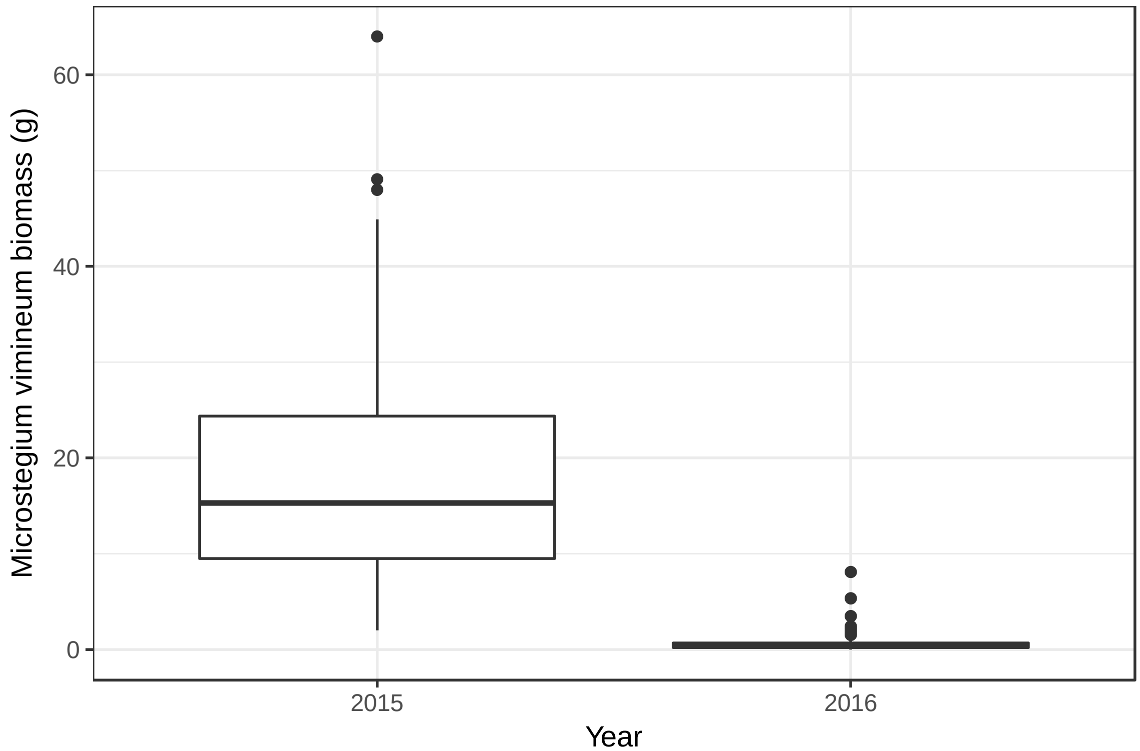
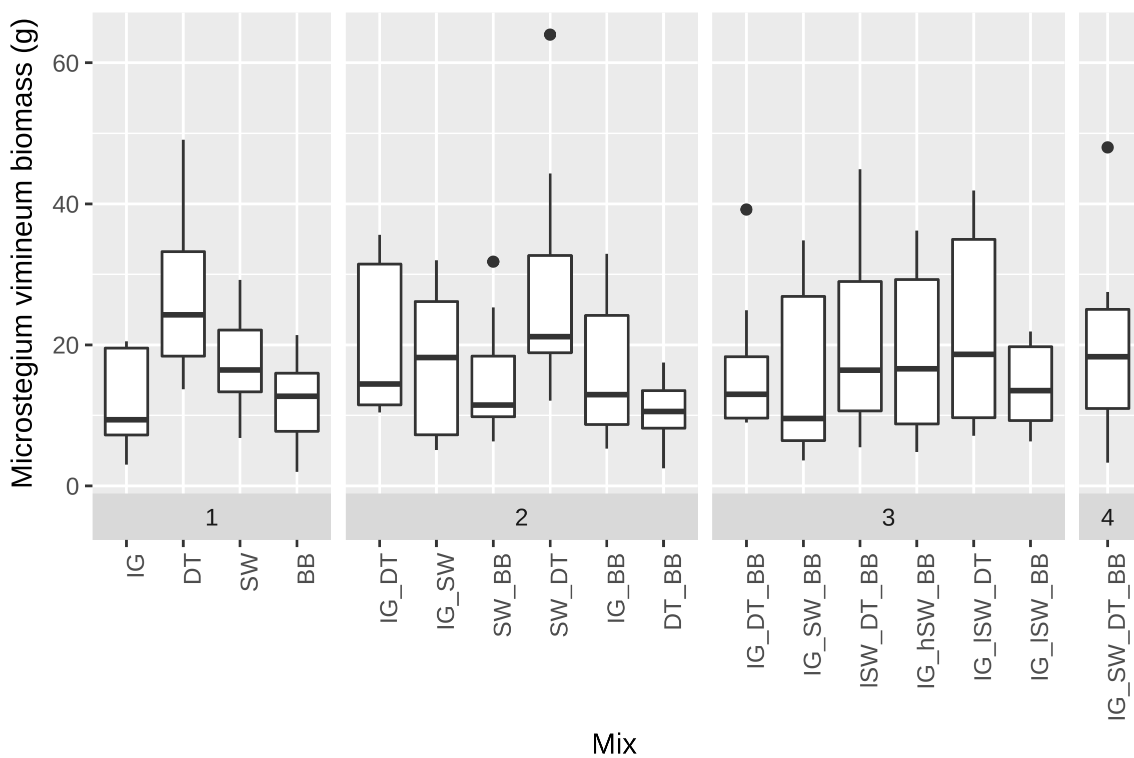
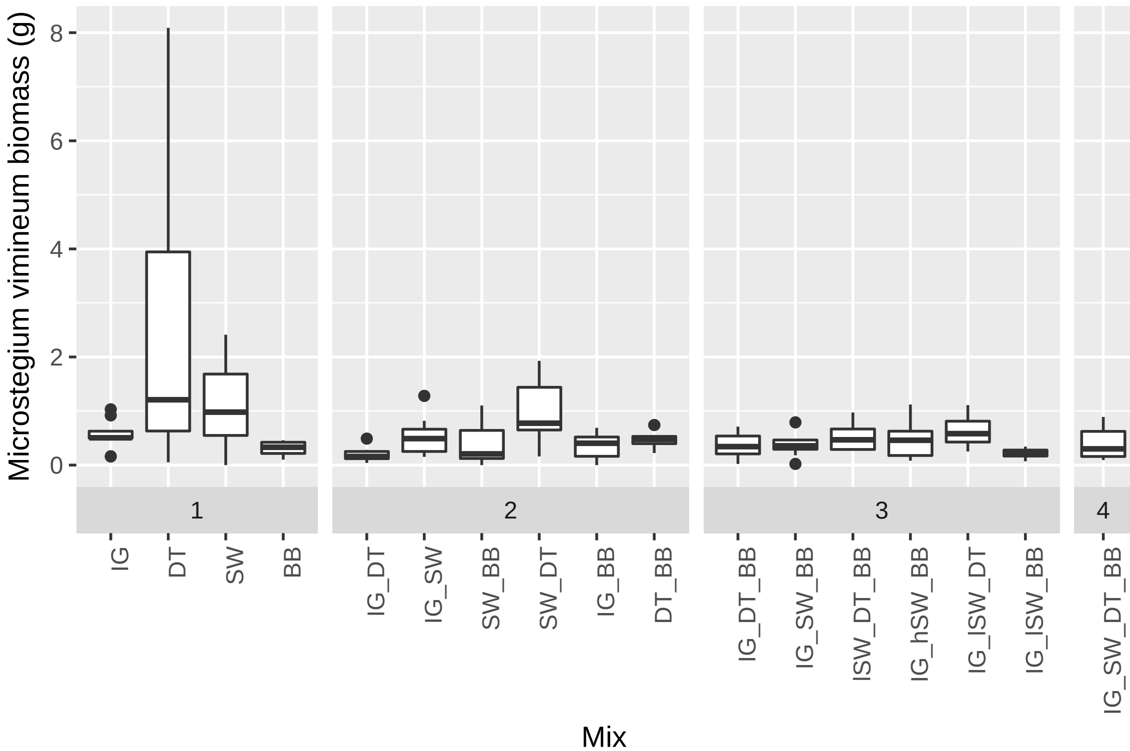


Figure 3.10. (a) *Microstegium vimineum* biomass (g) harvested in 2015 that was seeded into native perennial grass monocultures and mixtures in 2014. (b) *Microstegium vimineum* biomass (g) harvested in 2016 that was seeded into native perennial grass monocultures and mixtures in 2015. IG = Indiangrass (*S. nutans*), DT = Deertongue (*D. clandestinum*), SW = Switchgrass (*P. virgatum*), lSW = low seeding rate, hSW = high seeding rate, and BB = Big bluestem (*A. gerardii*)

a.



b.


Microstegium vimineum biomass (g)
8
6
4
2
0
1
2
3
4
IG
DT
SW
BB
IG_DT
IG_SW
SW_BB
SW_DT
IG_BB
DT_BB
IG_DT_BB
IG_SW_BB
ISW_DT_BB
IG_hSW_BB
IG_ISW_DT
IG_ISW_BB
IG_SW_DT_BB
Mix

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Chapter 4

Soil pH influences patterns of plant community composition after restoration with native-based seed mixes

Abstract

Reclamation of highly disturbed lands typically includes establishing fast-growing, non-native plants to achieve rapid ground cover for erosion control. Establishing native plant communities could achieve multiple ecosystem functions beyond soil erosion, such as providing wildlife habitat. Disturbed corridors through a landscape, such as pipelines, present unique challenges for establishing native plant communities given the heterogeneity of soil environments and invasive plant propagule pressure. We created a structural equation model to address multiple related hypotheses about the influence of soil traits (soil pH) on plant community composition (seeded mix diversity and cover, invasive plant historic and current density) of a highly disturbed landscape corridor restored with native species. To test our hypotheses we conducted a plant survey on a gas pipeline crossing two state forests in north-central Appalachians that had been seeded with a native-based mixture eight years prior. Low soil pH was a strong predictor of density of the invasive annual plant, *Microstegium vimineum*, and, influenced the seeded mix cover as a strong predictor of the realized mix diversity eight-years post-establishment. The seeded mix cover was negatively correlated with *M. vimineum* dominance in the plant community. Overall our data provide evidence that native-based grass and forb mixtures can establish and persist on a wide range of soil environments and in competition with invasive plants. Advancing knowledge on restoration methods using native species is essential to improving restoration practice norms to incorporate multifunctional ecological goals.

Introduction

Improved restoration practices and policies are needed to rehabilitate ecosystem functions and services in heavily disturbed landscapes. Ecological restoration is particularly challenging after the complete removal of vegetation and severe disturbance of soil physical properties and associated soil ecology (Schlesinger 1986; Suding & Hobbs 2009). Such soil disturbance is typical of natural resource extraction, like that associated with coal mining (Lupton et al. 2013), and more recently with shale oil and gas (Souther et al. 2014; Moran et al. 2015). Restoration of disturbed sites should be multi-functional by design (Lovell & Johnston 2009) and have a process-based approach which takes a landscape perspective (Hobbs & Norton 1996). A comprehensive approach would consider the complexities of community assembly in light of the disturbance (Hobbs et al. 2007), restore dynamic soil properties and hydrology (Drohan & Brittingham 2012; Fink & Drohan 2015), and successfully establish plant communities with resistance to plant invasion (Jordan et al. 2011). The establishment of stable native plant communities that match the soil, climate and fauna of the region could serve as a baseline for ecosystem restoration. Here we investigate the establishment and persistence of a native-based plant mix along a gas pipeline and the invasion resistance of that plant community.

Pipeline infrastructure extent and restoration challenges

Currently, there are more than 349,228 km of gas pipeline in the US, and pipeline capacity is expected to increase 1.0-1.2 billion m^3 of natural gas per day between 2015 and 2030, some of which will require new pipeline (US DOE 2015). A disproportionate amount of that development will occur in the north-central Appalachians across the Middle Atlantic states of Pennsylvania and West Virginia (Feijoo et al. 2018). Transmission lines represent a significant portion of the shale oil and gas footprint (Langlois et al. 2017), creating linear corridors of grass and shrublands across a landscape matrix that, in the Eastern US, is comprised predominantly of core, relatively undisturbed forest to a mix of forest, arable fields and riparian buffers. Wildlife sensitive to forest fragmentation in the Eastern US will likely be affected by shale gas infrastructure (Kiviat 2013; Brittingham et al. 2014).

Revegetation should be consistent with the surrounding landscape to contribute to habitat and mitigate the effects of fragmentation. Yet, historically, restoration of coal mines, gas well pads, and gas and electric transmission corridors, particularly in the Eastern US, have relied on inexpensive non-native mixes of fast-growing cool-season grasses and legumes. These species may provide rapid erosion mitigation, but have many unintended consequences for birds, mammals, and invertebrates (Ellis-Felege et al. 2013; Kelt & Meserve 2016). Such practices became the norm in part because restoration regulations prioritize site stabilization (i.e. the 1977 Surface Mining Control and Reclamation Act, Zipper et al. 2011) over a more multi-functional ecological approach.

Ecological restoration with native plant materials on pipelines

Native prairie grasses can provide significant shoot and root biomass (Mahaney et al. 2015), provide rapid carbon sequestration (Guzman et al. 2016), limit invasive plants by increasing litter and reducing N availability (Mahaney et al. 2015), provide habitat for multiple trophic levels (Whiles & Charlton 2006; Fetcher et al. 2015), and support native fauna (Ballard et al. 2013; Kaiser-Bunbury et al. 2017). Research on incorporating native grasses to restore highly disturbed soils, such as mine lands, has been on-going, yet sparse, for several decades (Thompson et al. 1984; Skeel & Gibson 1996; Thorne & Cardina 2011). The long-term plant community dynamics of restored native perennial prairies have been well documented (Kindscher & Tieszen 1998; Trowbridge et al. 2017), but less is known about the range of environmental conditions that contribute to recruitment success and persistence in the various plant cover types, soils, and climate of Eastern forests (Thorne & Cardina 2011; Miller 2013).

Since transmission corridors represent transects across a heterogeneous landscape, studies of the persistence of restoration mixes along this heterogeneity gradient would aid our ability to identify mixes and species within a mix that perform well across the range of site variation (Stuble et al. 2017). Soil pH is highly variable in the diverse array of lithology and topography of the Appalachians and will likely be an important factor in species recruitment and establishing sufficient cover of the desired species. Soil pH is a primary driver of plant diversity and dominance given the plant species pool (Gough et al. 2000; Pärtel 2002; Sebastia 2004), Understanding the effects of soil pH on mix diversity over time, mix cover, and invasion resistance will be crucial to expanding support for the use of native plants.

Mix establishment and invasion resistance to Microstegium vimineum

Severe soil disturbance could impair recruitment success of the desired species, resulting in an increased availability of soil resources for weedy and invasive species. In prairie restoration, perennial tall-grass communities have been found to have strong resistance to plant invasion (Blumenthal et al. 2005; Foster et al. 2015). Unlike most other non-native herbaceous species of concern in temperate northeastern forests, the shade tolerant C_4 annual grass, *Microstegium vimineum*, has potential for persistence in forest understories (Leicht et al. 2005) and is therefore of concern to forest managers on public and private lands from southern New England to the Southeastern U.S. Invasion of *M. vimineum* on forest soils has been found to increase soil pH, carbon cycling, and nitrification leading to a reduction in microarthropod and plant diversity (Ehrenfeld et al. 2001; McGrath & Binkley 2009; Warren et al. 2010). Changes in a plant community could impact local food networks. For example, Simao et al. (2010) found that when disturbed soil was replanted with native forbs *M. vimineum* indirectly reduced arthropod abundance because of the loss of native plant richness. The niche requirements of *M. vimineum* include greater growth and seed production in higher light and to a lesser extent with greater soil water (Warren et al. 2010). Previous research has shown that *M. vimineum* presence and abundance is positively correlated with soil pH (Cole & Weltzin 2004; Nord et al. 2010), but can persist in more acidic soils (Gibson et al. 2002) and can raise soil pH (Ehrenfeld et al. 2001; McGrath & Binkley 2009).

The high light availability of pipeline corridors, particularly in low laying areas with high pH could result in new source sites that can spread into adjacent, less disturbed forest interior.

Eradicating *M. vimineum* is not realistic, but impeding spread and dominance with a competitive native plant community would allow for a suite of other species to provide ecosystem function benefits to the landscape. Including native dominants could aid in suppressing invasive plants (Simmons 2005). For example, *Dichanthelium clandestinum* has been shown to out compete *M. vimineum* in full sun, with the reverse outcome in full shade (Flory et al. 2007). A plant mix that contains species with the same active growth period, such as the C_4 *P. virgatum*, might also impede the expansion of *M. vimineum* (Funk et al. 2008). A dense mat of leaf litter has been shown to prevent germination and establishment of *M. vimineum* (Warren et al. 2010). Native warm season grasses have high leaf litter and might help to impede new seedling establishment of *M. vimineum*.

To evaluate the effect of soil pH on native-based mix species recruitment, persistence, and invasion resistance we assessed plant community composition eight years after reclamation of a gas pipeline in Pennsylvania state forests with a spatially explicit dataset of invasive plant density spanning 5 years. To investigate these complex species-site interactions we used two structural equation models (SEM). We first tested the interplay between soil pH and plant community composition; categorized as, the invasive grass *M. vimineum*, the background flora (not intentionally seeded, and excluding *M. vimineum*), and the seeded mix (SEMa). We hypothesized that sites with moderate to high soil pH would result in a reduction of the mix cover, in part caused by a positive response by the background flora cover and dominance of *M. vimineum* (Figure 1.). Each arrowhead in the SEM indicates an outcome variable. Therefore, in SEMa we assessed the direct relationship of soil pH to mix cover, as well as the indirect effects from the non-seeded plant community on mix cover. Secondly, we expanded the variable relationships in SEMa to test more indirect effects of soil pH on the native mix cover. In addition to greater background flora reducing mix cover, we hypothesized in SEMb that greater soil pH would have a positive effect on mix cover via mix species recruitment, and that historically greater densities of *M. vimineum* could be predicted by soil pH and would in turn predict current *M. vimineum* cover (Figure 1.).

Methods

Survey sites

Survey sites were selected along the Dominion Transmission, Inc. pipeline within the Rothrock (RSF) and Tuscarora State Forests (TSF) of central Pennsylvania, USA. The pipeline was installed in 2007, alongside Spectra's Texas Eastern Transmission pipeline completed in 1961. A 14.5 km section of that pipeline runs through RSF and 7 km section through TSF. Soils typically consist of low pH (3.0 – 4.5) Ultisols and Inceptisols with varying rock fragment contents (0-90%) and a wide variability in depth to bedrock (Ciolkosz et al., 1990). Average annual temperature in central Pennsylvania is 13.2 ºC and average annual precipitation is 103 cm (NOAA, 2018).

The following spring a native-based seed mix was broadcast along the pipeline at a density of 16.8 kg ha^{-1} then mulched with 3362.5 8 kg ha^{-1} of straw mulch (Table 1). In 2009 steep slopes in RSF were reseeded with a hydroseeder where establishment was poor with a different native-based mix at 56 kg ha^{-1} (Table 1). The section of the pipeline in RSF was mowed annually in the summer. The section in TSF was mowed only twice since establishment.

Just prior to the installation of the pipeline GAI Consultants, Inc. mapped the extent of the *M. vimineum* infestation along the planned pipeline corridor through RSF and TSF. Data collection sites were geo-referenced forming a contiguous grid of 22.9 m x 45.7 m plots along the pipeline where 22.9 m is the width of the pipeline corridor, resulting in 149 sites in TSF and 294 sites in RSF. GAI Consultants, Inc. then conducted an invasive weed survey (including *M. vimineum*) at the resolution of the "site" classifying the extent of invasion as: absent (0% cover), trace (<1% cover), low (1-5% cover), moderate (5-25% cover), or high (25-100% cover) according to the Montana Noxious Weed Survey and Mapping System (Cooksey & Sheley 2002). Each year from

2009 through 2013 they repeated the survey at the "site" level of the contiguous grid along the pipeline corridor. We chose 76 of those sites that varied by 2 levels of density of *M. vimineum* ground cover from 2009 to 2013 (Figure 2). Sites were categorized as having "low" density where *M. vimineum* was either; absent from 2009-2013, or the density changed from low or absent in 2009 to trace in 2013, or trace/low in 2009 to trace/low in 2013 (39 sites, Rothrock n=27, Tuscarora n=12). Sites were categorized as having a "high" density where *M. vimineum* was either; absent in 2009 but had expanded to moderate/high in 2013, or remained moderate/high from 2009 to 2013 (37 sites, RSF n=22, TSF n=15).

Soil sampling and plant community surveys

We surveyed each of the selected 76 sites along the pipeline corridor continuous grid for the soil pH and plant community composition variables of the SEMs. Soil samples were taken in April 2017. At each site we collected soil from 5 random locations along the linear transect used for the plant surveys, described below, and removed a soil sample from the surface to a 10 cm depth with a 0.02 m internal diameter soil corer. Soil samples were homogenized by site, and air dried. Mineral soil pH was measured in a 1:2 (weight:volume) 0.01 M $CaCl_2$ using a VWR SympHony pH meter (Soil Survey Staff 2014).

Plant surveys were conducted from early August through early October 2016. Prior to conducting the field work we marked a starting point for the transect within each of the 76 sites in the geographic information system application, QGIS (QGIS Development Team 2009). Once on site we used a tape measure to mark a 20 m linear transect running south from the starting point for the plant community surveys. Similar to Begley-Miller et al. (2014) we used the line-intercept method to quantify plant cover by species, both species from the seed mixes and the background flora. For each species, all plants with vegetative parts within 0.2 m of either side of the tape were included and recorded as the length on the measuring tape, to the nearest 0.05 m. For example, if species *i* was recorded along the tape from 15-75 cm and from 150-300 cm, the total cover for that species was 210 cm. “Mix cover” is the proportion of the transect cover by species from the seed mix, calculated as the sum of all mix species along the 20 m transect divided by the length of the transect, therefore cover could be over 100%. The “background flora” is the proportion of the transect covered by species not in the seed mix, except for *M. vimineum*. *Microstegium vimineum* cover was determined in the same manner on the transect, but was transformed to a binary variable, “*M. vimineum* dominance”, for the model; non-dominant (<25% cover) and dominant (≥25% cover). This split was based on methods used by GAI Consultants, Inc., as described above, where *M. vimineum* cover was greater than 25% the site was categorized as high density to remain consistent with the historical dataset. Individual species cover was used to determine the recruited seed mix species diversity. “Mix diversity” is based on individual species cover and is reported as a measure of Simpson diversity ($D = 1 - \sum_{i=1}^{s} p_i^2$) in which p_i is the proportion of individuals in the i^{th} species and S is species richness.

Statistical analysis

We report first on seed mix recruitment by species and overall plant community composition trends on study sites. As study sites were selected based on the variation in the historical density levels *M. vimineum*, and the SEMs are structured to assess the influence of soil pH, we report on composition trends based on these factors. We note the number of sites that shifted from low to high *M. vimineum* density from 2013 to our 2016 survey, and vice versa. As plant cover could indicate the overall potential growing conditions we tested for differences in total plant cover on sites with low versus high density changes of *M. vimineum*. Additionally, we predicted that plant community diversity would rise with soil pH. For both tests we used linear mixed effects models, with the forest district (n=2) as a random effect (‘nlme’ R package, Pinheiro et al. 2016).

The piecewise structural equation models (SEM) were constructed as in Figure 1 according to Lefcheck (2016). SEMs contain multiple linear models with variables that can be used as both predictor and outcome variables, indicated by arrows in the model Figure. Response variables were tested for normality with the Shapiro-Wilk test and transformed as needed. “Mix diversity” and the “background flora cover” were transformed using Tukey's Ladder of Powers with the transformTukey function in the ‘rcompanion’ package (Mangiafico 2016). Each SEM used linear mixed models for response variables with normal distributions, or binomial mixed models for binary outcomes, and included the forest district (n=2) as a random effect. Piecewise SEMs model fit uses the Fisher C test statistic which incorporates the p-values of all linear models and is indicated by $p > 0.05$ (Lefcheck 2016). All data analyses were performed in R version 3.3.2 (2016) with RStudio version 1.0.136 (2016).

Results

Soil pH filters seed mix establishment and plant community composition

Soil pH across the survey sites ranged from 3.89 to 7.19, with a median of 5.63. In the SEMs we used soil pH as a continuous predictor of broad compositional changes of plant cover and diversity, as well as *M. vimineum* density. We observed a clear split in *M. vimineum* density at soil pH 5 (21 sites), with greater density below pH 5 (55 sites). Given that our SEM hypotheses focus on the effects of soil pH and *M. vimineum* on plant communities resulting from restoration with native seed mixes we report on the plant species compositional differences in low (<5) versus moderate to high (>5) soil pH.

Across the 76 sites plant community diversity ranged from 0.49 to 0.92 with a median of 0.82, and the diversity of mix species ranged from 0.00 to 0.83 with a median of 0.66. The mean ground cover of the species from the seed mix across the 76 sites was over 100%, with a range of 2% to 282%. Only 2 sites had less than 5% cover of the mix species, and 13 had less than 50% cover. Sites with a low proportion of mix species cover to total vegetative cover also had a greater proportion of *M. vimineum* dominance (Figure 3). Average *M. vimineum* cover was 49% ±0.05 s.e. across survey transects. Amongst the mix species only *Dichanthelium clandestinum* compared to

M. vimineum with an average cover of 51% ±0.03 s.e., the rest of the mix species each averaged less than 10% (Figures 4 and 5).

Dichanthelium clandestinum was the most frequent species across sites, followed by *Panicum virgatum*, *Chamaecrista fasciculata*, *Sorghastrum nutans*, and *Lotus corniculatus*, all present on over 50 survey sites (Figure 6). These same species, and in addition, *Dactylis glomerata*, *Juncus effusus*, *Scirpus cyperinus* and *Schizachyrium scoparium*, had the greatest cover in the mix all on average over 10% ground cover. *Rudbeckia hirta* and *Elymus virginicus* were both frequently present but had consistently low ground cover. *Andropogon gerardii* was only present on 3 survey sites with low cover. *Elymus riparius*, *Heliopsis helianthoides*, *Lolium multiflorum*, *Persicaria pensylvanica*, *Sporobolus compositus*, and *Symphyotrichum prenanthoides* were not detected in the survey.

The 2016 survey sites were selected according to the historical density of *M. vimineum* (2009-2013), to determine whether it was associated with its current distribution and density. Overall, *M. vimineum* density was stable from the final GAI survey in 2013 to our 2016 survey. During this interval *M. vimineum* populations remained in the "high density" class at 31 sites, 30 sites remained in the "low density" class and only 15 sites changed density classes. Nine sites shifted from low to high density, 2 in Tuscarora and 7 in Rothrock, resulting in 40 of the 76 sites in 2016 with *M. vimineum* dominance. The density of *M. vimineum* declined in 6 sites from high to low, 3 in Tuscarora and 3 in Rothrock. Sites with the historically high density had greater overall plant cover than sites with a low density (t = 3.013, p = 0.004). Yet, when *M. vimineum* cover is subtracted from the overall plant cover there is no difference in plant cover between sites with historically low and high density levels (t = -0.845, p = 0.401).

The background flora on the survey sites were largely native, early successional species and only four of the most abundant species were non-native, *Daucus carota*, *M. vimineum*, *Phleum pretense*, and *Taraxacum officinale* (Figure 5). Although we did not find evidence that diversity of the overall plant community (background flora and seeded mix) was influenced by soil pH ($t = 1.195$, $p = 0.232$), mix diversity increased with greater soil pH. Of the top 15 most frequent species present in low soil pH nine species were part of the seed mix, yet in sites with low soil pH only 5 out of the top 15 were from the seed mix. In low pH sites *M. vimineum* was the most frequent species present, followed by species in the original seed mix *Scirpus cyperinum, Dichanthelium clandestinum*, and *Juncus effusus*. In sites where the soil pH was low the mix species *D. clandestinum*, *Panicum virgatum*, and *Chamaecrista fasciculata* were among the top four most frequently present species, along with *M. vimineum* which was the second most frequent. *M. vimineum* had a wide range of tolerance to soil pH and was present in 90% sites with low pH, and 96% sites with moderate to high pH. *Dichanthelium clandestinum* has a wide range of tolerance to soil pH and was present on 96% of sites with moderate to high pH and 67% of sites with low pH. Non-native components of the mix, *Lotus corniculatus* and *Dactylis glomerata*, thrived on moderate to high soil pH (78% and 64% of sites), but did not on low pH (29% and 19% of sites). *Panicum virgatum*, *Chamaecrista fasciculata*, *Sorghastrum nutans*, and *Elymus virginicus* were present on 65-96% of sites with moderate to high pH, but on sites with low pH were present on only 19-48% of sites.

Native mix cover indirectly influenced by soil pH

The data support both SEMs (Figure 7); SEMa Fisher C = 4.07, $df = 2$, K = 13, $p = 0.131$, and SEMb Fisher C = 24.07, $df = 16$, K = 16, $p = 0.088$. The SEMs identified several mechanisms in which soil pH influences plant community composition. SEMa suggests that soil pH influences mix cover indirectly by a reduction in weedy plant cover in lower soil pH. SEMb further clarifies the role of soil pH on plant community composition; soil pH drives greater recruitment of the seed mix which results in greater mix cover, and the historical density of *M. vimineum* which was greater in acidic soils and resulted in current *M. vimineum* dominance. Two missing paths were identified in SEMb; the historical density of *M. vimineum* predicted reduced recruited mix diversity (unstandardized coefficient -0.076, $p < 0.05$), and the background flora cover predicted a high density of *M. vimineum* (unstandardized coefficient 4.143, $p < 0.05$).

Discussion

Native seed mix establishment success on highly disturbed soils

We found that restoration with a native seed mix was broadly successful across the pipeline with only 20% of surveyed sites having less than 50% coverage of the mix. Most species in the mix are native to the American prairie, or Eastern old fields and pastures, yet were persisting on the highly disturbed forest soils where site conditions ranged from steep slopes with thin, rocky soils to deeper soils typical of low laying positions in this region. Originally sown in 2008 and 2009, many of the restoration mix species persisted as part of the pipeline plant community through 2016. Given the establishment success of species recruitment and coverage we see great potential for incorporating more native seed in restoration mixes.

Influence of soil pH on plant community composition

Our surveys provide evidence that, with time, soil pH filtered the seed mix species altering plant diversity. Soil pH was a strong predictor of plant community composition 8 years after the restoration mixes were sown. Soil pH across our study sites was representative of soils that had been minimally limed resulting in an atypically higher than normal pH value >5. Native soil pH in the study area is typically 3 - 4.5 in undisturbed soils (Ciolkosz et al., 1990).

We found that the impact of soil pH on mix cover was mediated by the historical *M. vimineum* density and the mix diversity. Mix diversity increased with soil pH, and this greater diversity resulted in greater mix cover. Conversely, where soil pH was low, *M. vimineum* had either increased in density much faster during the first four years post planting or remained at high density resulting in *M. vimineum* dominance continuing to 2016. Those sites where *M. vimineum* was a dominant had lower seeded mix cover in 2016. As we do not have initial mix recruitment data we cannot confirm if initial recruitment and persistence of the mix are linked. Some mix species, which were not present in our survey (such as the nurse crop *Lolium multiflorum*), could have had initial high recruitment and yet failed to persist.

Mix species diversity was likely affected by a number of abiotic and biotic factors we did not measure during the time between planting and our survey, such as soil moisture, nitrogen availability, and herbivory. At the time of our survey we observed more weedy species in low pH sites. This suggests that either fast-spreading *M. vimineum* populations limit the establishment of the native mix, or low mix recruitment in acidic soils results in unfilled niche that is filled by *M. vimineum* and other weedy species already present at the site; likely both were factors. We had not hypothesized a relationship between the historical *M. vimineum* density levels and mix diversity, but a negative relationship between the historical *M. vimineum* density levels and mix diversity was identified in SEMb suggesting that *M. vimineum* limited recruitment in early establishment.

We were surprised by the reduction in *M. vimineum* dominance in high pH soils given previous research had reported greater *M. vimineum* abundance on soils with high pH (Nord et al. 2010). In our surveys *M. vimineum* was the most frequent species on sites with lower pH while the proportion of the flora along the pipeline that was comprised of species in the seeded mix increased with soil pH. In fact, compared to more acidic soil sites, the cover of the background flora was 7% lower and *M. vimineum* was 60% less likely to be dominant in increasing soil pH. Mix cover had an 81% reduction in cover with increasing background flora cover, and a 28% reduction when *M. vimineum* was dominant. This effect of mix cover suggests that *M. vimineum* was one of the most competitive species in the species pool in low soil pH, or one of the few in the pool that can persist in low soil pH. We found no difference in total plant cover between the historically high and low density sites when subtracting *M. vimineum*, which suggests that *M. vimineum* is not necessarily out-competing other vegetation, and that the site conditions are not necessarily limiting plant growth in the low density sites.

The background flora cover (here, all flora minus the mix and *M. vimineum*) also declined with soil pH. This overall decline in weedy flora (all background flora, including *M. vimineum*) resulted in greater mix cover. Given SEMa we might presume that soil pH has a stronger effect on the weedy flora than on the mix cover. Yet with the results of SEMb we see that the species in the mix had lower recruitment success in lower soil pH. Mix diversity increased by 6.5% with increasing soil pH, and an increase in mix diversity led to an increase in mix cover by over 600%. We conclude that the mix had greater establishment success in soils with higher pH and were therefore able to compete with *M. vimineum* and other background flora.

Including the historical density of *M. vimineum* in SEMb provides evidence that soil pH was a factor driving community composition since early establishment. Through the four years following seeding for every unit increase in soil pH the likelihood that *M. vimineum* had a historically high density fell by 69%. Low pH sites in contrast had higher *M. vimineum* historical densities and low mix establishment. The odds that a site with a historically high density in 2013 had a dominant population of *M. vimineum* in 2016 was 16 times greater than a site with a historically low density. Over time *M. vimineum* could play a role in changing soil environments by raising the soil pH (Ehrenfeld et al. 2001; McGrath & Binkley 2009) and increasing the suitability for native species. Further research is needed to parse the plant-soil interactions for invasions in native plant communities to inform management.

Implications for mix design, monitoring and management

The lower mix diversity in acidic soils has implications for mix design. We recommend including more species that are better adapted to acidic conditions to increase the cover of the desired plant community and reduce the cover of weedy flora. *Dichanthelium clandestinum*, which is known for tolerance to acidic soils (Sankaran & Ebbs 2007), was by far the most frequently present (95% of surveyed sites) and the dominant species from the mix with an average of 50% cover across survey sites.

The *Panicum virgatum* cultivar 'Shelter' used in the mix was released in 1978 by the USDA NRCS for wildlife cover and can be grown on "shallow, acid and droughty soils" (USDA-NRCS 2015), which represents growing conditions on soils found on many steep slopes of the Ridge and Valley or Appalachian Plateau physiographic province. We did not explicitly test for interspecific competition but we did see in our survey data that *P. virgatum* was prevalent throughout the pipeline corridor, second only to *D. clandestium* in frequency and cover. In tallgrass prairie restoration this species has been known to dominate (Baer et al. 2005), but not necessarily limit diversity (McCain et al. 2010). For highly degraded soils and steep slopes where mix options that can achieve both erosion control and wildlife benefit may be limited *P. virgatum* would be a good choice. If varying mix species proportions along a pipeline corridor with growing conditions less limited by moisture and soil pH, reducing the number and density of dominant grass species (e.g *D. clandestium* and *P. virgatum*), may result in a more diverse plant community (Dickson & Busby 2009) and support a broader range of wildlife habitat.

Andropogon gerardii is less tolerant to acidic soils (optimal range 6.0-7.5) and was only found on 3 sites in Tuscarora with less than 6% cover and was not present in Rothrock. Anecdotally, we only saw a few other small populations of this species while walking the entire length of the pipeline through both state forests. Because *A. gerardii* is pH and drought sensitive (Thorne & Cardina 2011) we do not recommend incorporating the species into restoration mixes for highly disturbed soils with low inputs and maintenance.

Chamaecrista fasciculata is often recommended as a native legume alternative to *Lotus corniculatus* and *Trifolium repens* because it provides abundant floral resources. Across the pipeline *C. fasciculata* was just as frequent and abundant as *L. corniculatus* and its cover exceeded that of *T. repens*. We therefore highly recommend replacing non-native legumes in mixes with *C. fasciculata*. While not evident in our dataset, given our data collection methods only captured presence and linear cover, *C. fasciculata* appeared to be poor competitor when growing with the dominant grasses in the mix. Our observations are corroborated by Dickson & Busby (2009) and should be taken into consideration when designing mixes. *Desmodium canadense* is another native legume planted for additional nitrogen fixation and floral resources, yet in our survey it was only present on a few sites. As *D. canadense* is typically a more expensive seed we suggest that unless there is an opportunity for management or planting in strip monocultures, *C. fasciculata* is a better competitor on highly disturbed soils.

As expected, the prairie ruderal species *Rudbeckia hirta* was a frequent part of the realized mix along the pipeline corridor. Yet many of the forbs included in the mix either did not establish because of environmental site conditions or were outcompeted by the dominant grass species in the mix. To improve floral resource these mixes could have included more competitive forbs. *Monarda* species were not included in the mix, but *Monarda fistulosa* is another disturbance-adapted species with spreading rhizomes and significant floral resources (Rowe et al. 2018) that has potential to serve multiple ecosystem functions. To improve restoration with native species further research is needed to identify and test establishment success of a variety of forbs with dominant grasses and long-term persistence of species with high pollinator value.

Understanding the species filtering effects of soil pH combined with the propagule pressure of invasive weeds will be important for guiding management over the long-term. Our data suggest that native perennial grass mixes can persist, even when competing with an invasive annual warm-season grass, such as *M. vimineum*. As we found that low soil pH predicted the historically high densities of *M. vimineum* we suggest including more ecologically desirable species tolerant of acidic soils to compete with *M. vimineum*. When restoring corridors with heterogenous soil environments, such as our study site, it is unlikely managers will be able to treat each environment separately but should seed mixes with a wide range of pH tolerances to ensure coverage of ecologically desirable species.

Monitoring pipelines and other utility corridors in forests will be crucial as corridors could facilitate the invasion of non-native weedy species, not just through the disturbed area itself, but also in the adjacent landscape (Prach et al. 2015). If collecting site soil or plant indicator data is not possible, monitoring for *M. vimineum* could be prioritized based on course predictions of sites with low soil pH. Expect when off-site soils are used for restoration, soil pH can be predicted by topography and parent soil data. Similar to the spread of *M. vimineum* by road graders on forest roads (Rauschert et al. 2017), this species is spreading on pipelines by mowers. Identifying the ideal window for mowing as to not spread invasive weed seed will be important to persistence of the desired plant community.

Restoration of highly disturbed and degraded soils, which require anti-erosion and sedimentation strategies to prevent significant soil loss, should not be limited to short-sighted revegetation goals met with species poor non-native mixtures. Broadening revegetation to ecological restoration is a global issue. The practice of revegetating temperate forests with species of little habitat value and the tendency to arrest succession will have long-term ecosystem consequences. We report here, as others have found, that native-based mixes have great potential for successful establishment, even with the competition of an invasive weed, and should be assessed further for additional ecosystem functions and services. Given the extensive soil and hydrological disturbance associated with expansion in gas development, a significant shift in reclamation practices and norms must occur to support a diversity of ecosystem functions, such as increasing reliance on native plants.

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Tables and Figures

Table 4.1. The composition of seed mixes planted on the Dominion pipeline in Rothrock and Tuscarora State Forests. The 2008 general native grass and forb mix is indicated by $^{\Psi}$, the 2008 native-based steep slope mix by *, and the 2009 modified mix by ^, each followed by the percent of the total seed (kg/ha) of each mix.

Family	Species	Nativity Status	Life Cycle	Wetland Status
Poaceae	*Agrostis perennans* ($^{\Psi}$2%, *10%)	Native	Perennial	FACU
	Andropogon gerardii ($^{\Psi}$10%)	Native	Perennial	FACU
	Bouteloua curtipendula (^6%)	Native	Perennial	not classified
	Dactylis glomerata (^24%)	Non-native	Perennial	FACU
	Dichanthelium clandestinum ($^{\Psi}$10%)	Native	Perennial	FACW
	Elymus canadensis (*20%)	Native	Perennial	FACU
	Elymus riparius (^3%)	Native	Perennial	FACW
	Elymus virginicus ($^{\Psi}$15%)	Native	Perennial	FACW
	Lolium multiflorum (*20%, ^30%)	Non-native	Annual/Perennial	FACU-
	Panicum virgatum ($^{\Psi}$5%, ^5%)	Native	Perennial	FAC
	Schizachyrium scoparium ($^{\Psi}$20%, *20%)	Native	Perennial	FACU
	Sorghastrum nutans ($^{\Psi}$10%, ^6%)	Native	Perennial	UPL
	Sporobolus compositus (*10%)	Native	Perennial	not classified
Fabaceae	*Chamaecrista fasciculata* ($^{\Psi}$5%)	Native	Annual	FACU
	Desmodium canadense ($^{\Psi}$3%)	Native	Perennial	FAC
	Lotus corniculatus (^18%)	Non-native	Perennial	FACU
	Trifolium repens (^8%)	Non-native	Perennial	FACU
Asteraceae	*Heliopsis helianthoides* ($^{\Psi}$5%)	Native	Perennial	FACU
	Rudbeckia hirta (*5%)	Native	Annual/Perennial	FACU
	Solidago nemoralis (*2%)	Native	Perennial	not classified
	Symphyotrichum prenanthoides (*3%)	Native	Perennial	FAC
Asclepiadaceae	*Asclepias syriaca* ($^{\Psi}$5%)	Native	Perennial	UPL
Polygonaceae	*Persicaria pensylvanica* ($^{\Psi}$5%)	Native	Annual	FACW
Cyperaceae	*Scirpus cyperinus* ($^{\Psi}$2%)	Native	Perennial	OBL
Juncaceae	*Juncus effusus* ($^{\Psi}$3%)	Native	Perennial	OBL

Figure 4.1. Hypothesized structural equation model for factors contributing to seeding mix establishment and invasive abundance.

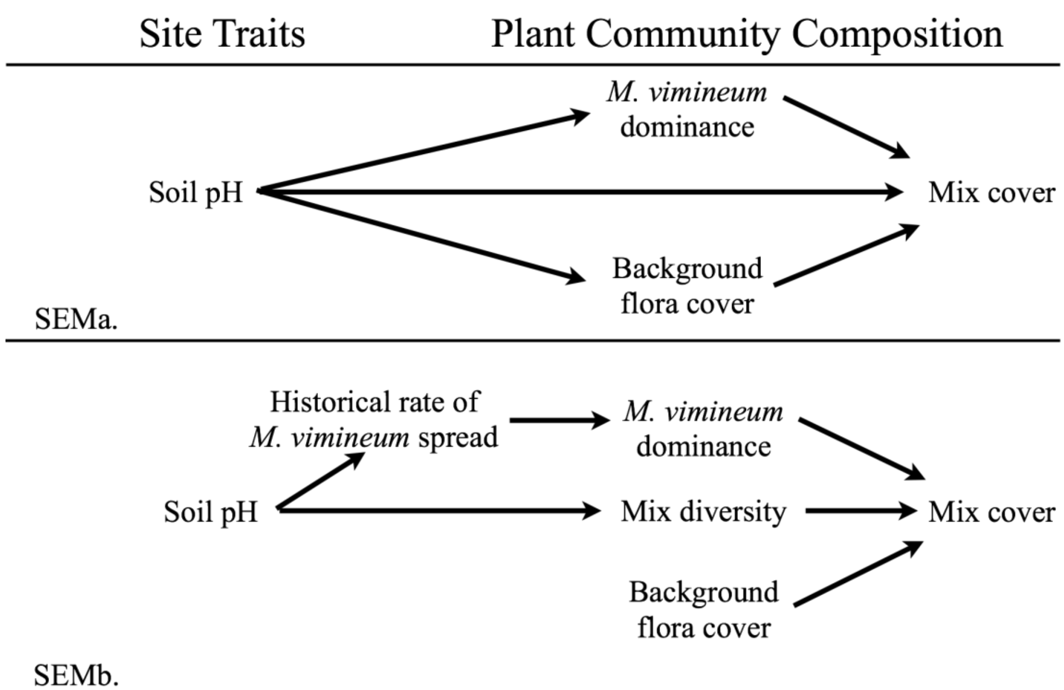


Figure 4.2. The 76 survey sites in Rothrock (left) and Tuscarora (right) State Forests categorized by *M. vimineum* density were located in the Ridge and Valley of central Pennsylvania, in the Northeast region of the United States. The map was created with QGIS (2009), using Google Terrain (Map data: Google, DigitalGlobe) basemap imagery and ESRI Satellite (2017) basemap imagery for the inset map of the Northeast US.

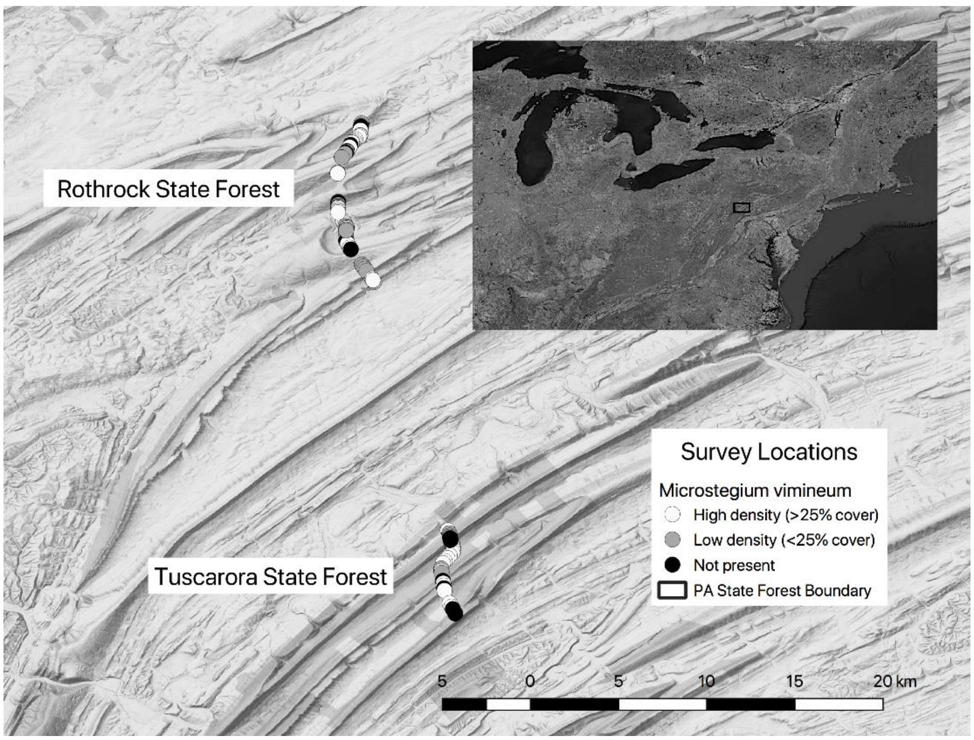


Figure 4.3: Frequency of seeded mix cover as a proportion of the total vegetation cover across survey sites. Sites on which *Microstegium vimineum* was dominant (>25% cover) are marked off in black. Sites where the mix species had a lower proportion of the total vegetative cover also had a greater proportion of *M. vimineum* dominance.

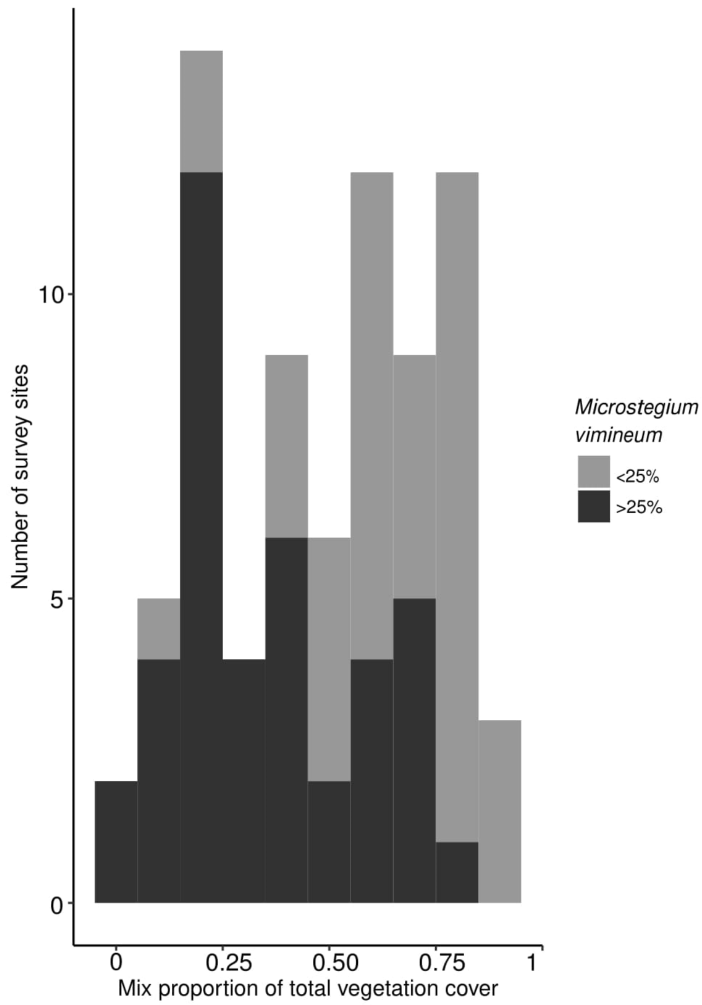


Figure 4.4: Seed mix species vegetative cover across the 76 pipeline survey locations on two Pennsylvania state forests.

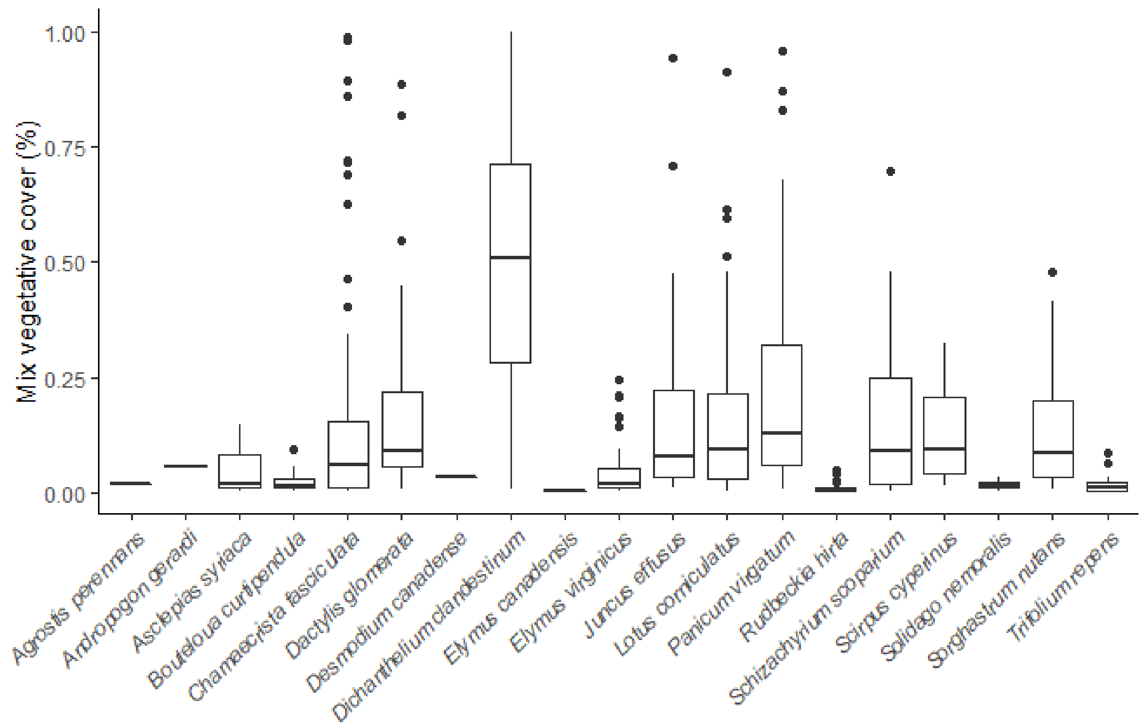


Figure 4.5: Vegetative cover of the ‘background flora’ or weeds across the 76 pipeline survey sites. Species that were present on 25% or more of the survey sites (19 sites) were included in this figure.

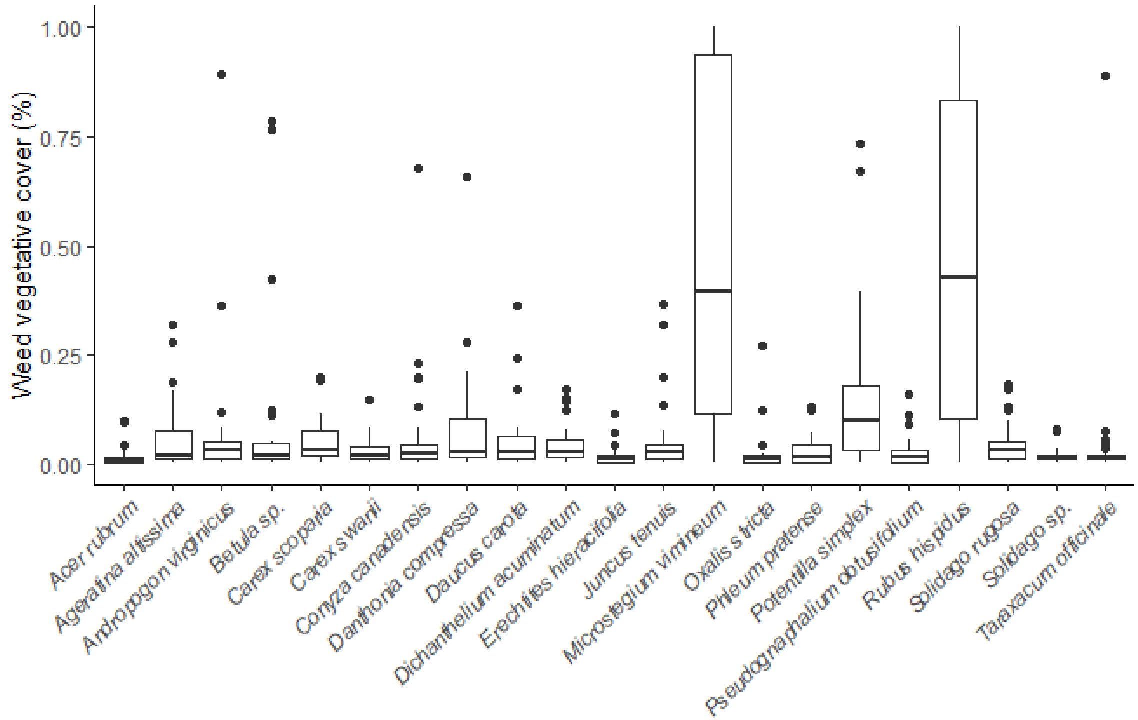


Figure 4.6: Frequency of mix species occurrence across the 76 pipeline survey location sites on two state forests.

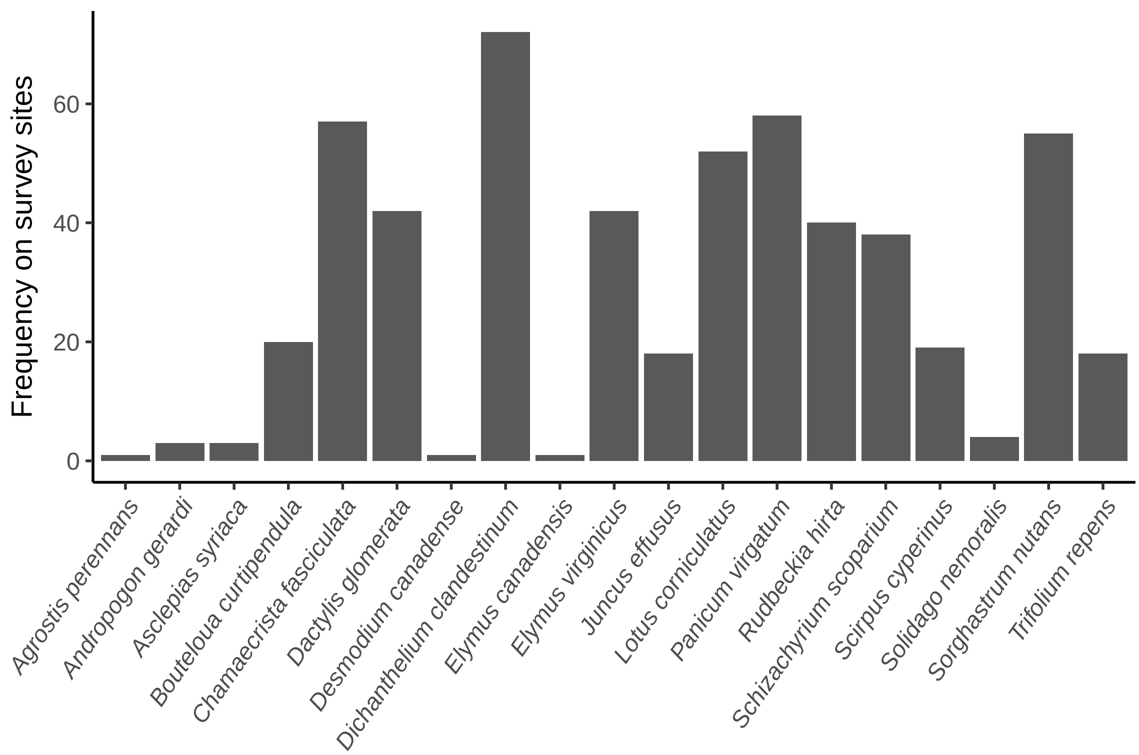
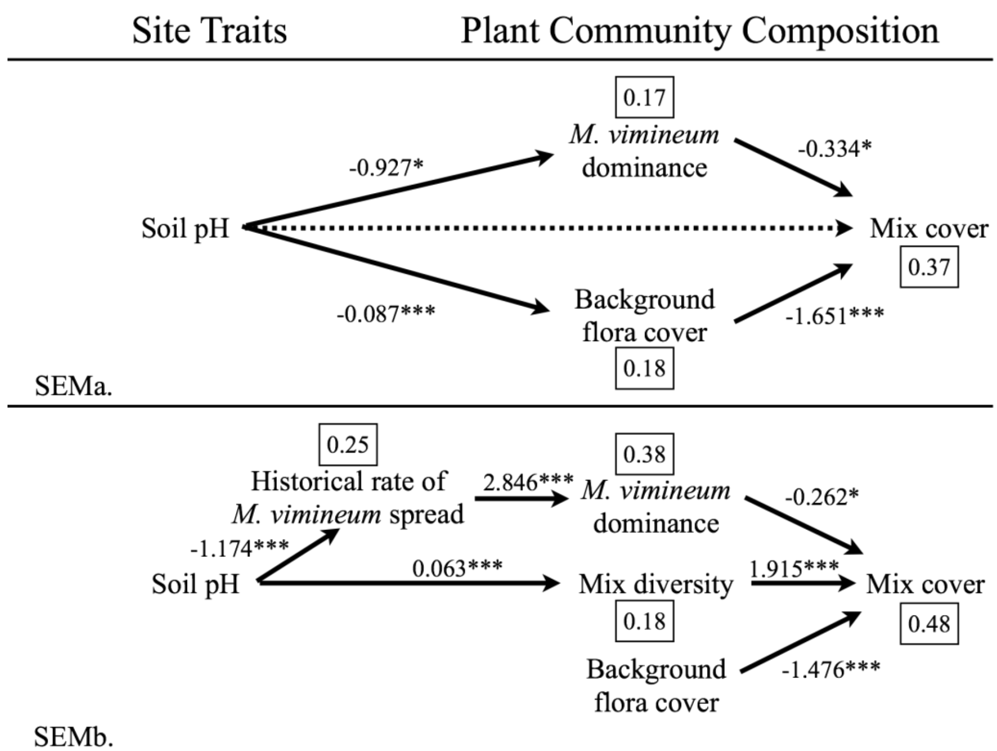


Figure 4.7. SEMa describes the influence of soil pH on plant community composition. SEMb clarifies the indirect influence of soil pH on mix cover. Dashed lines indicate insignificant causal paths in the model. The solid lines with unstandardized path coefficients are significant causal paths in the models, marked with the level of statistical significance (* = $p > 0.05$, ** = $p > 0.01$, *** = $p > 0.001$). Conditional r^2 values for each component model are in rectangles.



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Chapter 5

Critical Ecological Renovations needed for Reclamation Regulations

> *Earth disturbance activity*—A construction or other human activity which disturbs the surface of the land, including land clearing and grubbing, grading, excavations, embankments, land development, agricultural plowing or tilling, operation of animal heavy use areas, timber harvesting activities, road maintenance activities, oil and gas activities, well drilling, mineral extraction, and the moving, depositing, stockpiling, or storing of soil, rock or earth materials.
>
> PA Code § 102.1. Definitions.

Abstract

Reclamation regulations for earth disturbance activities lack an ecological foundation. The focus, and often central metric of successful reclamation, is a proxy measure of erosion and sedimentation control. Though necessary to prevent significant soil loss, these regulations do not convey the full value of terrestrial ecosystems. Terrestrial ecosystems support not only soil loss, but carbon sequestration, watershed integrity, and habitat. These limited regulations demonstrate the short-sightedness of human decision-making with natural resources. Unconventional oil and gas development is a case study in human decision-making and valuation of terrestrial ecosystems versus subsurface fossil fuels. We present the common themes in state regulations regarding reclamation of oil and gas and propose a new regulatory approach based in ecology. Updating regulations to support ecological function is critical to sustaining ecosystem services.

Humans are earth movers; excavators, agriculturalists, and restorationists. Through infrastructure, farming and logging humans have modified over 50% of terrestrial Earth (Hooke et al., 2012), and now cause ten times more soil erosion than all other natural processes (Wilkinson, 2005). The extent and locations of ‘earth disturbance’, and the subsequent nature of the repair of ecosystems, is revealing of human nature and values. As Earth’s “premier geomorphic agents” (Hooke, 2000) humans have demonstrated that they are driven to modify the earth for energy development (Trainor et al., 2016) prioritizing advancing technological innovations of energy production over the sustainable use of natural resources. The financial resources and attention devoted to technological advances for subterranean resource extraction is vastly greater than for surface reclamation. This notable difference in investment and attention reveals the short-sightedness of human societies to profit from fossil fuels without evaluating the long-term consequences. Here we call attention to, and investigate a critical driver of, this disparity: limited and outdated reclamation regulations. Drawing attention to the legal code that is the interface between the dismantling of natural ecosystems and the rebuilding of ecosystem function will help shed light on the mechanisms that maintain the status quo for extraction decision-making and keep restoration practices from advancing.

The shale oil and gas production boom that began in the mid-2000s in North America provides a critical case study on the disjunction between human modification of the environment to remove fossil fuels and the body of ecological knowledge for sustaining human societies. Ecological science has progressed much further than the human ability to use information and science to make informed decisions on resource extraction in terms of location and extent of development (Allred et al., 2015; Castro-Alvarez et al., 2018; Drohan et al., 2012), and common reclamation practices demonstrate (Sheoran et al., 2010; Zipper et al., 2011). This gap between current understanding of ecosystem function and environmental policy is a well-known global challenge (Fleishman et al., 2011; Palmer et al., 2004; Sutherland et al., 2011). In the 1990s scientists from multiple fields called for practices and policies to be updated with an ecological approach and based on current ecological findings; call came from the Ecological Society of America (Lubchenco et al., 1991), agroecology, the merging of ecology in agricultural systems (Gliessman, 1990), and natural resource management (Brown & MacLeod, 1996). Within the natural resource management discourse, Brown and MacLeod (1996) had argued that basing management and policy on the 'climax' theory of ecosystem succession (i.e. Clements) led to contentious and irrelevant management practices, and that, "models of how ecological systems function should serve as a basis for decision making to achieve some level of sustainability". We argue that the language used in reclamation management and policy for energy development must embody this approach for ecosystem function, address the complexities of site ecology, and reflect ecological principles.

Rare and endangered species are federally protected in the United States, but there is no protection for representative plant communities or ecosystems. Therefore, forest fragmentation (Donnelly et al., 2017; Drohan and Brittingham, 2012; Langlois et al., 2017) and significant forest biomass loss (Young et al., 2018) that occurs as a result of shale oil and gas development cannot be regulated. Without legal protection of ecosystems, further trophic losses, such as core forest birds (Barton et al., 2016), and native plant communities (Barlow et al., 2017) from shale gas infrastructure will continue. Less visible losses, such as the long-term decline of soil microbial biomass and diversity from mining (Quadros et al., 2016), will be left unaccounted for without ecological considerations. At the landscape scale, neglecting most components of ecosystem function could result in losses in ecosystem resilience, and trigger regime shifts (Pardini et al., 2010).

To address the above concerns within the restoration discourse and propose alternatives for instigating change for the shale gas era, we chose to focus in on key sections of reclamation regulations pertaining to reclamation plans and methods. We began with a thorough review and assessment of shale gas reclamation regulations. Then, to provide historical context that ties the regulation focus to erosion, we traced the language and concepts back to coal mining and the Dust Bowl, the origins of soil erosion centered regulations.

Reclamation regulations lack of foundation of ecological principles

> *Accelerated erosion*—The removal of the surface of the land through the combined action of human activities and the natural processes, at a rate greater than would occur because of the natural process alone.
> *BMPs—Best management practices*—Activities, facilities, measures, planning or procedures used to minimize accelerated erosion and sedimentation and manage stormwater to protect, maintain, reclaim, and restore the quality of waters and the existing and designated uses of waters within this Commonwealth before, during, and after earth disturbance activities.
>
> PA Code § 102.1. Definitions.

The shale oil and gas industry is required to meet federal statutes concerning the environment: National Environmental Policy Act (NEPA) (1970), Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA, also known as the Superfund Act) (1980), Clean Water Act (1972), Clean Air Act (1970/1990), and Endangered Species Act (1973). NEPA requires federal agencies to complete Environmental Impact Statements (EIS) to address the environmental impacts of development on federal land, but is only for federal lands and often lacks influence in decision-making (Jay et al., 2007). The other statutes point to impacts on off-site ecology or individual species. Notably, there is no federal statute to protect natural systems from the unique threats posed by hydraulic fracturing for shale oil and gas. Regulation development has been an unsystematic process, leading to variable standards by states within the same eco-regions. Standards from coal mining and conventional gas have been incorporated into these new regulations, which do not account for scientific progress made in restoration research and do not protect against critical sources of pollution unique to unconventional development (Holahan and Arnold, 2013). The challenges resulting from this unsystematic approach to policy, planning and research has been noted across multiple fields from social science, ecological economics, energy policy, ecological restoration and the intersection of these fields (Bazilian et al., 2014; Finkel et al., 2013; Goldthau, 2016; Pifer, 2011; Souther et al., 2014; Zirogiannis et al., 2016).

We carefully reviewed state regulations in central Appalachian forests pertaining to site restoration post 'earth disturbance' construction activities for shale oil and gas development. These regulations typically covered soils, plants, and how revegetation relates to wildlife habitat. We began with Pennsylvania, the center of the Marcellus and Utica formations, some of the most productive and active shale basins in North America. Zirogiannis et al.'s (2016) assessment of state regulations of shale oil and gas found that in the East, West Virginia and Pennsylvania were among the most stringent according to the language of the law, but in practice both the letter and the spirit can be lacking. All environmental laws in Pennsylvania[1] have a resource or risk focus, such as toxicology and sediment control in air and water systems. In Pennsylvania, as in many states in the region, the only regulation tying the unconventional gas and mining industry to a responsibility for the removed surface terrestrial systems is to replace them with vegetative cover to prevent soil erosion and sedimentation. This 2010 set of regulations, 25 PA Code § 102 "Erosion and Sediment Control and Stormwater Management", falls under the authority of the federal 1937 "Clean Streams Law" (P.L. 1987, No. 394) and the "Conservation District Law". The "Statement of Need" for 25 PA Code § 102 addresses the "ecological impacts" of sediment and other pollutants in water bodies as "physical impacts on water bodies and biological impacts on aquatic ecosystems", and the cascading effects of pollutants on human water resource needs. The emphasis is on the off-site impacts; the effects that occur when detrimental amounts of soil are carried from a site to water bodies downstream. The required actions – attain vegetative cover – required on-site do not acknowledge the on-site ecology. The "site stabilization" section gives the "earth disturbance" actors two options:

[1]www.elibrary.dep.state.pa.us/dsweb/View/Collection-8703

> (i) A minimum uniform 70% perennial vegetative cover, with a density capable of resisting accelerated erosion and sedimentation. (ii) An acceptable BMP [best management practice] which permanently minimizes accelerated erosion and sedimentation.

If the site was a forest, shrubland, or meadow, the “earth disturbance” actor need only seed a vegetative cover that putatively prevents soil loss. The actual on-site ecology has little legal protection and is given less value. The designation of “perennial” may seem desirable over establishing annual species but this specification complicates establishing some native perennial species that require an annual cover crop that will account for the majority of the ground cover in the first year. The responsibility to select ecologically-appropriate plant species, direct succession and to encourage long-term species diversity, or to guarantee any ecosystem functioning beyond minimizing erosion and sedimentation are not required or envisioned.

The general requirements for an erosion and sedimentation plan (25 PA Code § 102.4.) do require operators to describe the topography, soils, and riparian buffers while proceeding to give great leeway for accountability in terms of restoration of the geophysical and ecosystem components. Phrases such as, “to the extent practicable” (b.4.), and vague standards such as, “Maximize protection of existing drainage features and vegetation” and “Minimize soil compaction”, may give the land owner more options for end-use but also gives little protection to site ecologies that take anywhere from decades to centuries to replace.

Several states within the Marcellus and Utica formations use specific vegetation cover percentage requirements. West Virginia, like Pennsylvania, uses a 70% vegetative cover requirement for revegetation (WV DEP Section IV Revegetation), stating:

> The objective is to provide sufficient vegetation to control erosion and sedimentation on and off the site. A vegetative cover of 70% or greater would generally meet this requirement.

Neither state has a protocol specifying the spatial dimensions of 70% cover or how to visually determine cover. Visual cover estimates are not without complications, but do not lack in existing, tested, and accepted methods (Bergstedt et al., 2009; Fehmi, 2010; Kent, 2011). In informal conversations with Pennsylvania and West Virginia Department of Environmental Protection regulators, multiple landowners and land managers tasked with meeting this requirement, all stated that meeting this requirement was subjective and varied by the regulator.

The northernmost extent of the Marcellus and Utica formations reach into north central upstate New York. Although New York maintains a ban on unconventional oil and gas development, regulations for mining reclamations would likely be transferred to the unconventional industry should the ban be lifted. New York's Department of Environmental Conservation regulations for 'earth disturbance' activities (Chapter IV Part 422.3) are specific on soil depth ("minimum of six inches"), the plant composition of revegetation, and the spatial dimensions of what constitutes restoration failure ("areas of failure must be randomly distributed, shall not exceed one-half acre in every two acres"). The vegetation should be "indigenous to the area and where revegetation is consistent with the land-use objective", and have 75% coverage of non-woody vegetation. The regulation standards for New York contain much more detail than Pennsylvania, Ohio and West Virginia, but also give only a nod to ecological considerations.

The specified vegetation coverage requirement is intended to serve as an enforcement mechanism to measure success in preventing erosion and sedimentation from the disturbance on-site to off-site locations, such as streams. But, ground coverage as a percentage alone is a limited measure of success. Soil erosion research demonstrates that slope, soil type, vegetation type, plant growth form above and below ground, seasonality, and climate must be considered along with plant cover (Gutierrez and Hernandez, 1996; Zuazo & Pleguezuelo, 2008). Not surprisingly, recommended coverage varies widely in research studies given multiple site variables (Lang, 1979; Orr, 1970; Stewart & Forsling, 1931), just as it varies in state regulations. Zuazo and Pleguezuelo (2008) reviewed multiple studies assessing the relationship between plant cover and water runoff and found a high degree of variability.

Initially, on highly disturbed soils, seeding alone will not likely meet erosion and sedimentation goals (Wachal et al., 2008). Modeling erosion and sediment control practices with RUSLE 2.0[2] for Texas gas well sites, Wachal et al. (2008) found that mulching, erosion blankets, silt fences, filter strips, and sediment basins were all more efficient for immediate erosion prevention than seeding alone. A metric of plant cover alone is not a measure of successful restoration, and without incorporating other site factors, it is an essentially arbitrary number.

Ohio, Kentucky and Tennessee have no specific plant coverage percentage requirements. In Ohio, the focus of reclamation is also erosion and sedimentation under the section titled, "Duty to restore disturbed land surface" (Ohio Department of Natural Resources, Division of Oil & Gas Resources, Title XV, Chapter 1509.072). This statute states;

> (A) within six months after the date upon which the surface drilling of a well is commenced in all other areas, the owner or the owner's agent shall grade or terrace and plant, seed, or sod the area disturbed that is not required in production of the well where necessary to bind the soil and prevent substantial erosion and sedimentation.

Furthermore, the responsible actor may be released from this duty by part (B) of this section;

> The owner shall be released from responsibility to perform any or all restoration requirements of this section on any part or all of the area disturbed upon the filing of a request for a waiver with and obtaining the written approval of the chief, which request shall be signed by the surface owner to certify the approval of the surface owner of the release sought.

Ohio gas regulations lack any standards that recognize the value of the surface ecosystem directly degraded by the development activities.

In Kentucky, the reclamation regulations (Kentucky Administrative Regulations, Title 805 Energy and Environment Cabinet, Chapter 1 Division of Oil and Gas, section 170 Content of the

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[2]Erosion prediction model, Revised Universal Soil Loss Equation; www.ars.usda.gov/southeast-area/oxford-ms/national-sedimentation-laboratory/watershed-physical-processes-research/docs/revised-universal-soil-loss-equation-rusle-welcome-to-rusle-1-and-rusle-2/

operations and reclamation plan) state that the industry must create a reclamation plan that is agreed upon by the surface owner, and this plan must be completed for well closure. The details of reclamation are left to the discretion of these parties and regulations focus mainly on erosion and sedimentation procedures and methods. This statute cites the "best management practices" standards as listed in Kentucky Revised Statutes (Title XXVIII – Mines and Minerals, Chapter 353 Mineral Conservation and Development, .510 under Oil and Gas Conservation), which centers on erosion and sedimentation; "demonstrated practices intended to control site runoff and pollution of surface water and groundwater to prevent or reduce the pollution of waters of the Commonwealth". The outdated list of recommended herbaceous mixtures for revegetation contains mainly non-native cool season grasses and legumes that have long been used on compacted mine soils (Crews, 1984), but are known now to inhibit establishment of native, perennial grasslands (Bauman et al., 2015; Johnson and Sandercock, 2010), and forest succession (Skousen et al., 2009). For "site closure" the well owner is expected to have established a "diverse and effective permanent vegetative cover". However, the terms "diverse" and "effective" are not defined and therefore unenforceable. "Permanent" is also an undefined term and is not based in ecology, given that plant communities are dynamic and move through successional stages.

Tennessee's reclamation regulations are covered under Chapter 0400-52-09, "Well Plugging and Abandonment". Again, the term "permanent" is used to described the type of "vegetative cover" to be established within 90 days. In this case the language is more descriptive of what is "diverse, effective, and permanent" reclamation. The planting should "not impede natural vegetative cover", be "capable of long-term stabilization of the soil surface from erosion", have the "same seasonal characteristics of growth as the original vegetation", be "capable of self-regeneration", and "compatible with existing plant and animal species existing in the areas". The seeded vegetation must also follow state and federal laws on "poisonous", "noxious", or "introduced" plants. Acknowledging that the target plant and animal species should align with the on-site ecology is helpful to ecological integrity, but could still be improved by outlining how such components could be measured and enforced.

Western states with unconventional gas production have a similar focus on plant coverage for erosion control, and some include more detail on soil reclamation and vegetation types, for example, Colorado "encourages" establishing "species consistent with the adjacent plant community", and cover must be "at least 80% of pre-disturbance levels or reference areas, excluding noxious weeds" (Code of Colorado Regulations 404-1-1003-e). Basing a post disturbance vegetation coverage on "pre-disturbance" or "reference" conditions requires the land owner to have collected sufficient data prior or agreement on what is a reference condition for the site, both of which are unlikely to happen. New Mexico's regulations (New Mexico Statutes 19-15-17-13 for 'Closure and site reclamation requirements') offer confusing language on cover;

> uniform vegetative cover has been established that reflects a life-form ratio of plus or minus fifty percent (50%) of pre-disturbance levels and a total percent plant cover of at least seventy percent (70%) of pre-disturbance levels, excluding noxious weeds.

Requiring a "life-form ratio", or growth forms, that can vary by "plus or minus fifty percent" is a range that has little practical meaning if the pre-disturbance data was not collected. Further, plant communities are rarely "uniform". Again, the illogical and inconsistent vegetative cover requirements across states could make sense if it were based on ecoregional variation and required extensive pre-disturbance data collection. But from our reading there are no references in the regulations we reviewed that tie in landform, soil type, and climate variables to support these measurements.

In a 2014 report the US Congressional Research Service similarly points to the centrality of erosion and sedimentation control as the main concern for regulators, demonstrating the lack of an ecological perspective. This national level assessment cited the "potential direct risks" to the environment of the new exploration for shale oil and gas as "Water Quality Issues" and "Air Emissions" (Ratner and Tiemann, 2014), the direct losses of plant biomass and habitat are not considered. The U.S. Department of Energy review (DOE, 2014) of the environmental impacts of unconventional gas, in summarizing the analysis of Considine, et al. (2012) on notices of shale industry violations in Pennsylvania, further points to the limited value given to ecosystems and connectivity. Considine et al. (2012) reported two out of 25 "major" violations were from "site restoration failures". This review states (emphasis is mine);

> Site restoration events result when the operator does not restore a drilling site in accordance with PADEP guidelines, including removal of drilling equipment and waste and restoration of 70 percent of the perennial cover within nine months. Erosion was a problem cited in most NOVs [notices of violations]; in some cases, equipment was not removed or vegetation was not restored. *Land disturbances have an environmental impact, but they can be remediated with minor reclamation efforts* and *are not as serious as spills and water contamination*.

Erosion and toxic waste contamination *are* serious concerns for long-term ecosystem management (Burgos et al., 2017), but to suppose that attaining green cover is the mechanism for 'remediation' of land disturbance is to discount the complex role of vegetation as the foundation for ecological integrity.

We demonstrate that the central focus of 'earth disturbance' regulations, and therefore practice, is on the management of soil erosion. Significant soil erosion *is* a major concern for the construction activities of gas development (Wachal et al., 2008), and should be carefully managed. And, limiting soil erosion during construction activities is a necessary component to site stabilization and watershed quality (Harbor, 1999). However, most state regulations, and national reviews, for on-site management end with attaining sufficient vegetative ground cover for site stabilization and lack any further ecological guidance. The U.S. federal and state laws convey that as long as the development does not release toxins or harm critical habitat for endangered species, then the surface ecosystem is less valuable than the subterranean resources. When the value of the ecosystem is placed no higher than achieving sufficient ground cover to prevent erosion and sedimentation according to bond release regulations, it is logical that humans will continue to exploit subterranean resources at an unsustainable rate. Without feedback, change is unlikely to occur.

Without a federal and state legal "authority" that elevates the on-site loss of and impacts to plant communities, it will be difficult to move forward with improving reclamation practices for ecological integrity. The preservation of soils and clean water now have greater historical legitimacy in North American society's consciousness, and the regulatory language must reflect this. We hypothesize that this is likely due to the extreme, direct impacts to humans as experienced with the Dust Bowl, mountain top removal, acid rain, and pollution in the Chesapeake Bay. Identifying ways to direct society's attention on the stewardship value of ecosystems is imperative to any response to climate change.

The focus on managing soil erosion with appropriate ground cover is not unlike critiques of agricultural systems, such as stated by Robertson and Swinton (2005);

> This challenge requires an ecological approach to agriculture that is largely missing from current management and research portfolios. Crop and livestock production systems must be managed as ecosystems, with management decisions fully informed of environmental costs and benefits.

Establishing rapid ground cover for soil erosion is a vital concern for restoration, but our understanding of ecological functions has long surpassed simply limiting soil erosion. This nearly century-old focus must be updated with the latest ecological understandings that moves beyond green cover goals. Regulations to guide on-site restoration after "earth disturbance" must have an ecological framework and enforceable measures to support ecosystem-level benefits.

The historical momentum of green cover goals

The central Appalachians have a long history with natural resource extraction and energy production; timber, coal, and oil and gas. As we have shown in our review of shale oil and gas regulations the Central Appalachian states have largely sourced regulatory standards and language from coal mining reclamation regulations. As a result, the same focus on toxicology concerns in soils and water and a lack of attention to the direct loss of ecosystems (Wickham et al., 2013) have fed the momentum for green cover goals for the shale gas era.

The national scale concern for soil erosion stems farther back to the devastation caused by pervasive removal of mid-Western prairie vegetation and extensive tillage that caused the Dust Bowl of the 1930s. The Soil Conservation Service and the Soil Conservation and Domestic Allotment Act of 1936 were created in reaction to the devastation that become known as a "national menace"[3] throughout the discourse, this language even appears in the *Ecology* journal in 1935 (Weaver & Noll, 1935). Outside of agriculture in the 1930s and 1940s the emphasis for mining reclamation had been on restoring tree species, but often with little success, especially with hardwood species. The growing concern for stemming erosion from surface mining, particularly due to contour coal mining led to the research focus on erosion in the U.S. Forest Service in the 1950s (Plass, 2000), and the shift in planting herbaceous forages for rapid cover in the 1970s (Skousen et al., 2009).

Unlike oil and gas, coal mining is federally regulated under the 1977 Surface Mining Conservation and Reclamation Act, P.L. 1198, No. 418 (SMCRA), which designates States to enforce the Act given the environmental and terrain variability across coal country. The Act Introduction claims that (emphasis mine):

> surface mining and *reclamation technology are now developed* so that effective and reasonable regulation of surface coal mining operations by the States and by the Federal Government in accordance with the requirements of this chapter is an appropriate and necessary means *to minimize so far as practicable the adverse social, economic, and environmental effects of such mining operations.*

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[3] Hugh Bennett and William Chapline published their 1928 article, "Soil Erosion: A National Menace" as a warning of the impending disasters from agricultural practices in the mid-West that led to the Dust Bowl.

The "reclamation technology" – recountouring, compaction, aggressive cool-season grasses – has led to vast expanses across Appalachian coal country of degraded, non-native grasslands that must be deep ripped to relieve the soil compaction to reestablish forests (Angel et al., 2009). Recontouring methods, originally designed to address soil loss, created this severe soil compaction from the heavy equipment necessary for reconstruction (Thorne and Cardina, 2011).

SMCRA does ask operators to consider reclamation as more than controlling soil erosion. The Act stipulates that pre-disturbance land conditions must be stated in the reclamation plan, but land conditions are framed only in terms of "uses", particularly those uses with a "yield" (Chapter V, Section 508 §1258. Reclamation plan requirements). The "reclamation plan requirements" also must demonstrate "consideration" of how the plan aligns with the "local physical environmental, and climatological conditions", and "describe" the "quantity" and "quality" of water resources impacted. "Considerations" and "descriptions" may document what was lost, which is more than is asked for in most shale gas regulations, but does not hold operators accountable to those losses. Additionally, the phrase in SMACRA, "achieve an approved post-mining land use", has been used as a loophole to plant non-native forage species in former deciduous forests and has for example, left 80,000 ha of Ohio in a state of arrested succession (Thorne and Cardina, 2011).

Despite the attempts at placing value on surface ecosystems through "considerations" and "descriptions" the mining industry term for the terrestrial systems above fossil fuels, "overburden", detracts from helping elevate its value. Overburden, in SMCRA is "the strata or material overlying a mineral deposit or in between mineral deposits in its natural state and shall mean such material before or after its removal by surface mining". This term is also used in academic research for fossil fuel development and reclamation (Zipper et al., 2011). "Burden" refers to the literal weight of rock and sediment over the desired resource, and this word choice is telling of the contrast in value given by humans. Continuing to refer to the land that supports life as "overburden" is certainly not helpful to shifting the discourse and perceptions that factor into land-use decision-making. Using ecological terminology within the natural resource extraction discourse might help to change perceptions of value and stewardship.

From 'Ecosystem Services' to Stewardship

The dominant language in a discourse influences how society perceives the discourse topic, consciously or unconsciously (Young and Fitzgerald, 2006). The general perception feeds into practice and policy (Feindt and Oels, 2005). We need to upend the discourse for reclamation to perceive the value of surface ecosystems above natural sources of energy to change the manner in which humans interact with the environment. The predominant global language used in science and policy to convey the need for humans to value multiple ecosystem functions provided by nature for human benefit is 'ecosystem services'. The use of this conceptual framework has gained considerable momentum since the early 2000s (Cornell, 2011), solidifying at the global scale with the Millennium Ecosystem Assessment in 2005 (cite?). This language that is used to convey concepts of ecology spans scientific fields from agriculture (Zhang et al., 2007) to forest restoration (Zipper et al., 2011), with more recent calls to social scientists to join the conversation to address 'issues of poverty, justice, commodification, governance, ethic, rights, biodiversity and social–environmental relationships' (Chaudhary et al., 2015). Scientists, practitioners, and policy-makers aim to socialize the application of ecological science under the premise that human well-being is linked to biodiversity (Soliveres et al., 2016) and a diverse set of ecosystem services (Bennett et al., 2015).

One way scientists have angled to convince society of the value of nature has been by demonstrating the potential to maximize ecosystem services in a landscape with 'service' trade-offs, such as in Foley et al. (2005). This type of visual tool can be a useful means to convince society to value natural ecosystems when making land use decisions and reclaiming them after a disturbance. An extreme case in point, Liu et al. (2010) calculated nature's value across the entire U.S. state of New Jersey to demonstrate that human societies do not have the financial capital to replace all the ‘services’ that nature provides. Applicable to our case study, Allred et al. (2015) have estimated that “oil and gas development from 2000 to 2012 reduced NPP by ~4.5 Tg of carbon” in Central North America, a critical ecosystem service in the age of climate change.

If reclamation regulations could convey the current and future ‘economic’ value of forest ecosystems, such as carbon stocks, air purification, and watershed maintenance, maybe decision-makers would be able to more accurately weigh the benefits subsurface resource extraction. But, often the ecosystem services approach can only help to alert society to the need for natural ecosystems, not bring systems change. When applied to a framework for decision-making in the context of a ‘loss of a capital asset’ as set out in the Millennium Ecosystem Assessment, ecosystem services concepts can do a disservice to sustaining natural systems (M. Robertson, 2012). Applying this framework can be problematic because it commonly lacks paths connecting nature’s service capitalist-based ‘values’ with critical sustainability goals (Schröter et al., 2017). To contribute to sustainability goals, human well-being (the ecosystem services’ target) must be more broadly defined with a long-term perspective (Raudsepp-Hearne et al., 2010), and clearer alignment must be made with implementation (Lautenbach et al., 2015). Convincing human societies to perceive their survival as it depends on functioning natural systems should not be limited to arguments of consumption. Similar to green cover goals, this is a band-aid solution to a problem of ecological blindness.

If human societies do not reframe reclamation regulation language on the basis of ecology, we will continue to devalue the natural ecosystems whose long-term value in sustaining life is less apparent than the short-term energy gain. Profit-driven ‘earth disturbance’ actors will not be convinced to act from a stewardship perspective, especially if not held to an ecological standard. If reframing in terms of ecology does not occur, human societies will continue to be left with extensive unproductive and degraded land areas which require significant philanthropic impact investment to revive a functioning ecosystem. Ecologically informed decision-making will require, among many societal shifts, a transformation in our regulatory system for reclamation. Using "economic metaphors", like ecosystem services and natural capital, "marginalizes transformative agendas" and new metaphors, frameworks and language are necessary to provoke effective change in how we integrate sustainably with our natural surroundings (Coffey, 2016).

In Table 5.1 we provide some examples of 'Earth Disturbance' regulatory language that does not align with ecological principles. To illuminate the need for change we point to what this language implies and propose alternative language with implied meanings that direct the listener (public constituents) and the decision-makers (land managers and policy creators) to think from an ecological perspective.

Table 5.1. Examples of ecologically-based language for regulations governing natural resource extraction.

Resource Extraction causing Ecosystem Disturbance - COAL MINING			
Current Regulatory Language Examples from Surface Mining Control and Reclamation Act (1977)	*Implied Meaning*	*Proposed Regulatory Language*	*Implied Meaning*
Overburden Term commonly used as follows, emphasis mine: "…using all available <u>overburden and other spoil and waste materials</u> to attain the lowest practicable grade" *Chapter V, §1265*	Soil and vegetation ecosystems are a useless weight and waste that must be removed for the more valuable coal	**Overlying terrestrial ecosystems**	Soil and vegetation ecosystems are recognized as ecosystems with value
Pre-disturbance statement: **Condition of the land** **Capability of the land** **Productivity of the land** *Chapter V, §1258.*	The value of the land is in its ability to produce yields. Land with rare plant and animal communities but low fiber or timber yields are not accounted for.	**Pre-disturbance statement:** **Connectivity value within the natural landscape** **Represented diversity within the regional landscape**	Land that is a lynchpin for connectivity, land that represents critical species, habitat diversity or representiveness in the broader landscape will be recognized as such prior to a decision for disturbance.
Resource Extraction causing Ecosystem Disturbance – OIL & GAS EXTRACTION			
Current Regulatory Language Examples from PA Code § 102.1.	*Implied Meaning*	*Proposed Regulatory Language*	*Implied Meaning*
Earth disturbance activity *A construction or other human activity which disturbs the surface of the land…the moving, depositing, stockpiling, or storing of soil, rock or earth materials.*	Earth material moved out of place can be remedied by putting the components back together	**Human activities which degrade or destroy ecosystem functions**	Humans are the cause of the degradation or loss of ecosystem functions when they disturb an ecosystem

Site stabilization	The land must be secured from losing soil	**Facilitate site function**	The land is composed of dynamic processes, organisms adapting, evolving, and interacting
A minimum 70% perennial vegetative cover, with a density capable of resisting accelerated erosion and sedimentation	Perceives reclamation as achieving sufficient vegetative ground cover as the end goal	**Establish ecologically suitable plant communities, along with any necessary erosion control materials, that will set the successional trajectory to functional habitat**	Plant communities are dynamic and provide the foundation for habitat. Initial soil erosion control measures beyond vegetation may be required.
Diverse and effective permanent vegetative cover	Plant communities, once properly established, will not change		

Conclusions and Recommendations

Changing societal values - which translate to action or inaction affecting sustainability options for human life - will require multiple and diverse efforts to create a sea change over time. Here we call attention to two changes that could help to add momentum to necessary change towards sustainable interactions with the planet. First, regulations for reclamation must be approached from an ecosystem perspective founded on ecological principles. Regulations generally only focus on on-site erosion and off-site sedimentation, which are critical to site stabilization and controlling point-source pollution after an ‘earth disturbance’, but must be designed to contribute to broader site ecological goals. Second, to help orient human value of terrestrial ecosystems we recommend changes in the legal discourse that accurately label disturbance activities as ecological functions.

Despite the wealth of scientific evidence on the value of surface ecosystems to human thriving, and because of the extractive industry’s perceived limited environmental costs, decisions will continue to be made at the expense of the natural landscape. The extractive industries, without exceptions and loopholes, must be required by law to restore a diverse, multi-functioning ecosystem on the footprint of the disturbance. The language of the law must convey the value of surface ecologies for practices to change at an ecologically meaningful scale. The vast attention given to the advancement of gas production, in financial investments, research, human labor, demonstrates significant human ingenuity and dedication. Given the uncertainty of ecosystem changes due to climate change, it is high time to direct some of that ingenuity to the protection of terrestrial ecosystems.

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Chapter 6

Epilogue

Ecological lessons learned from plant community dynamics

Investigating a challenge from multiple angles, and dwelling on the results for several years, has ingrained within me the resolve to not seek singular answers to ecological questions. When I began my studies I thought I might seek a way to suppress the invasive plant *Microstegium vimineum* by using plant species with functionally competitive traits. I thought that with the right mix design of functionally competitive species I might find *the* native plant mix that would suppress *M. vimineum* in northeastern forests and fields. What we found from observing and experimenting with a similar set of plant species in different landscapes was that species specific 'resistance' to an invasive, and plant assemblage outcomes, differed in different soil environments.

In our experimental work *M. vimineum* growth was the least suppressed by *Dichanthelium clandestinum* (Figure 3.10), the species we had hypothesized would suppress *M. vimineum* with functionally competitive traits. Both species have decumbent growth forms, and we expected that the typically dense ground cover of *D. clandestinum* would significantly reduce resources for *M. vimineum*, as was seen by Flory et al. (2007). In contrast, we expected that the tall bunch grasses present in the study plots, *Panicum virgatum*, *Andropogon gerardii*, and *Sorghastrum nutans*, would, comparatively, allow more light through and therefore result in greater *M. vimineum* biomass. The slow growth by *D. clandestinum* in the first three years of our experimental study was surprising. We are not clear on why this species performed so poorly on this site, it is possible that because the soil was not infertile, droughty, or acidic that the background flora and other seeded species were better competitors than *D. clandestinum*.

Dichanthelium clandestinum is typically a dominant perennial grass when growing in infertile, acidic soils. This was the case in our observational study where *D. clandestinum* was one of four, along with *M. vimineum*, most frequent species on sites with a soil pH less than 5. *Dichanthelium clandestinum* also had by far the highest average cover across pipeline survey transects (Figure 4.4). In these soil environments *D. clandestinum* was one of the few that did compete with *M. vimineum*. Given our initial hypotheses and the study designs we cannot confirm the contributing factors for the outcomes we observed, but we hypothesize that from what we found in both studies that when soil conditions are poor *D. clandestinum* is a better competitor with *M. vimineum*.

Nuanced decision-making, that considers multiple contributing factors to community assembly, must be the approach for ecological restoration. Long-term, when continued interventions are necessary to sustain a desired trajectory, land management must be just as nuanced. Plant competition is dynamic over time, so as resource availability changes different mechanisms of competition will determine assemblage (Mangla et al., 2011). We do not know which species of the restoration mix seeded on the gas pipeline were initially recruited in 2008. In 2016 we observed only one small stand of *A. gerardii* along the length of the pipeline corridor crossing two state forests. If we were to provide recommendations based on the results of the experimental study we would have said that *A. gerardii* was the most effective competitor with *M. vimineum* and this species, or one with similar traits, should be included in mix design. But this experimental study included data and observations from only the first three years of mix establishment. Perhaps *A. gerardii* was initially competitive on the pipeline corridor, but by 2016 other species had become the dominant competitors. Assessing mix species dynamics over time across environmental gradients will be critical to making management recommendations.

I have learned to understand these ecological rules as models – generalizations/averages of the processes and entanglements of matter – that are influenced by the observer (Barad, 2007), and change with different environmental encounters in our heterogenous planet (Török et al., 2011). I have much gratitude for this process of expansive learning and embodiment of ecological thought and it has well-positioned me for my career in applied conservation ecology.

Ecological lessons learned from rhetoric

Learning how to investigate the way language is used, consciously and unconsciously, has been an incredibly useful tool to experiment with over the time spent working on my dissertation research. I have been inspired to play with upending the common words I write and speak to transform the embedded thinking laced up in my language. I have watched my own thinking, choices and actions morph when I dig up the unconscious perceptions I had with words such as, 'invasive', 'plant', 'ecology'. Words, with their definitions, provide the necessary structure and boundaries that make discussion possible. But when the structure only permits a particular thought pattern we might miss out on the ecological connections not contained within those boundary lines. Is the 'invasive' always a negative impact to the ecosystem (Bartomeus et al., 2016; Byers et al., 2014; Miller and Bestelmeyer, 2016)? What is the spatial extent of spread that matters to the region? What timescale of impact matters for the species in a community? Scientists may have a more nuanced approach when studying 'invasives', but this nuance is not typically communicated to land managers. This research-management line of communication matters. Working with conservation land managers in the public and private sector across Appalachia I have seen that this structure around 'invasive plant management' drives a fragmented approach that seeks singular answers. Management actions with species that drive changes in ecological thresholds across a broad landscape may be an appropriate tactic when the ecological outcomes can be identified for decision-making. With a rapidly changing climate and rising human populations continuing to modify their environments the need to restore and sustain ecosystems from the perspective of ecological function, rather than a singular species approach, becomes paramount.

The restoration ecology discourse is another arena that must be examined to more effectively contribute to sustaining ecosystem functions for the future. While working on the empirical sections of my dissertation I was dismayed by the common practices to reclaim land transformed for energy production. Why are human societies not capable of applying knowledge derived from scientific evidence to on? Forests converted to low diversity, non-native, cool season grasslands for pipeline corridors and mountains crushed and left in piles to regenerate into shrublands, is the measure of "successful restoration" according to our legal structures. Over several years I attended workshops, conferences, met land managers in the field to ask as many individuals involved in the process of reclamation what they perceived were the barriers to changing practices to align with ecological principles. Unsurprisingly, the most common response was the lack of finances. Assessing the power structures that govern land-use decision-making, which industries have the power and finances to use the land for particular resources, is certainly well studied and documented and not likely to change soon. Until the overarching structural change happens we must find ways to address the disparity of attention and resources given to restoration. I chose to investigate the machinery of reclamation regulations that drive on-the-ground practices. We outline the limited scope given in the regulations governing 'earth disturbance' – erosion control, and the emphasis towards off-site impacts and a lack value given to the loss of on-site ecosystems. I have become even more convinced of the urgency of change since starting to work with an environmental non-profit. Early on I was tasked with identifying ways to move forward with the challenge of restoring the 300,000 acres of degraded mine lands in Central Appalachia. All the science others and I have done is no help when you cannot convince a landowner to spend $1000 per acre to reclaim degraded land when it is only worth $300 an acre. The impetus for changing land-use decisions must come prior to the disturbance. If the oil and gas industry is not made accountable to restore at the onset of production the region will most certainly be left with vast expanses of degraded land.

Currently reclamation regulations perceive only ‘green’: ground cover and money. Ecosystems are more complex than described in regulations and in the broader land management discourse. Alongside advancing restoration science and sharing results from empirical studies we must continue to find ways to tell ecosystem stories to help society perceive the components of ecosystems as other organisms do - the multi-dimensional structures as homes, the wafting chemical compounds as signals, the nectars as food, and the solar-powered energy-producers that cycle water and nutrients.

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