

Understanding Invasive Species Impacts on Reclaimed Surface-Mined Lands

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ABSTRACT

The Appalachian region of the US is home to the most extensive temperate deciduous forest in the world, which provides many ecosystem services and economic benefits. However, Appalachian forests have undergone significant disturbance due to surface mining, resulting in fragmentation and net loss of productive forests. Invasive species are common on these severely disturbed landscapes, challenging efforts to restore native forests as advocated by the Forestry Reclamation Approach. Autumn olive (*Elaeagnus umbellata*), a nitrogen-fixing exotic shrub, is one of the most prevalent invasive species in this region and is viewed as a major deterrent of reclamation success. We review the literature on the impacts of exotic species on restoration, as well as advocating for continued progress in our ecological understanding of plant function in reclaimed ecosystems. In addition, we characterized the following at an active surface mine: 1) the effect of substrate and reclamation vegetation on autumn olive establishment and performance, and 2) the effect of autumn olive management on native hardwood growth. We found that substrate and vegetation mixes significantly affected autumn olive growth, but we did not see any significant differences with autumn olive management's affect on hardwood growth at the end of the first growing season, indicating no impact of autumn olive. The continuation of these experiments will further our understanding of invasive species impacts on forest succession of reclaimed mine sites.

Public Abstract

Mining has caused ecosystem losses worldwide, with surface mining disturbing >2.4 million hectares in the United States since 1930. The Appalachian region of the US is home to extensive temperate deciduous forests that provide many ecosystem services and economic benefits. However, >400,000 hectares of forest have been lost due to surface coal mining, with most not being restored back to native forests or other productive land uses. These areas are left fragmented, heavily modified, unmanaged, and densely invaded by non-native plants. Autumn olive (*Elaeagnus umbellata*) is one of the most prevalent invasive species on reclaimed mines in Appalachia and viewed as one of the main hindrances to the successful reclamation of mined land to restore native forests. In order to better assess the impact autumn olive can have on reclamation success, we characterize autumn olive's performance in various reclamation scenarios and also how the management of autumn olive affects hardwood tree establishment. We review how exotic species impact restoration outcomes, and advocate for a better understanding of how these species could contribute towards a more ecological understanding of reclamation. Reclamation goals are currently assessed after 5 years, prioritizing short-term goals (e.g. erosion control) instead of longer-term goals such as the return of ecosystem function. With a better understanding of plant function and ecological processes, we hope to continue to advance successful reclamation on surface mined lands.

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Table of Contents

Acknowledgements.....	iv
Chapter 1: Exotic perspectives on reclamation: prioritizing functional species.....	1
Abstract	1
Land degradation concomitant with population growth and economic advancement.....	2
Restoration and Reclamation	3
Opposing perspectives of exotic species in restoring disturbed systems.....	5
Invasive plants: Disturbance opportunists	10
Reclamation and Exotic Species	13
Plant function in novel ecosystems - opportunities and challenges	16
An exotic perspective - taking a functional focus	18
Case study: Autumn olive on reclaimed mines in Appalachia	20
Literature Cited.....	26
Chapter 2: Autumn olive performance varies in different reclamation conditions.	34
Abstract	34
Introduction	35
Materials and Methods	39
Results.....	42
Discussion	43
Literature Cited.....	47
Figures	55
Chapter 3: Management of autumn olive and the effects on hardwood tree establishment	61
Abstract	61
Introduction	62
Materials and Methods	65
Results.....	67
Discussion	68
Literature Cited.....	72
Figures	80
Chapter 4: Summary	84

Chapter 1

Exotic perspectives on reclamation: prioritizing functional species

Abstract

With the continuous rise in the human population, the demand for food and natural resources will only increase, leading to further anthropogenic land use and degradation. Surface mining alone has disturbed >2.4 million hectares of land since 1930, the majority of which has not been restored to forests in Appalachia, but left unmanaged and unproductive. Currently, reclamation goals are assessed after 5 years, forcing a focus on short-term goals (e.g. erosion control) versus more long-term ecological goals (e.g. return of ecosystem function). Invasive species are common on these severely disturbed landscape, and present a variety of challenges when trying to return the landscape back to native hardwoods as advocated by the Forestry Reclamation Approach. While some nonnative species, such as ryegrass and birdsfoot trefoil are intentionally planted, other species such as autumn olive are condemned and deemed undesirable, though the focus has been on nativity and not ecological function. We review ways in which exotic species have played a role in restoration generally, and advocate for a better understanding of how these species could play a more ecological role in reclamation. We look at how a greater understanding of plant function could foster successful reclamation of ecological processes in novel ecosystems such as surface mined lands.

Land degradation concomitant with population growth and economic advancement

The world's human population is currently >7 billion people, and is predicted to exceed 9 billion by 2050 (United Nations Department of Economic and Social Affairs, 2013), which will accelerate the demand for more natural resources and agriculture globally. For example, the demand for grain alone is expected to double by 2050 (Alexandratos 1999; Cassman 1999; Fedoroff & Cohent 1999), requiring more land to be cleared, and the lands already in production will need to undergo more intense practices to achieve higher yields (Tilman *et al.* 2011). Throughout most of history there has been development of land for agriculture followed by a period of overexploitation (Saunders *et al.* 1991), which has resulted in negative situations such as regions in North Africa experiencing desert expansion (Le Houerou & Gillet 1986; Ehrlich & Ehrlich 1987) and the United States Dust Bowl (Hudson 1981).

While agriculture is one of the leading causes of land degradation, it is only one of many ways which natural land areas undergo extensive disturbance and experience a loss of native biodiversity. In fact, there are practically no areas left on Earth that have not been subject to human modification or influence in some form (Vitousek *et al.* 1997). For example, anthropogenic land modification can result in natural fire cycle disruption via suppression or acceleration (Rockström *et al.* 2009), nutrient loss or over accumulation (Hautier *et al.* 2014), species extinctions (Barnosky *et al.* 2011), habitat fragmentation (Saunders *et al.* 1991), and the addition of greenhouse gasses to the atmosphere, which contribute to climate change (Tilman *et al.* 2001). The extraction and production of fossil-based energy has also been found to be a large component of land

use change and often competes for land with food production or development (Dilly *et al.* 2010).

Mining and its associated activities have led to the loss and destruction of established ecosystems worldwide (Dilly *et al.* 2010). Surface mining, specifically, is considered one of the largest types of disturbances in the United States with >2.4 million hectares mined since 1930 (Zeleznik & Skousen 1996). In the Appalachian region, surface coal mining alone has affected >1.1 million ha (Bernhardt & Palmer 2011), with the majority of those lands being previously dominated by forests. Most of those lands have not been restored to productive forest or other economically viable land uses (Zipper *et al.* 2011a), resulting in heavily modified and poorly managed landscapes invaded by non-native plants (Burger *et al.* 2013). This pattern of severe disturbance is likely to continue as ever-more natural resources are required to meet the needs of our growing population – necessitating a renewed focus on pathways to achieve restored ecosystems.

Restoration and Reclamation

The National Research Council (1992) defined restoration as returning and managing disturbed landscapes back to some predefined historical state. The primary goal of restoring disturbed areas is to return the degraded ecosystem into a condition that comprises more desirable ecosystem functions, species composition, or community structure (Noss 1990). Returning a system to a predisturbance state can include altering physical processes, topography, soil composition, hydrology, and the resident plant community, but the implementation of these activities varies depending on the disturbance.

Due to the destructive and common disturbance globally as a result of mining, there is an entire subset of restoration that focuses exclusively on stabilizing and revegetating mined lands for some viable use in the future, called reclamation, instead of reestablishing the previous ecosystem. Motivated by the desire to mitigate the severe erosion, sedimentation, and landslides that commonly follow surface mining, reclamation practices became mandated and standardized specifically for coal mining in 1977 with the Surface Mining Control and Reclamation Act (SMCRA). In addition to returning original contours, SMCRA requires mined land be restored to its original use (e.g. forest) or a use that will yield a higher economic value. To ensure compliance, the mining operator has to obtain a mining permit and pay a bond to the federal government prior to resource extraction to ensure reclamation compliance. The most direct way to meet these requirements in Appalachia is to restore the native hardwood forest, and in some instances pasture (Holl, Zipper & Burger 2009).

However, because the assessment of successful reclamation is typically evaluated only five years after reclamation efforts are begun (Zipper *et al.* 2011b), many reclamation projects focus on the short-term goals of providing erosion control and minimizing acid mine drainage, and not on more long-term goals of community structure and ecosystem function and service (Lemke *et al.* 2013). Many of the parameters that determine successful reclamation, and thus bond release, are typically fairly simple to quantify (e.g., percentage vegetated ground cover). However, these shortsighted and short-term goals that dictate successful reclamation under SMCRA do not attempt to return the land to an approximate pre-mining functioning ecosystem, and focus on future viable land uses instead.

Outside the structural restoration of a landscape (e.g., topography, hydrology), restoring plant communities is a primary focus of most restoration and reclamation projects. The plant community directly and indirectly drives many ecosystem processes, and has been shown to have many benefits, including increasing site biodiversity, improving hydrology and the control of sedimentation and nutrients, as well as detoxifying areas that have been polluted (Edwards and Abivardi 1997). However, reestablishing the native (or desired) plant community, presumably to some pre-disturbance state, can be extremely challenging depending on species availability and site conditions. Additionally, the inherent disturbed nature of restoration and reclamation sites often favors opportunistic and more disturbance tolerant invasive plants (Hobbs & Huenneke 1992; Lozon & MacIsaac 1997; Mack & Antonio 1998).

Opposing perspectives of exotic species in restoring disturbed systems

Humans have intentionally, and unintentionally, moved plants across the globe to meet our food, fiber, fuel, and aesthetic needs (Levine *et al.* 2003). The rate at which species are being introduced into areas outside of their native range continues to grow rapidly, and is predicted to rise with ongoing climate change and global commerce (Bradley *et al.* 2012). This is increasingly important as invasive plants have negative impacts to native systems from the gene to the landscape level that can threaten life-sustaining ecosystem function (Ehrenfeld 2003). Due to their widespread impacts, which are emerging as systemic, extensive, and expanding (Levine *et al.* 2003; Pysek *et al.* 2012; Kumschick *et al.* 2015), non-native species are considered to be one of the five major elements of global change, and responsible for the loss of biodiversity worldwide (Mack *et al.* 2000; Kolar & Lodge 2001).

In general, there are three main factors that determine whether an invasion will actually occur in an ecosystem: propagule pressure, the traits that the exotic species has, and the receiving habitat's susceptibility to invasion by a new species (Lonsdale 1999; Barney & Whitlow 2008). Arguably the most common attribute associated with invasions is disturbance to the receiving habitat has been shown to facilitate the establishment of invasive species (Hobbs & Huenneke 1992; Lozon & MacIsaac 1997; D'Antonio *et al.* 1999). Disturbance can increase resource availability, and supports the fluctuating resource hypothesis put forth by Davis *et al.* (2000) in which "a plant community becomes more susceptible to invasion whenever there is an increase in the amount of unused resources."

This increase in resource availability occurs as a result of either a reduction in the demand from the standing community, or due to the particular resource increasing quicker than the native vegetation is able to utilize it, leaving excess resources available (Davis, Grime & Thompson 2000). However, disturbance is likely the most common factor that contributes to an increase in resources by decreasing or eliminating the native vegetation (Hobbs 1989; D'Antonio 1993), and thus increasing the likelihood of invasion (Davis, Grime & Thompson 2000). Even seemingly small alterations to the native ecosystem can make the entire habitat more susceptible to invasions. Thus, the harsh growing conditions, lack of intact native community, and historical planting of exotic plants has led to these landscapes commonly being dominated by invasive species.

Land managers employ a variety of tactics to achieve the restoration goals of returning an ecosystem as closely as possible back to its previous state. Invasive species are commonly problematic when trying to realize restoration goals due to their ability to

colonize disturbed areas, which can lead to direct competition with the native species or alteration of the ecosystem. Berger (1993) suggests that ecological restoration can actually be used to eradicate and control invasive species, so any restoration method that affects soil conditions, light availability, water and hydrological processes, or temperature could serve to discourage invasive species while maintaining the natural ecosystem and its processes (Berger 1993).

The National Park Service (1997) stated that 194 out of 368 of their parks face serious threats from exotic species, the management or eradication of which can be extremely costly and time consuming. For example, *Lonicera maackii* (Amur honeysuckle), native to northeastern Asia, has invaded much of North America (Luken 1988) and is a very aggressive colonizer and competitor (Luken & Mattimiro 1991). When trying to eradicate *L. maackii* from the understory of a temperate forest, there is often substantial resprouting from the basal stems, and the disturbance from the removal also presented additional space for the establishment of honeysuckle seeds present in the seedbank. In this instance, mechanical control by itself is not enough to manage *L. maackii*, and a multi-faceted approach is needed, which includes replanting native species (Luken & Mattimiro 1991). Thus, preventing new invasions is paramount to effective landscape management.

In more severely disturbed landscapes, however, invasive plants are often the first species to colonize, even if they weren't previously present in the pre-disturbed habitat. This early invasion can create a variety of challenges when trying to restore native communities. In California, controlled burning has been used as a method of restoration at the Vandenberg Air Force Base in order to encourage endemic species in the maritime

chaparral to regenerate (D'Antonio & Meyerson 2002). However, these restoration areas are also experiencing an invasion of a South African succulent, the Hottentot fig (*Carpobrotus edulis*), which regenerates quickly after fire (D'Antonio 1993) and has become very abundant throughout the burned restoration sites (Hickson 1988). *C. edulis* is very competitive with the native maritime chaparral species (D'Antonio & Mahall 1991), and there are large numbers of *C. edulis* seeds in the soil. The fires do not reach temperatures hot enough to kill *C. edulis* seeds and result in more desirable soil conditions for them to germinate and proliferate (D'Antonio 1993). Not only does the fact that they have been able to build up an extensive seedbank help their proliferation, but so do the deer which transport their seeds to other burned areas (D'Antonio 1993).

On the other hand, there are examples of invasive species aiding in achieving restoration objectives. This is especially apparent in areas where practitioners believe that the possible negative consequences of an invasive species are outweighed by the necessity to restore a degraded landscape or community (Ewel 1999) or preserve native species (D'Antonio & Meyerson 2002). For example, Chen (2001) conducted research on Saipan, an island in the Mariana chain in the Pacific Ocean that is experiencing habitat degradation, especially in the form of soil erosion. The exotic *Leucaena leucocephala* (haole koa) was introduced to aid erosion control, and not only did it provide that service, but it also became a prime habitat for the endangered nightingale reed-warbler (*Acrocephala lusciniia lusciniia*). *L. leucocephala* was very similar to native species found in the ephemeral wetland habitat that the nightingale reed-warbler preferred, and provided the warbler with new habitat in lieu of its degraded wetland habitat. Another reason this introduction was important was because on Guam, one of the largest

neighboring islands, was invaded by the exotic brown tree snake (*Boiga irregularis*), that has caused multiple extinctions of native bird species (Chen 2001). Therefore, the restoration practitioners and conservationists decided it was important to continue promoting this exotic plant species as a way to restore and provide additional habitat to the nightingale reed-warbler, hopefully allowing it to escape extinction.

Another example of an invasive nitrogen fixing tree species that has been beneficial to a degraded habitat is *Albizia lebbbeck*. In South America, there has been a tremendous loss of the native rainforest due to mining and destructive agricultural practices (Lanly 1982). *Albizia lebbbeck* (lebbbeck tree), native to New Guinea, Northern Australia, and the Indomalaya region (Brown 1997), has been intentionally planted in these degraded areas. In degraded pastures in Puerto Rico, areas intentionally planted with *A. lebbbeck*, the aboveground plant biomass was 11 times greater and the root and forest floor biomass was 7 times greater than in areas not planted with *A. lebbbeck* (Parrotta 1992). Also, in *A. lebbbeck* plots, there was 1.9 times higher total carbon and 1.6 times higher total nitrogen stores than in the plots without *A. lebbbeck*. With management and careful site selection, Parrotta (1992) stated that the planting of *A. lebbbeck* can provide much needed rehabilitation through increased nutrient cycling and accelerated succession.

This conversion of atmospheric nitrogen to plant available forms is an ecosystem function that some invasive species can offer systems that have been severely disturbed or degraded, which can jumpstart the nitrogen cycle (Ewel & Putz 2004), thus facilitating succession. There have been a number of species identified by agronomists and foresters to be effective at colonizing and replenishing nitrogen pools at disturbed sites

(MacDicken 1994; Englert *et al.* 2002), and thus some advocate for species to not be dismissed simply because they are not native. This is similar to the argument by Davis *et al.* (2011) for the focus to be applied to species function and impact, and not on biogeographic origins (Davis 2011).

There are other situations in which exotic species are able to provide certain services that native plants cannot, or they are just more effective than similar native species. For example, in some cases the exotic species are better at providing shade to protect native species and facilitate colonization by acting as a nurse plant (Parrotta, Turnbull & Jones 1997; Lugo 2004), provide habitat structure and food for native animal species (Holl 1998; Carrière *et al.* 2002), or provide phytoremediation, such as the brake fern (*Pteris vittata*) which accumulates arsenic (Ma *et al.* 2001). Thus, there is great potential for exotic species to be used for restoration in certain systems (Ewel & Putz 2004), the application of which must be examined on a case-by-case basis.

Invasive plants: Disturbance opportunists

The role invasive plants play in restoration/reclamation is somewhat controversial and often exists as a contradiction. For example, in many landscapes, restoration could be achieved simply through the removal of an exotic species. In other words, the impetus for the restoration is the invasion of a widespread, and likely damaging exotic plant. One widespread example of this is the invasion by *Phragmites australis* throughout North American wetlands (Chambers, Meyerson & Saltonstall 1999; Saltonstall 2003; Kettenring & Adams 2011). Not only does the invasion by *P. australis* contribute to a loss of biodiversity, particularly displacing the native *Spartina alternatiflora* (Chambers,

Meyerson & Saltonstall 1999; Keller 2000; Bertness *et al.* 2002), it also alters wetland biogeochemical cycles (Meyerson, Chambers & Vogt 1999; Meyerson *et al.* 2000; Findlay *et al.* 2003) and degrades the overall habitat for native fish and wildlife (Fell *et al.* 2003, 2006; Gratton & Denno 2006). In these circumstances where the invasive plant is believed to be the driver of the undesirable ecosystem impact, removal of the invader is the focus of the restoration program. However, this can be difficult to accomplish.

Managing *P. australis* is not only costly (Martin & Blossey 2013), but using just one strategy (e.g. herbicide only) is unlikely to be effective and thus restoration requires a more systems approach on a case-by-case basis (Hazelton *et al.* 2014). Removal of a particular invasive species is also not usually enough to result in native species recolonization (Kettenring & Adams 2011; Suding 2011), with secondary invasions commonly following the eradication of the primary invasive species (Pearson *et al.* 2016), which may require native species to be replanted.

In more severely disturbed landscapes, invasive plants are often the first species to colonize, even if they were not previously present in the pre-disturbed habitat. This creates a variety of issues when trying to restore the native plant community. For example, *Morella faya*, a nitrogen fixing species native to the Azores, Madeira and Canary Islands (Melville 1979), was originally introduced to Hawaii in the late 19th century from Portugal (Vitousek *et al.* 1987). *M. faya* was planted heavily in the 1920s and 1930s for watershed reclamation, but that effort was later abandoned when *M. faya* was deemed to be too aggressive, as it was colonizing unintended areas (Fosberg 1937). *M. faya* colonizes lava flows and volcanic ash that are nitrogen depauperate (Vitousek *et al.* 1987; Vitousek & Walker 1989) and, as a consequence, significantly increases the

total nitrogen input in areas where the invader is abundant, fixing >4 times the amount of nitrogen as all the other native biological sources combined (Vitousek & Walker 1989). This high rate of nitrogen fixation facilitates both the early colonization of new lava flows, increasing soil fertility and availability of nitrogen to *M. faya* invaded areas, as well as leading to more rapid community development following *M. faya* invasion (Vitousek & Walker 1989), possibly initiating an invasion meltdown (Simberloff & Holle 1999). This is one of many examples of early colonization by exotic species that shifts the ecological trajectory of community development, and thus possibly ecosystem function.

The invasion of *Morella faya* on lava flows in Hawaii is a classic example of one avenue in which the effects of an invasive plant can persist, even following their removal, termed legacy effects (D'Antonio & Meyerson 2002; Suding, Gross & Houseman 2004). Legacy effects are the changes in the biological community, physical environment, or soil chemistry that persist after the invasive species has been removed. Well-known legacy effect by invasive species can occur through changes to soil chemistry, such as altering pH, organic content, salinity, and in particular, nitrogen (Liao *et al.* 2008; Ehrenfeld 2010). The legacy effects of invasive species can occur via alternative avenues as well, such as leaving behind a large seedbank, making a one-time removal impractical (Corbin & D'Antonio 2012), or through the hybridization between invasive species and native species, leading to decreased native biodiversity (Vila, Weber & Antonio 2000).

Ironically, invasive plants are sometimes promoted by restoration practitioners if native species are not available to restore a particular desirable function (D'Antonio &

Meyerson 2002) or to perform a role in the development or maintenance of a particular habitat (Ewel & Putz 2004). This occurs more often in reclamation situations than restoration projects, primarily due to the site being too heavily degraded for native plants to thrive and perform the necessary functions, or in some cases, the sites are so degraded, the natives cannot survive and establish (D'Antonio & Meyerson 2002).

For example, the Appalachian region of the United States, which extends from southern New York to northern Mississippi, is home to the most extensive temperate deciduous forest in the world, and includes forest types that are some of the most biologically diverse non-tropical ecosystems in the world (Ricketts *et al.* 1999). However, there has been significant forest loss in Appalachia due to surface mining (Drummond & Loveland 2010), which has resulted in fragmentation and a net loss of >400,000 ha of productive forests (Wickham *et al.* 2007; Saylor 2008), and with each following year, >10,000 ha are slated to be mined (Zipper *et al.* 2011b). Reclaiming these extremely disturbed landscapes to restore native and fully functioning ecosystems is extremely difficult, especially considering the countless examples where disturbance, anthropogenic or natural, promotes exotic species invasion (Hobbs & Huenneke 1992; Lozon & MacIsaac 1997; D'Antonio *et al.* 1999). Invasive plants are extremely common across the reclaimed landscapes of Appalachia and have a contentious history with landowners.

Reclamation and Exotic Species

Given that reclamation is focused on the most severely degraded landscape, many of which experience tremendous physical disturbance, exotic species have historically been a common component of the post-disturbance plant community. In fact, many

invasive species occur on older coalmines as a legacy of past reclamation practices where they were intentionally planted, and often become established on more recent reclaimed coalmines as a result (Zipper *et al.* 2011b), and are viewed as precluding development of native forests. However, generalizations of exotic plants as universally bad can be misleading and could overlook a long-term perspective focused on ecosystem function.

For example, post-SMCRA, the conventional reclamation ground cover mix included fast growing and competitive grasses, such as tall fescue (*Festuca arundinacea*) and aggressive legumes, like sericea lespedeza (*Lespedeza cuneata*). These species were chosen to meet the short-term goals of providing erosion control and minimizing acid mine drainage (Burger *et al.* 2009). However, these exotic species are considered invasive in most systems, and could potentially hinder development of post-mining land use goals that require establishment of native species.

To address the conflicting short-term (establish ground cover in 5 years) and long-term goals (achieve a functioning native ecosystem approximating the pre-mining state) occurring under SMCRA, the Appalachian Regional Reforestation Initiative (ARRI) formed in 2004. ARRI is focused on facilitating the restoration of high quality forests on reclaimed coalmines, which they promote through the Forestry Reclamation Approach (FRA). The FRA is intended to establish site conditions that will create a suitable environment for native tree survival and growth while also enabling recruitment and establishment of unplanted native vegetation (Zipper *et al.* 2011b). The FRA has become popular with landowners and mine operators as an effective way of improving reclamation of productive forests while reducing reclamation costs (Burger *et al.* 2009).

Under FRA, the herbaceous vegetation is designed to provide ground cover goals for erosion control, and require few nutrients and water. However, unlike traditional reclamation mixes the FRA approach reduces competition with the desired native colonizers and planted trees, which it does through utilizing less competitive species. Current ground cover mixes include grasses such as redtop (*Agrostis gigantean*) and perennial ryegrass (*Lolium perenne*), and legumes such as birdsfoot trefoil (*Lotus corniculatus*) and white clover (*Trifolium repens*). Though all exotic, these species are much less competitive with the desired tree species than the plants typically used before the FRA was implemented, such as tall fescue (*Festuca arundinacea*) and other clover species (*Trifolium* spp.), They are also generally slow-growing, tolerant of a range of soil conditions, and have sprawling growth forms. (Burger *et al.* 2009; Zipper *et al.* 2011b). Burger *et al.* (2009) describes four stages for how succession and ground cover development should occur:

Stage 1: grasses dominate and provide most of the cover during the first two years;

Stage 2: legumes and native plants dominate and provide most of the cover around years four through six;

Stage 3: fast-growing nurse trees make up 10-20% of the total trees planted and are most noticeable around years six through eight;

Stage 4: by the time the trees close canopy, the planted crop trees dominate and provide the majority of cover after year eight.

This tree-compatible ground cover should allow for increased resources to native tree seedlings, but it does not cover the ground very rapidly or completely. Ideally, the

low cover would allow for faster native plant recruitment through seed dispersal, but there is concern that it may allow unwanted exotic species to invade alongside the desirable native species (Fields-Johnson *et al.* 2012). Indeed, while the FRA is more successful at achieving native forest regeneration, invasive plants are common on FRA reclamation sites. However, the question remains of what the long-term consequences of exotic plants are on reclaimed sites, which must consider both reclamation goals (e.g., native forest species) and ecosystem function (e.g., nutrient cycles).

Plant function in novel ecosystems – opportunities and challenges

Undisturbed ecosystems maintain resilient nutrient and hydrologic cycles with succession moving the plant community toward a climax. However, once that system has undergone an abiotic or biotic disturbance, nutrient cycling and plant species composition often change (Seastedt, Hobbs & Suding 2008). Human-altered or built systems, so called “novel” ecosystems (Milton 2003; Hobbs *et al.* 2006; Williams & Jackson 2007) “differ in composition and/or function from present and past systems” (Hobbs, Higgs & Harris 2009). In other words, novel ecosystems have no naturally occurring counterpart (e.g., cities), which makes identifying the restoration objective difficult. Surface mines, and in particular mountain top removal, is a widespread novel ecosystem that is fundamentally different from the systems that predated the mining activity. Yes, forests previously existed on many of these locations, but the “soil”, hydrology, nutrient cycling, and plant community were eliminated such that these landscapes are better considered novel ecosystems.

When a system undergoes a dramatic abiotic change, it can be difficult for any plant species to be able to survive, thus changing the system drastically from its historical state and creating difficult work for restoration (Hobbs, Higgs & Harris 2009). The entire

notion of nativeness is nonsensical in novel ecosystems, which have no evolutionary history or historical ecosystem function. In these new and continually changing ecosystems—due to ongoing climate change that many restoration projects are now taking into account—many argue that the nonnative and invasive species could actually have key ecosystem roles (Williams 1997; Ewel & Putz 2004).

When implementing restoration activities, there is never a guarantee that it will end up as planned. In fact, failures are so common that some scientists have called into question our ability to actually meet the often ambitious goals (Miller & Hobbs 2007). By removing the idealistic view for a return back to a historical ecosystem, the options for restoration are now much less limited, and there could potentially be a reduction in resources and effort in order to achieve similar structural and functional goals (Hobbs *et al.* 2006). Holding to the rigid view that ecosystems have to continue with a particular species composition in a distinct place is becoming increasingly impractical, and creates unreasonable goals and expectations for restoration initiatives (Hobbs *et al.* 2006). Because of this, restoration goals are beginning to focus more on climate change resilience and ecosystem services (Nellemann & Corcoran 2010).

Surface mining creates some of the most severe impacts on ecosystems, with none of the original topography, vegetation, or soil remaining (Zedler, Doherty & Miller 2012), though SMCRA does require the approximate original contour to be restored to a given area, as well as topsoil being salvaged and replaced when possible (Zipper *et al.* 2013). Given these drastic changes in the abiotic and biotic environment, restoring the area back to the historical ecosystem, though optimal, is often unattainable (Pickett & Parker 1994; Harris *et al.* 2006; Hobbs *et al.* 2011). When looking at restoration and

reclamation from the perspective of the short-term, it is difficult to assess whether the path will continue to progress towards the goal or experience deviations, and whether or not those deviations will be a lasting result (Suding 2011). Thus, perhaps a shift in focus in reclamation is needed to a long-term functional ecological approach.

An exotic perspective – taking a functional focus

Reclaiming post-mining novel landscapes to historical structure and function exclusively through the use of natives is myopic, and may actually miss restoration opportunities. However, many restoration goals can still be achieved if species are chosen with a primary focus on function instead of nativity. It is well documented that there have been many restoration project failures, because it is difficult to actually meet the ambitious goals set forth during restoration projects (Hobbs 2007). If the perspective were to shift to achieving an ecosystem with realistic goals of ecosystem function and services, fewer resources could be expended to reach similar outcomes (Hobbs 2007). Exotic species may be favored, especially when their benefits and ability to restore areas that are severely disturbed outweighs any negative consequences that might be associated with their presence (Ewel *et al.* 1999). One important way in which this often occurs is through exotic nitrogen fixing species, which can “kick start” the nitrogen cycle in severely disturbed landscapes, providing nitrogen in nutrient limited systems (Ewel & Putz 2004), which may be best achieved through exotic species.

In novel ecosystems, nativeness is nonsensical, and it is argued that key ecosystem roles could in many cases be filled by exotic species when appropriate natives are not available (Williams 1997; Ewel & Putz 2004). Del Tredici (2010) cogently argues that our cities have been colonized by a spontaneous flora that is pre-adapted to these

urban novel ecosystems, and are providing valuable ecosystem services. Most of the plants that colonize cities are exotic, but filter water, provide habitat, cycle nutrients, and store carbon. Similarly, Pickett et al. (2008) have demonstrated that cities contain high functioning ecosystems (Pickett *et al.* 2008), and provide a valuable lens with which to view reclaimed surface mines.

Reclamation in the Eastern US is moving towards the return of native hardwood forests, has countless benefits, and is a laudable goal. However, the condemnation of exotic species is largely unfounded and contradicts the widespread use and application of some exotics. Viewing reclamation projects on a longer, more ecological time scale with an explicit functional perspective, though, could broaden our species pallet and may hasten goal attainment. The issue is not one of impact, but rather one of perspective on timescales and nativity, and whether or not that is more important than overall ecosystem functionality on these severely disturbed landscapes.

Thus, we echo and promote choosing species for reclamation that will meet the needs of that landscape. Without the confines of nativity, species should be chosen that meet the following criteria:

1. Documented to provide the ecosystem function(s) in question (e.g., wildlife habitat, nitrogen fixation);
2. Survives and thrives under the growing conditions under consideration;
3. Has a low probability of surviving in the surrounding landscape matrix to limit invasion outside the novel ecosystem in question;

4. Occupies a successional stage that will make way for the desired climax community on an ecological time scale and will not arrest the succession in an undesired state.

The criteria outlined provide guidance for choosing exotic species to meet reclamation goals, and should be evaluated on a case-by-case basis for the species under consideration, as well as the reclamation conditions. No species should be considered universally fit or unfit for reclamation, as invasions are not universal. To better illustrate our proposal and criteria, below we discuss a case-study of a specific exotic species in Appalachia.

Case study: Autumn olive on reclaimed mines in Appalachia

An exotic species that exists as a duality in reclamation is autumn olive (*Elaeagnus umbellata*), which is one of the most common species on former coalmines in the Appalachian region (Zipper *et al.* 2011b). It is a large shrub native to Pakistan, China, and Eastern Asia, and was brought to the United States in 1830 as an ornamental species (Dirr 1983). Around the 1960s, *E. umbellata* was planted for erosion control, wildlife conservation, and as a nurse species for tree plantations (Fowler & Fowler 1987; Lemke *et al.* 2013). As a result, *E. umbellata* was commonly planted on reclaimed mines post-SMCRA (Zipper *et al.* 2011b). However, it has since become reviled as interfering with FRA reclamation and preventing bond release.

E. umbellata can produce up to 24,000 seeds per mature plant annually (Ahmad, Sabir & Zubair 2006) and is a popular food source among wildlife in its native range, with drupes persisting on the branches into the winter and providing nourishment for birds and mammals when other food sources are not abundant (Fowler & Fowler 1987;

Kohri, Kamada & Nakagoshi 2011). Almost 99% of the *E. umbellata* seeds can germinate six days after the fruits are consumed and passed through a native crow species (*Corvus corone*) in Japan (Kohri *et al.* 2002). *E. umbellata* also has a rapid growth rate and a broad, dense crown (Evans *et al.* 2013). Once *E. umbellata* is established, it can develop intense shade, which suppresses native species, and hinder the dispersal and establishment of desirable woody species (Zipper *et al.* 2011b).

E. umbellata also fixes atmospheric nitrogen through an actinomycete symbiosis in root nodules (Ahmad, Sabir & Zubair 2006), which enables it to establish as a pioneer species on degraded landscapes. Due to their competitive advantage in nutrient deficient locations (Walker & Syers 1976; Vitousek & White 1981), nitrogen fixing species are often the dominant species during primary succession that occur following large disturbances (Stevens & Walker 1970; Gorham, Vitousek & Reiners 1979; Reiners 1981). *E. umbellata* is no exception, as it readily colonizes reclamation sites and can quickly become dominant.

Once *E. umbellata* is present in an ecosystem, its potential to impact other plant species and ecosystem processes, as well as to fix nitrogen (Schlesinger & Williams 1984), gives it the potential to alter the long-term development of forests (Moore *et al.* 2013). For example, in a restoration experiment conducted by Evans *et al.* (2013), volunteer species represented 65% of the total individual plants present in the fourth growing season, with autumn olive covering 27% of the of the total experimental area and producing much greater volume than native tree species. Unaided ecological succession is usually not adequate to restore native vegetation if invasive species are present and persisting on reclaimed mine sites. Therefore, it is believed that if invasive

species such as autumn olive are not managed, successful reclamation will be hindered. Since the time autumn olive was intentionally planted on mine sites in the 1960s, it has thrived on these degraded sites and has become a major invader on many reclaimed mine sites (Lemke *et al.* 2013). As is the case with many intentionally introduced species, autumn olive has gone from degraded landscape benefactor to pariah.

In these primary successional stages, nitrogen-fixing species are known to increase ecosystem nitrogen availability (Binkley, Cromack & Fredriksen 1982), accrue a large amount of macronutrients in their tissues (Turner & Singer 1976; Boring, Monk & Swank 1981), and accelerate the accumulation of organic matter and nitrogen in the soil (Turner & Singer 1976; Youngberg & Wollum 1976), and they are typically fast growing species. *E. umbellata* continues to provide these valuable ecosystem services, the value of which have been superseded by concerns of meeting short-term policy mandates. For example, when the reclamation goal is pasture, the percentage of wildlife shrubs, which *E. umbellata* is considered to be due to habitat and forage provisions, should be very small. Therefore, when *E. umbellata* is the predominant species on a pasture site, the bond will not be released due to failure of successful reclamation (personal communication with Eddie Clapp of Red River Coal Company, January 16, 2015). For example, >40 ha of land was recently bulldozed on a Southwest Virginia reclamation site to remove a near monoculture of *E. umbellata*(personal communication with Eddie Clapp of Red River Coal Company, January 16, 2015).

Therefore, in reclamation sites not yet released from bond, some effort and expense is expended to manage *E. umbellata*. For example, Byrd *et al.* (2012), looked at the effectiveness of *E. umbellata* management methods on an older legacy mine area, and

they found that treating *E. umbellata* with a foliar herbicide costs \$741 and 2 hours per hectare in areas where the autumn olive cover is >95%, but it is not effective at controlling re-sprout. However, the more effective re-sprout control, the cut/stump herbicide method costs \$1,166 and 6.5 hours per hectare to complete. All of the management techniques employed were determined to require continual management, and each method caused additional disturbance, consequently promoting secondary invasions by other exotic species (Byrd *et al.* 2012).

Though *E. umbellata* is currently viewed as preventing successful reclamation, there is little research showing the negative ecological or successional impacts of *E. umbellata* in these systems. When an open pasture is the desired land use, the invasion by this woody shrub can be a threat to birds that require open grassland areas in order to nest and raise their young (Bajema *et al.* 2001; Ingold, Dooley & Cavender 2009). However, since reclamation is moving more towards the return and development of native forests, we need a better understanding of the actual impact *E. umbellata* is having on forest succession, as there may be only a perceived ecological impact. With a long-term ecological perspective, *E. umbellata* could be accelerating succession and improving ecosystem function, in a similar fashion to *Morella faya* and *Albizia lebbek*. If we look beyond simply the presence of this reclamation pariah where it was not planted, are *E. umbellata*-invaded landscapes having reduced ecosystem function across the range of relevant processes (e.g., hydrology, microbial dynamics, biogeochemical cycles, wildlife trophic dynamics)? Or is this primarily a matter of perspective on nativity and impact that is clashing with shortsighted regulation? A perception of negative impacts from invasive species is widespread and largely unsupported with empirical evidence (Barney *et al.*

2013). Policy mandates notwithstanding, broader ecological considerations must be acknowledged in severely degraded novel ecosystems, and plants must be chosen, and tolerated, that best meet ecological goals.

However, referencing *E. umbellata* against the criteria we set out for choosing reclamation species does not leave us with a simple answer. We do know that *E. umbellata* fixes nitrogen, provides habitat, and has been documented as a food source for many species in the Appalachians – all ecosystem functions needed on reclaimed mine sites. We also know that *E. umbellata* is very good at surviving and dispersing in reclaimed areas. However, *E. umbellata* can form very dense stands, which have the potential to shade and outcompete many other plant species. While it is a primary successional species on these sites and is ameliorating the soil through nitrogen addition, it is not known whether or not over a more ecological time scale of forest succession (i.e. 100 years) *E. umbellata* will make way for the desired climax community or whether it will arrest succession at that stage until further management action is taken. Our last criteria is whether or not *E. umbellata* has a low probability of surviving in the surrounding landscape matrix. This is another hard one to know, since it depends on the mine adjacent areas. If a mine was adjacent to an intact, relatively undisturbed forest, *E. umbellata* would probably not be very successful in establishing in that area. However, the reality of these mined areas is that they are often adjacent to other types of disturbances (i.e. roads, development). Since we know *E. umbellata* is a disturbance opportunist and is seen commonly along roadsides and other disturbed areas, the likelihood that it would not spread into the surrounding landscape matrix is likely low. However, with all of these criteria and *E. umbellata*, there is still not enough empirical

research that has been conducted to answer many of these questions. Therefore making conclusive remarks on whether or not the pros of *E. umbellata* outweigh the cons in the context of reclamation is not a practical decision to make currently.

In our era of rapid global change and a predicted continued expansion of natural resource extraction, returning disturbed landscape to desirable ecosystem functions and services is of upmost priority. Shifting our perspective to one of function calls into question the censure of some nonnative species whose ecological benefits have not been fully appreciated or are unknown. Reclamation, much like restoration, must continue to progress and meet societal, ecological, and conservation goals, utilizing the best species available, native or nonnative.

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Chapter 2

Autumn olive performance varies in different reclamation conditions

Abstract

Surface coal mining has caused significant disturbance in the Appalachian region of the United States, fragmenting and causing the loss of >400,000 hectares of the world's largest temperate deciduous forest. The Forestry Reclamation Approach (FRA) is being promoted across Appalachia for its use of a "tree-compatible ground cover" seed mixture, which creates more exposed areas to encourage colonization by native species, but also makes reclaimed sites more susceptible to invasion by exotic species. A problematic invasive species on reclaimed sites in Appalachia is autumn olive (*Elaeagnus umbellata*), an exotic nitrogen-fixing shrub. To better understand how reclamation conditions affect autumn olive, we conducted two separate, complementary studies assessing effects of substrate and vegetation seeding on autumn olive recruitment and performance. In each experiment, we also manipulated the standing plant community of each experiment into four plant community treatments to further examine the effects of resident plant composition on autumn olive recruitment and performance. In spring 2014, we planted 6,800 autumn olive seeds into both experiments, but saw no germination. In spring 2015, we transplanted 480 one-year-old autumn olive seedlings into both experiments. At the end of the 2015 growing season, autumn olive performed better in weathered substrates and bare ground community plots, and also performed better in the conventional seeding treatment area. While the first year shows clear differences in autumn olive performance and site conditions, the continuation of this experiment will further our understanding of reclamation conditions and invasive species establishment.

Introduction

The Appalachian region of the United States, which extends from southern New York to northern Mississippi, is home to the most extensive temperate deciduous forest in the world, and includes forest types that are some of the most biologically diverse non-tropical ecosystems (Ricketts et al., 1999). Forests in the Appalachian region provide many ecosystem services, including watershed control, water quality protection, carbon sequestration, wildlife habitat, and native plant diversity (Burger *et al.* 2009). They are also important in terms of their economic value through the forest industry, providing commercial timber, and generating raw materials for products used worldwide (Zipper *et al.* 2013).

However, there has been significant forest loss in the Appalachian region due to surface mining primarily for coal (Drummond & Loveland 2010), which has resulted in fragmentation and a net loss of productive forests (Wickham *et al.* 2007; Sayler 2008). Since 1930, >2.4 million hectares across the U.S. have been surface mined (Zelevnik & Skousen 1996). Coal mining alone has affected >600,000 hectares in Appalachia, and the majority of this previously mined land remains unproductive, unmanaged, and invaded by non-native plants (Burger *et al.* 2013) largely due to a lack of uniform reclamation standards.

Reclamation practices became mandated and standardized in the US in 1977 with the passage of the Surface Mining Control and Reclamation Act (SMCRA), driven by the desire to solve the severe erosion, sedimentation, and landslides that were common following surface mining. SMCRA requires mined land be restored to its original use or use of higher value, and the mining operator has to obtain a permit and pay a bond to the

government prior to mining to assure the land will be reclaimed. The most direct way to meet these requirements in Appalachia is to restore the native hardwood forest (Holl, Zipper & Burger 2009), though pasture is also a common reclamation target. However, because the reclamation is typically evaluated only five years after reclamation efforts are completed (Zipper *et al.* 2011b), many projects focus on the short-term goals of providing erosion control and minimizing acid mine drainage, and not on more long-term goals and the restoration of ecosystem services which are much harder to quantify (Lemke *et al.* 2013).

The Appalachian Regional Reforestation Initiative (ARRI) formed in 2004 as a result of the conflicting short and long-term goals occurring under SMCRA. ARRI's focus is to encourage the restoration of high quality forests on reclaimed coalmines (Angel *et al.* 2005), which they do by promoting the Forestry Reclamation Approach (FRA). The FRA is intended to establish site conditions that will create a suitable environment for planted tree survival and growth while also enabling colonization by native vegetation (Zipper *et al.* 2011b). The FRA is becoming more popular with landowners and mine operators as an effective way of improving the post mining productive forest (Burger *et al.* 2009). However, land reclaimed using the FRA results in more bare ground compared to non-FRA reclamation sites. The additional bare ground allows colonization by desirable native species and reduced competition for the planted natives. However, an unintentional consequence of this strategy may be the increased opportunity for invasion by exotic plants (Fields-Johnson *et al.* 2012).

The highly disturbed environment of reclaimed sites, combined with the poor growing conditions and lack of competition, often favor invasive plants (Lemke *et al.*

2013). Invasive plants have the potential to change ecosystems and influence their economic and ecological productivity (Lemke *et al.* 2013) through alteration of basic ecological processes, such as nutrient cycling and alteration of soil biota (Evans *et al.* 2013). Since many reclamation efforts are focused on rapid plant establishment and erosion control, fast-growing exotic species are commonly planted which may slow or prevent the establishment of later-successional, native species (Ricciardi 2007). Thus, many invasive species occur on older coalmines as a legacy of past reclamation practices, and often become established on more recent reclaimed coalmines as a result (Zipper *et al.* 2011b). In addition to the historic planting of invasive species, current reclamation practices may be facilitating the establishment of unplanted invasive species colonizing from surrounding areas.

Autumn olive (*Elaeagnus umbellata*) is one of the most common invaders on current and former coalmines in the Appalachian region (Zipper *et al.* 2011a). Autumn olive is a large shrub native to Pakistan, China, and Eastern Asia, and was brought to the United States in 1830 (Ahmad, Sabir & Zubair 2006). In the 1960s, autumn olive was planted for erosion control, wildlife conservation, and as a nurse species for tree plantations, as well as for mine reclamation (Lemke *et al.* 2013). Autumn olive is a pioneer species due to its ability to fix atmospheric nitrogen, grow in acidic, loamy soils, and produce up to 14 kilograms of fruits per plant (Lemke *et al.* 2013). Autumn olive also rapidly produces a broad, dense crown (Evans *et al.* 2013), which suppresses native species possibly through shading (Lemke *et al.* 2013). Thus, autumn olive is being actively managed on pre-bond-release reclamation sites in particular to mitigate negative effects on reclamation success. Invasive species, and autumn olive in particular, present a

common challenge to reclamation success, but little information exists on how specific reclamation conditions affect invader recruitment and performance (Zipper *et al.* 2011b).

Vegetation succession on reclaimed coalmines is primarily constrained by the substrates available for that area (Alday, Marrs & Martínez Ruiz 2011). When coalmines are no longer in production, the available substrate materials are often unsuitable due to their structural and physiochemical characteristics (Felinks and Wiegand, 2008). For example, Alday *et al.* (2011) found that when resident topsoil was used, the native shrub community was able to develop within 15 years, while in contrast when topsoil was not added, it took >40 years for the native shrub community to develop (Alday, Marrs & Martínez Ruiz 2011).

The soils found on the Appalachian Mountains are difficult to salvage on pre-mining slopes because they are very thin and rocky (Daniels and Amos, 1985). Therefore, on surface mines in Appalachia, landscapes are rebuilt using fractured geologic materials, otherwise known as minespoils (Fields-Johnson *et al.* 2012). In order to create suitable rooting medium for tree growth on reclaimed mine sites, the FRA recommends the use of salvaged topsoil, weathered sandstone, or whatever the best available material is at a specific site. Soil pH is known to have a strong influence on vegetation (Ellenberg *et al.*, 1991), and soils with a pH of 5.0 to 7.0, low pyritic sulfur content, and loamy textures for drainage purposes are preferred for reclaiming lands for forest establishment and use (Zipper *et al.* 2013). The FRA also advocates for loosely graded substrate to facilitate planted tree root growth, infiltration, and aeration (Fields-Johnson *et al.* 2012). Thus, preference is given to more highly weathered minespoils, which have more favorable, though still harsh, growing conditions.

Once substrate is selected, mining companies will seed a vegetation mix using a hydroseeder that sprays the seeds, as well as fertilizer, across the desired reclamation site. Since one of the main criteria for species selection is erosion control, most conventional mixes contain fast-growing, exotic species. However, the FRA developed a tree-compatible mix that intentionally selected less-competitive species in order to promote forest succession through native colonization (Fields-Johnson *et al.* 2012), but this mix's susceptibility to colonization and invasion by undesired species is not well known.

Here we investigated how different reclamation conditions affect the establishment and performance of autumn olive. Specifically, we compared the two elements of reclamation that are most likely to influence invader establishment and performance: substrate material and planted vegetation mixes. In each experiment we manipulated the resident plant community prior to planting autumn olive to identify possible facilitation or suppression on autumn olive recruitment and performance. This study will inform both basic ecological interactions and reclamation practices.

Materials and Methods

The following experiments were carried out at the Powell River Project (PRP), located on mined land in Wise County, Virginia. Started in 1980, the PRP is a 1,100 acre area that is used cooperatively by Virginia Tech, natural resource industries, and other educational institutions to conduct research aimed at enhancing Appalachian restoration techniques and water protection. The PRP has a long research history addressing environmental protection and restoration.

To meet our objectives, we conducted two separate, but complementary studies. To look at the effect of substrate on autumn olive, we identified a site with one half of the

mountain face laid with a brown, weathered sandstone substrate and the other half laid with a gray, unweathered sandstone substrate. The weathered substrate had a pH of 5.6 and originated from a stratigraphic location closer to the surface and directly beneath the original topsoil, thus it was more weathered. The unweathered substrate had a pH of 7.2, and was less weathered due to its origination from lower in the rock strata further beneath the surface. Both sides were seeded in 2011 by the mining firm with the same conventional species mix containing the following: perennial ryegrass (*Lolium perenne*); birdsfoot trefoil (*Lotus corniculatus*); ladino clover (*Trifolium repens*); weeping lovegrass (*Eragrostis caryula*); rye grain (*Secale cereal*); orchardgrass (*Dactylis glomerata*); Korean lespedeza (*Kummerowia stipulacea*); and redtop (*Agrostis gigantean*).

To look at the effect of plant community composition that has established post-initial seeding on autumn olive recruitment and performance, we identified a site with one half of the mountain face seeded with the conventional vegetation mix, and the other half seeded with the tree-compatible vegetation mix in 2008 by the mining firm. The underlying substrate is a mostly unweathered sandstone and mudstone mix with a pH 7.4. Both mixes contained perennial ryegrass (*Lolium perenne*), birdsfoot trefoil (*Lotus corniculatus*), ladino clover (*Trifolium repens*), and weeping lovegrass (*Eragrostis caryula*). The tree compatible mix also contained annual ryegrass (*Lolium multiflorum*) and timothy (*Phleum pratense*), whereas the conventional mix contained rye grain (*Secale cereal*), orchardgrass (*Dactylis glomerata*), Korean lespedeza (*Kummerowia stipulacea*), and redtop (*Agrostis gigantean*). The following trees were planted on both sites: white ash (*Fraxinus americana*), white oak (*Quercus alba*), sugar maple (*Acer*

saccharum), black cherry (*Prunus serotina*), red oak (*Q. rubra*), chestnut oak (*Q. prinus*), black oak (*Q. velutina*), yellow poplar (*Liriodendron tulipifera*), gray dogwood (*Cornus racemosa*), red mulberry (*Morus rubra*), redbud (*Cercis canadensis*), white pine (*Pinus strobus*), and shagbark hickory (*Carya ovata*).

In both the substrate and vegetation sites, we also wanted to examine the effects of resident plant composition that has formed post-seeding on autumn olive establishment and growth. We used the following four plant community treatments in both locations established in 10 randomly located 2x1m plots:

- 1) full resident community;
- 2) grasses only;
- 3) broadleaves only;
- 4) no plant community - bare ground;

The bare ground, grass, and broadleaf only plots were initially treated with herbicides, and then managed by hand clipping. The overall design was a split-plot design with the main plot being the plant community, and one side of the plots were planted with 42 autumn olive seeds in the spring of 2014, while the other side was planted with 3 one-year-old bare root autumn olive transplants in early spring 2015.

Plots were monitored monthly for germination and survival. We also recorded autumn olive height and basal diameter from May to October of 2014 and 2015. Basal diameter is also being recorded due to the fact that autumn olive puts out many branches, starting at a young age, and having both measurements could also account for multiple types of growth. Percent cover of autumn olive and all other species was taken throughout the spring and summer of 2014 and 2015. At the conclusion of the project, the

plots will be sprayed with herbicide to eradicate all autumn olive and will be monitored and spot treated for at least a year.

For each experiment, we conducted a logistic regression to test for treatment effects on autumn olive survival. We also conducted two-way analysis of variance (ANOVA) to test for the effect of substrate types or vegetation mixes and plant community composition on change in height and change in basal diameter of autumn olive. Differences in survival, change in height, and change in diameter among treatments were determined using Tukey-Kramer HSD. Statistical analyses were conducted using JMP Pro 11.

Results

For both experiments, none of the 6,800 autumn olive seeds germinated in any of the plots in 2014 or 2015. Therefore, we are only reporting results of the establishment and performance of the autumn olive transplants from 2015.

Growth in different substrates

There were no differences in autumn olive survival among substrate types or community plots ($p > 0.5$). Autumn olive grew twice as tall ($p < 0.0001$, Figure 1) with double the growth in basal diameter ($p < 0.0001$, Figure 2) in the weathered substrate compared to the unweathered substrate. Autumn olive also grew almost four times taller ($p = 0.00017$, Figure 3) and had almost double the basal diameter growth ($p = 0.0030$, Figure 4) in the bare ground community plots than the full resident community plots and forbs only community plots. However, the interaction between the community composition and substrate type was not significant for height or basal diameter

($p=0.7797$; $p=0.2374$). Height growth and basal diameter growth were highly correlated ($p<0.001$).

Growth in different vegetation mixes

There were no differences in autumn olive survival among vegetation mixes or community plots ($p>0.5$). Autumn olive grew approximately 10 cm taller ($p=0.0194$, Figure 5) and had almost double basal diameter growth ($p=0.0244$, Figure 6) when planted into the conventional vegetation mix vs. the tree compatible vegetation mix. There were 38 plant species in our plots in the tree-compatible site, whereas there were 27 plant species in our plots in the conventional site. Of those plant species, there were 21 common species between both sites, with the 17 unique species present at the tree-compatible site and 6 unique plant species at the conventional site. There were no significant differences in autumn olive transplant's growth (height or basal diameter) among the four plant community treatments ($p=0.1026$; $p=0.2262$), nor were there any significant growth differences (height or basal diameter) in the interaction between the community treatment plots and vegetation mixes ($p=0.0983$; $p=0.5372$). Height growth and basal diameter growth were significantly correlated ($p=0.029$).

Discussion

Substrate Comparisons

Overall, autumn olive growth was much greater in the more highly weathered sandstone substrate, even though we saw no difference in the initial survival between substrate types. The greater growth in weathered sandstone is not surprising, considering

the overall better growing conditions of the weathered sandstone versus the unweathered sandstone. Unweathered gray mine spoils, though they will eventually weather into soils, are generally not suitable for restoring forests. For example, Zipper *et al.* (2013) reviewed comparison studies looking at survival and growth on weathered and non-weathered materials, and they found that planted tree survival was not different between weathered and unweathered minespoil, but tree growth was usually significantly greater on weathered materials (Torbert, Burger & Daniels 1990; Angel *et al.* 2008; Emerson, Skousen & Ziemkiewicz 2009; Miller *et al.* 2012). Early tree growth is commonly suppressed in unweathered minespoils (Zipper *et al.* 2011b). The more organic materials in the substrate, the greater the forest restoration potential. However, use of unweathered materials may be unavoidable depending upon availability from site to site (Skousen *et al.* 2011). Unweathered minespoils are available in much greater quantities on modern mines than weathered minespoils or native soils (Zipper *et al.* 2011b).

Among the plant community types, across both substrates, autumn olive grew taller and larger in bare ground plots, without competition than in the forb only or standing community plots. Bare ground creates the potential for the autumn olive transplants to exploit the limited available resources instead of having to compete with other fast growing forbs or the combination of forbs and grasses that are typically present four years following seeding of a site. This site was seeded in 2011, thus the vegetation was only 3 years old when we began and very small throughout the entire experiment. This could also explain why we didn't see meaningful differences in autumn olive growth between the three community plots that still had vegetation (grasses, forbs, or standing community). No matter what vegetation type was present, the percent cover of plots

(excluding bare ground plots) was very similar, especially at the sandstone site (sandstone average plant cover: $39\pm 14\%$; mudstone average plant cover: $67\pm 11\%$).

Vegetation Comparisons

Contrary to our expectation, the autumn olive transplants grew significantly larger in the conventional mix versus the tree-compatible mix. This result is surprising considering the conventional mix was used pre-FRA, and focused on using species that were more aggressive colonizers to prevent soil erosion. The conventional mix incorporated species such as the very competitive Korean lespedeza, unlike the tree-compatible mixture which incorporates species that would be intentionally less competitive to allow for native colonization of desired, forest-compatible species (Fields-Johnson *et al.* 2012). However, this site was set up in 2008 and had well-established plant communities, with species that were included in the initial mixes as well as volunteers. When looking at just the species richness of each site, the tree-compatible site had 38 species total, compared to the conventional site, which had 27 total species. There were 21 common species between both sites, but the variation in species assemblage at the sites could speak to the difference in autumn olive growth, with the larger number of unique species in the tree-compatible site hindering the success of autumn olive in some way. Again, we are unable to comment on the competitive ability autumn olive could have if it had been planted or colonized the same area during the initial reclamation seeding.

Restoration approaches of seeding herbaceous species have not always been successful in creating self-sustaining ecosystems (Gonzalez-Alday *et al.*, 2009). Nutrient

availability also affects the composition of the success of the plant communities (Evans *et al.* 2013). In the Appalachians, soil organic nitrogen pools and cycles have been identified as essential processes for ecosystem recovery (Zipper *et al.* 2013). Thus, nitrogen-fixing plants, such as autumn olive, often make up large components of plant communities on reclaimed mine sites, which suggests that nitrogen available to plants is a common influence on the plant community (Zipper *et al.* 2011b). In order to adequately inform management decisions, there needs to be a better understanding of the different aspects that promote successional processes after the large land disturbances created by surface mining (Alday, 2011). The Forest Reclamation Act is a step in the direction of aiding forest restoration on mine sites, especially in the Appalachian region of the United States.

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Figures

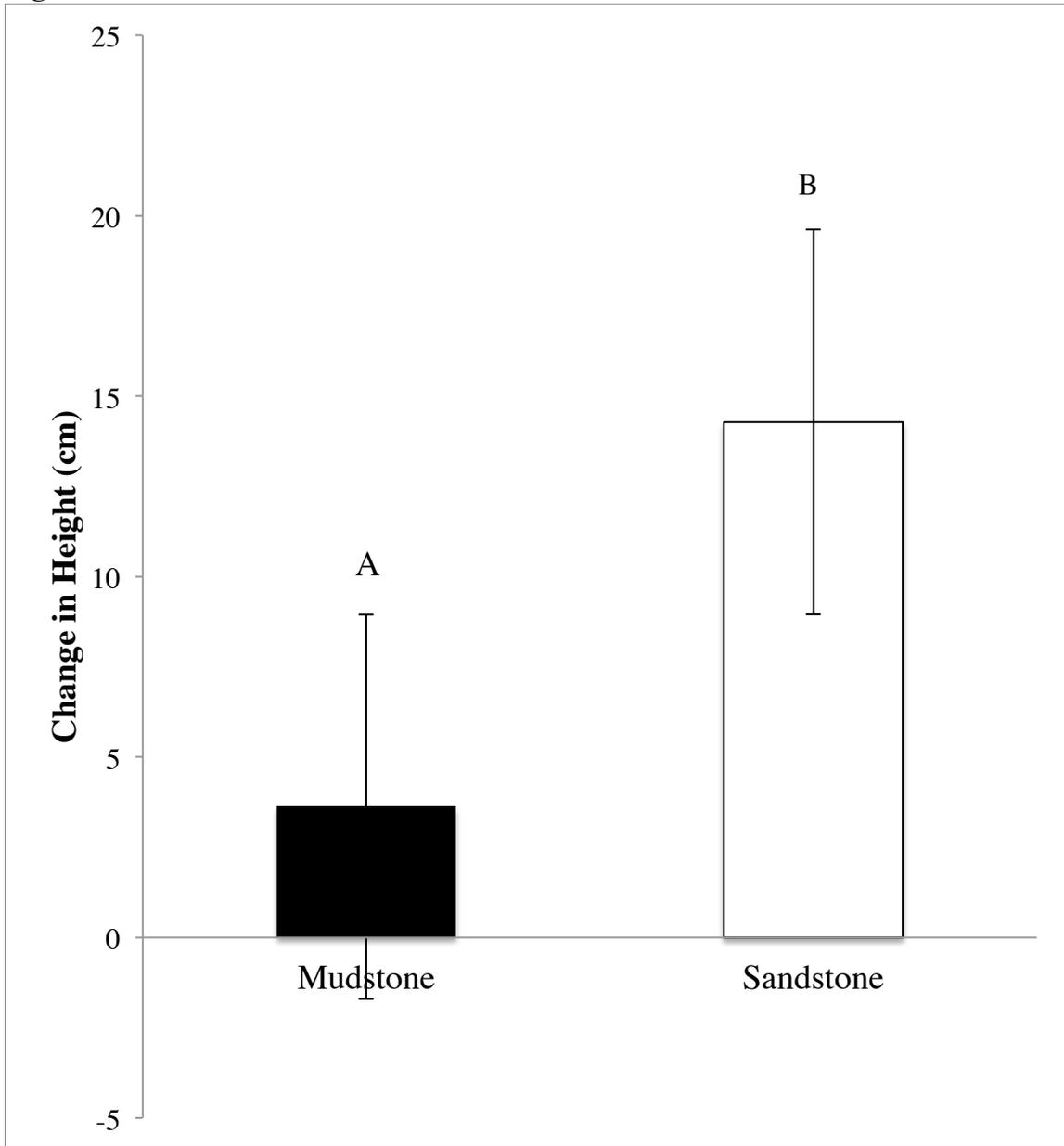


Figure 1. Mean change in height of autumn olive by substrate type with standard error.

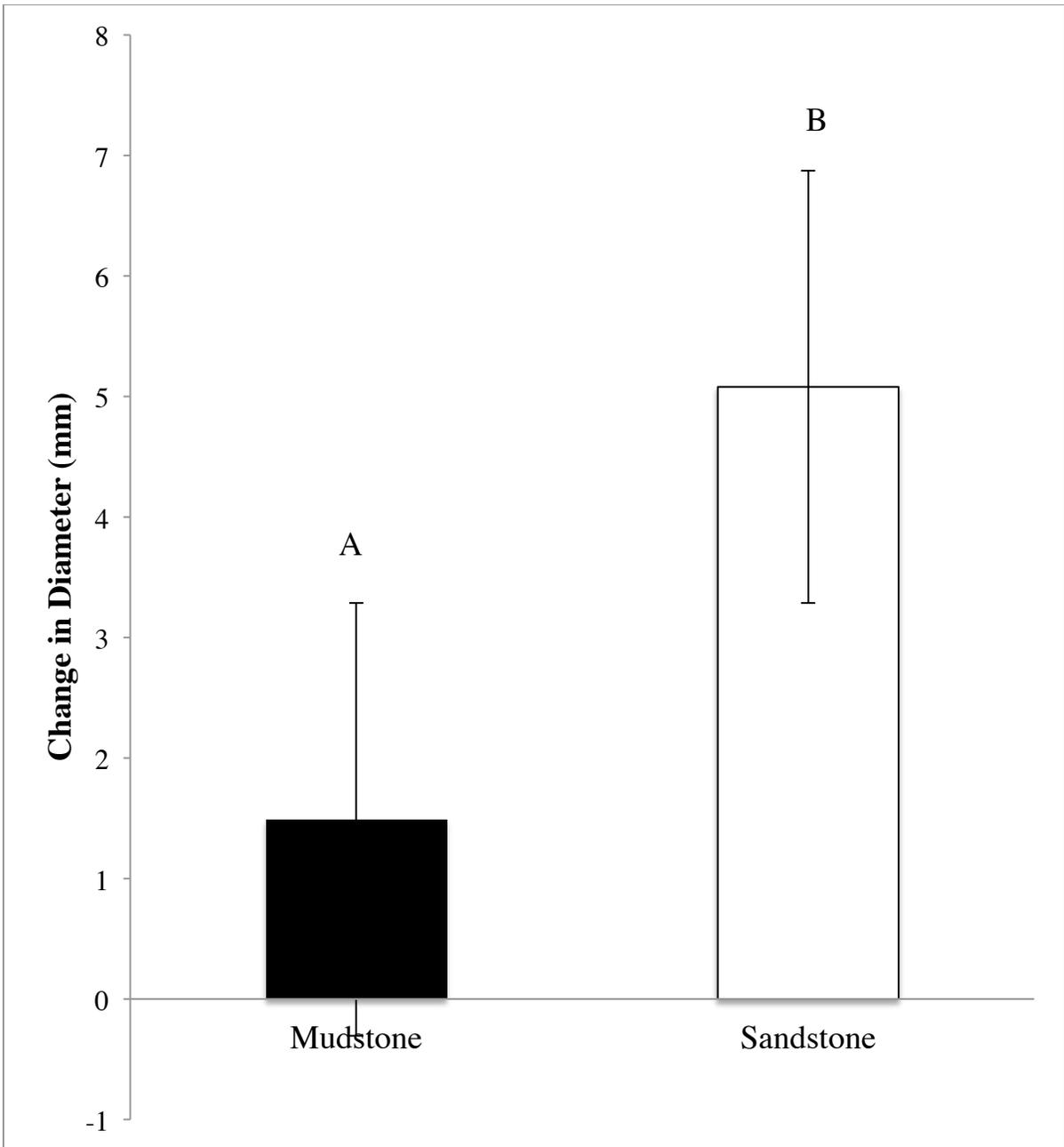


Figure 2. Mean change in basal diameter of autumn olive by substrate type with standard error.

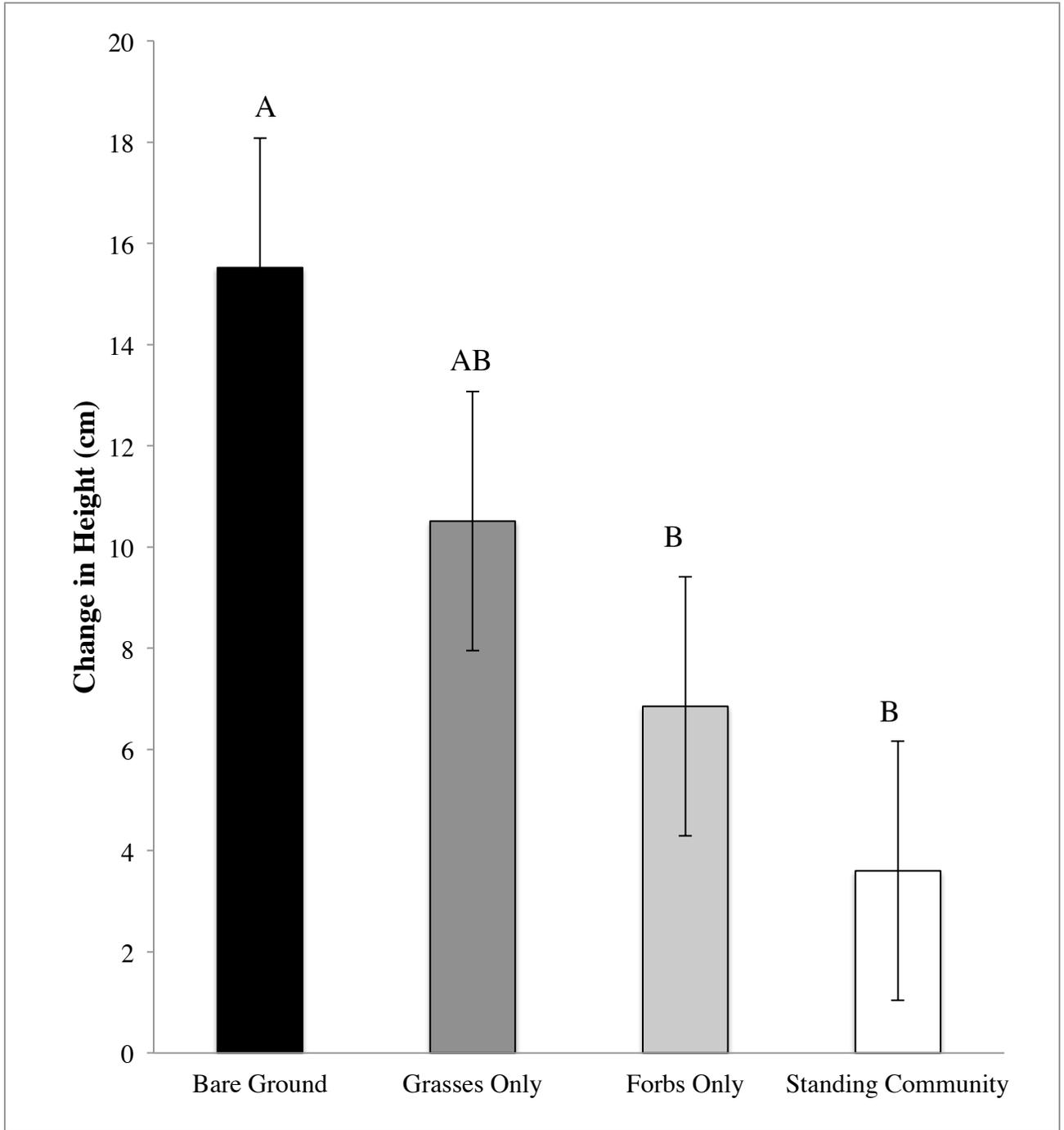


Figure 3. Mean change in height of autumn olive by community plot in substrate sites with standard error.

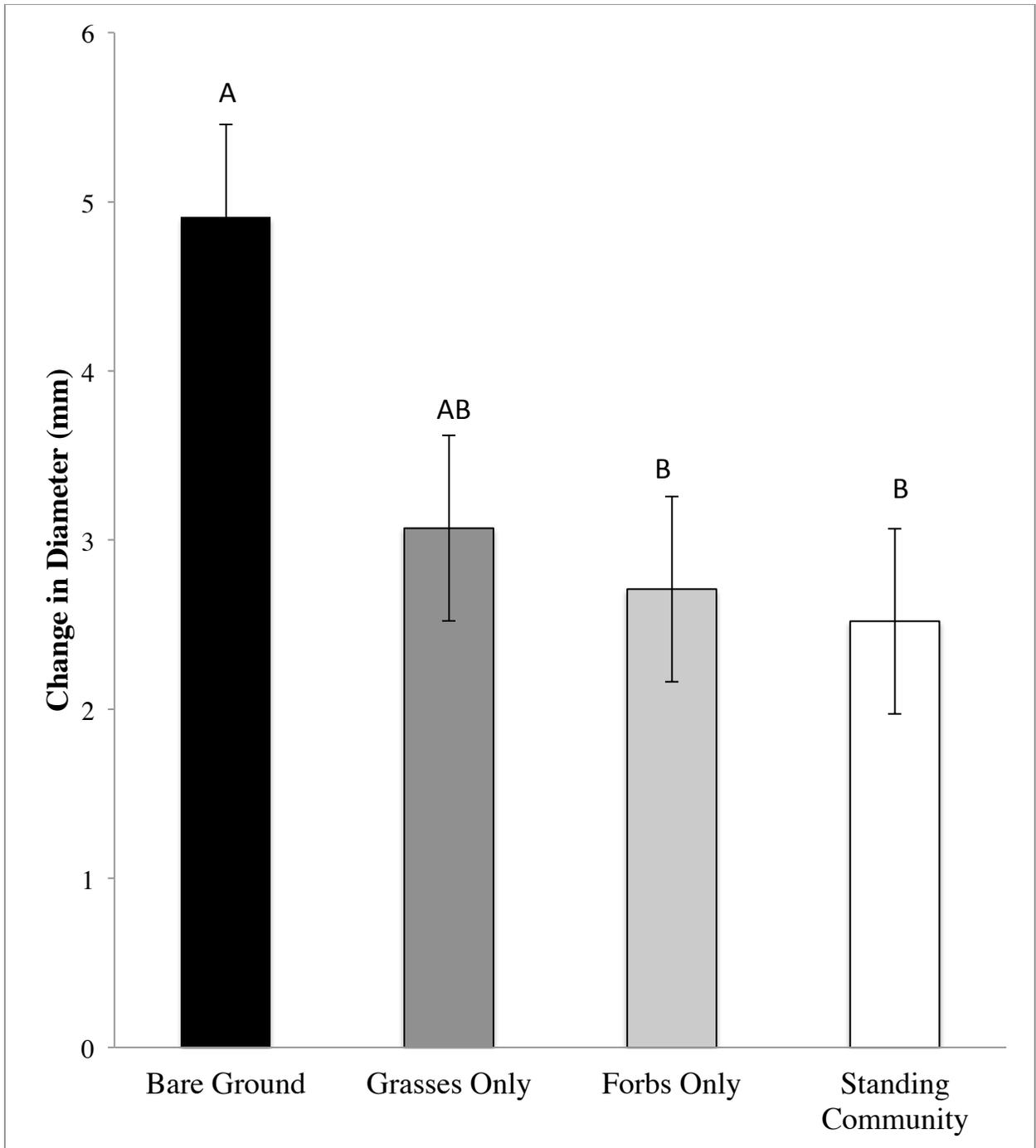


Figure 4. Mean change in basal diameter of autumn olive by community plot in substrate sites with standard error.

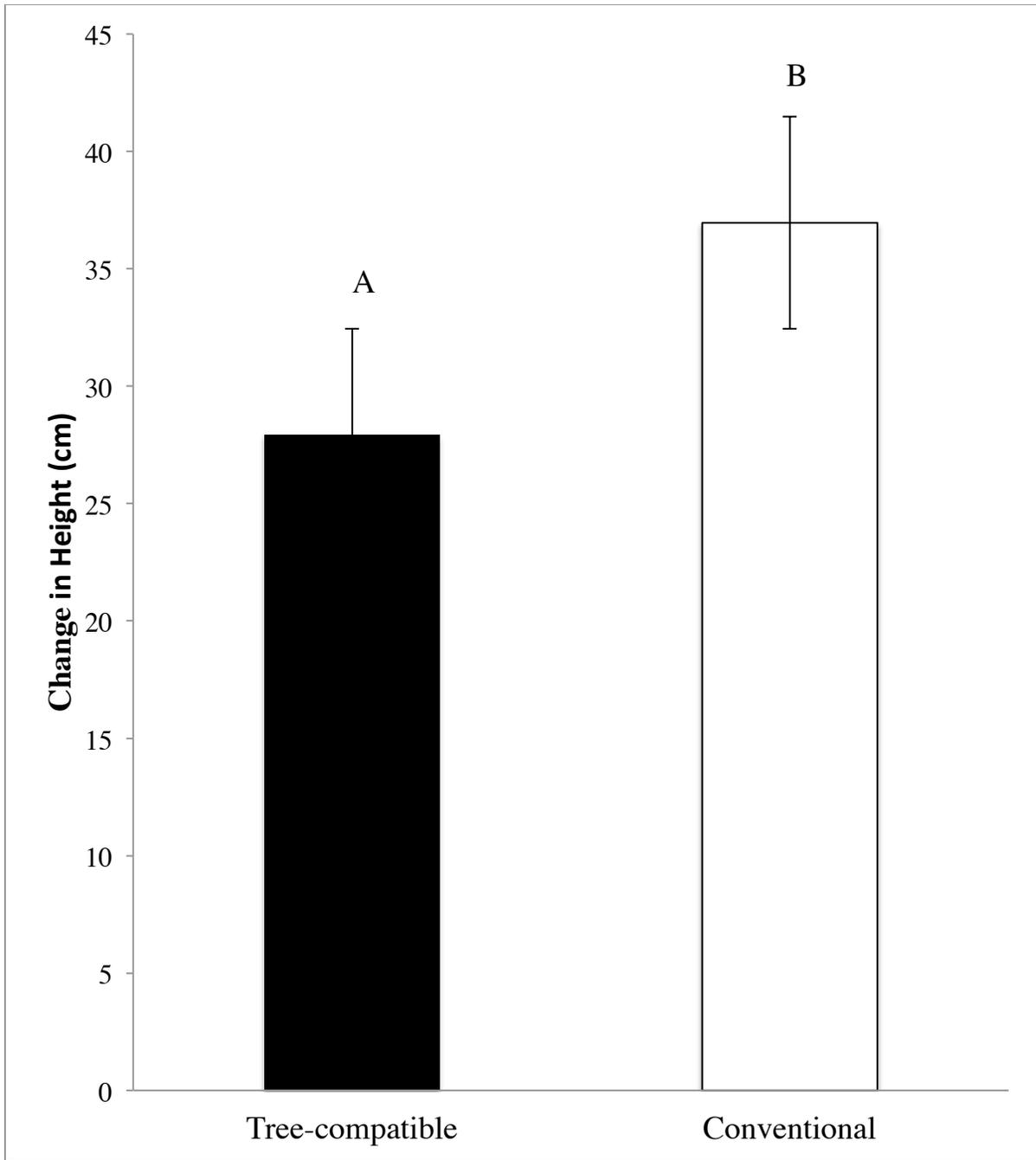


Figure 5. Mean change in height of autumn olive by vegetation mix with standard error.

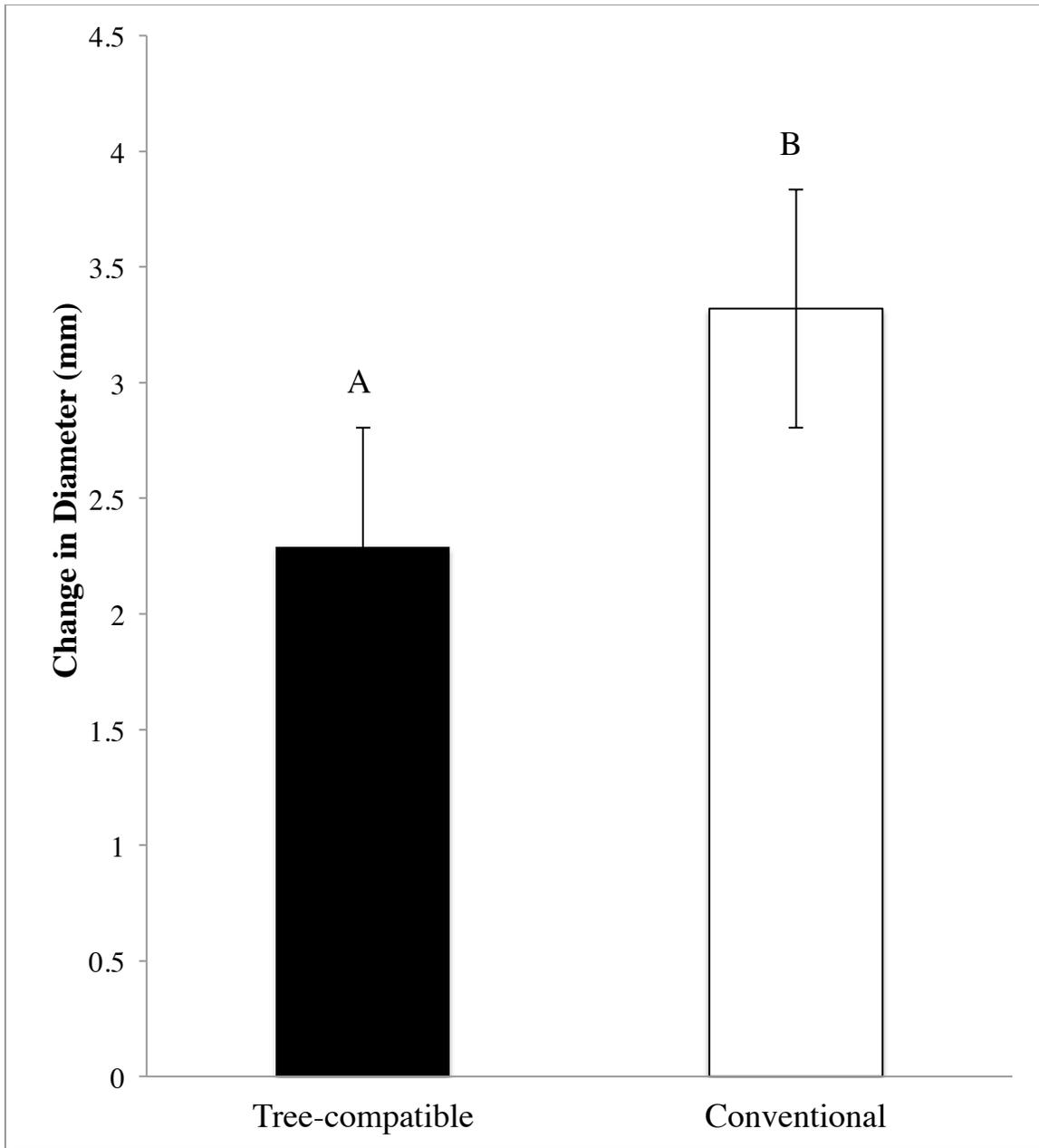


Figure 6. Mean change in basal diameter of autumn olive by vegetation mix with standard error.

Chapter 3

Management of autumn olive and the effects on hardwood tree establishment

Abstract

The Appalachian region of the United States is the most biodiverse region in non-tropical areas and home to the largest temperate deciduous forest in the world, providing many ecosystem services and economic benefits. However, surface coalmining has created significant disturbance, resulting in fragmentation and a net loss of productive forests. The Forestry Reclamation Approach (FRA), used to reclaim post-mining landscapes, seeks to achieve hardwood tree canopy cover following the establishment of a “tree-compatible ground cover” seed mixture. However, this approach leaves land exposed, making reclaimed sites susceptible to invasion by exotic species. Autumn olive (*Elaeagnus umbellata*) is a non-native, nitrogen-fixing shrub that is commonly viewed as a major obstacle to reclamation success. To better understand the impact autumn olive has on hardwood establishment, we managed autumn olive using mechanical and combination mechanical/chemical controls. In each treatment, we measured plant available nitrate (NO_3^-) and ammonium (NH_4^+) using ionic exchange membranes (IEMs). Three native tree species were planted in the winter 2015, and we followed their survival and performance. At the end of the first growing season, native tree survival was high, and the presence or absence of autumn olive had little effect on tree survival or growth, despite the higher plant available nitrate where autumn olive was present.

Introduction

The Appalachian region of the US is home to the largest temperate deciduous forest, and some of the most biologically diverse forest systems in non-tropical regions (Ricketts *et al.* 1999). The forests in the Appalachian region, extending from northern Mississippi to southern New York, provide numerous ecosystem services, such as water quality protection, carbon sequestration, and wildlife habitat (Burger *et al.* 2009), as well as many economic benefits (e.g. forest industry) (Zipper *et al.* 2013). Surface mining, however, has contributed a net loss of productive forests and fragmentation (Wickham *et al.* 2007; Saylor 2008), and has resulted in a significant loss of forests through large-scale disturbances (Drummond & Loveland 2010). Coal mining in particular has affected >600,000 hectares in Appalachia, and the majority of this mined land has not been returned to forests or any other type of productive land use (Zipper *et al.* 2011b), resulting in unmanaged land often invaded by non-native plants (Burger *et al.* 2013).

Severe erosion, sedimentation, and landslides were a significant issue for coal surface mines prior to 1977 (Zipper *et al.* 2011b). As a result, US Congress passed the Surface Mining Control and Reclamation Act (SMCRA) to standardize and mandate reclamation practices (SMCRA, Public Law 95-87). SMCRA required coal mining companies to obtain a permit and pay a bond prior to mining to ensure land reclamation adequate to restore its original use or a use of higher value [Sec. 515(b)(2)]. In Appalachia, the easiest way to meet SMCRA requirements is through reclamation of native hardwood forest (Holl, Zipper & Burger 2009) or pasture.

However, under SMCRA, reclamation assessment occurs only five years after reclamation has been completed (Zipper *et al.* 2011b). Historically, this resulted in many projects focusing less on long-term goals, such as the return of ecosystem services, and

more on short-term mitigation of erosion and minimizing acid mine drainage (Lemke *et al.* 2013). In an attempt to bridge the long-term and short-term goals of SMCRA, the Appalachian Regional Reforestation Initiative (ARRI) formed in 2004. ARRI promotes the Forestry Reclamation Approach (FRA), which is intended to help restore high quality, native hardwood forests on reclaimed coalmines through the creation of suitable site conditions for tree survival and growth as well as the colonization by native species (Zipper *et al.* 2011b). In recent years, landowners and mine operators have been embracing the FRA because it improves the reclaimed forest through the development of better ecosystem structure (Burger *et al.* 2009). However, FRA reclamation sites result in more bareground, which leaves the land open not only to colonization by the desired native species, but also vulnerable to invasion by exotic plants (Fields-Johnson *et al.* 2012). Invasive plants have become a primary concern for those responsible for reclamation as these exotic plants have become widespread and are implicated in a variety of negative ecosystem impacts, including impediments to reclamation success (Barney *et al.* 2013).

On former coalmines in the Appalachian region, autumn olive (*Elaeagnus umbellata*) is one of the most common invasive species (Zipper *et al.* 2011a), and it has become a major invader of reclaimed sites across most of the region (Lemke *et al.* 2013). Autumn olive was brought to the US in 1830, and is native to Pakistan, China, and Eastern Asia (Ahmad, Sabir & Zubair 2006). In the 1960s, it was intentionally planted for erosion control, as a nurse species for tree plantations, and provide habitat and food for wildlife on disturbed lands, (Fowler & Fowler 1987; Lemke *et al.* 2013). This resulted in autumn olive being planted widely on reclaimed mine sites (Zipper *et al.* 2011b).

Autumn olive has many traits that contribute to its success on reclaimed mine sites, including the ability to fix atmospheric nitrogen, produce numerous drupes, and grow in acidic, loamy soils (Ahmad, Sabir & Zubair 2006). Autumn olive has the potential to produce multiple stems from the main root (Moore *et al.* 2013) and quickly forms a broad, dense crown (Evans *et al.* 2013) which suppresses native species through intense shade (Lemke *et al.* 2013) and formation of dense patches (Catling *et al.* 1997).

In a recent reclamation experiment, researchers found that in the fourth growing season, volunteer species represented 65% of the total growth on the site (Evans *et al.* 2013). In that study, autumn olive produced greater volume than native trees and covered 27% of the entire study site. If species such as autumn olive are not managed, there is evidence to suggest that it will interfere or prevent successful restoration (Evans *et al.* 2013). Due to its rapid growth on disturbed lands and dispersal capabilities, autumn olive is considered a pioneer species with the potential to alter the successional trajectory of reclaimed landscapes. In order to achieve the desired post mining land use, management is required to control autumn olive, and mine operators must determine ways to eradicate autumn olive to achieve bond release (Lemke *et al.* 2013). Autumn olive is commonly managed on reclamation sites using a variety of techniques, but no information exists regarding management efficacy or the subsequent effects on planted native trees.

Here we investigated how the management of autumn olive affects the establishment and growth of planted hardwood tree seedlings. Specifically, we compared hardwood seedling growth in reclaimed areas where there was autumn olive present, reclaimed areas where autumn olive was not present, and reclaimed areas where we managed autumn olive in two different ways (i.e., mechanical removal and mechanical

removal plus cut stump herbicide). This study will inform both management and reclamation practices, as well as basic ecological interactions among invasive and native woody plants.

Materials and Methods

The following experiments were carried out at the Powell River Project (PRP), a 445 hectare area used cooperatively by Virginia Tech, natural resource industries, and other educational institutions to conduct research that enhances Appalachian restoration techniques and water protection. The PRP is located on a mountaintop removal mine in Wise County, Virginia, and has a long history of research addressing environmental protection and restoration since its beginning in 1980.

To meet our objectives, we conducted an experiment to record hardwood tree seedling growth on reclaimed land in four different autumn olive control treatments. We identified a reclamation site that was heavily colonized by mature autumn olive, but also had adjacent large areas where there was no history of autumn olive. We set up the following four treatments in the fall of 2014 in a randomized complete block design with 8 total blocks:

- 1) autumn olive present;
- 2) mechanical control of autumn olive;
- 3) mechanical control of autumn olive followed by cut-stump herbicide application;
- 4) autumn olive never present,

To simulate how autumn olive would be managed on a large scale, we cut autumn olive with a chainsaw at 10-15cm above the soil surface for treatments 2 and 3. For treatment 3, we also applied the herbicide Garlon 4 Ultra (active ingredient triclopyr) at 50% v/v in

basal oil. Each plot size was approximately 3 x 3m depending on the size of the autumn olive, but in each case 2-4 mature individuals were cut for treatments 2 and 3.

Reclamation specialists typically hand transplant bare rootstock seedlings in late winter. Therefore, we used bareroot seedlings of pin oak (*Quercus palustris*), red maple (*Acer rubrum*), and black cherry (*Prunus seritona*), which were chosen for their local use and rapid growth rate on reclaimed mine sites (Davis *et al.* 2012). Three individuals of each species were randomly planted into each treatment at each block in mid-March 2015. Within each plot we monitored hardwood survival, hardwood height and basal diameter, and autumn olive regrowth (for treatments 2 and 3) from February to October 2015.

Autumn olive is a nitrogen fixer and has the potential to alter nutrient cycles, which may affect tree growth differently across treatments. Therefore, in each plot, ionic exchange membranes (IEMs) (GE Osmonics, Inc., Trevose, PA) were used to quantify NO_3^- and NH_4^+ (Subler, Blair & Edwards 1995; Bowatte *et al.* 2008; Duran *et al.* 2013). All IEMs were cut into 5 x 10 cm rectangles, and hole-punched at the top. IEMs were immersed in a 1 M solution of sodium chloride (NaCl), which allows either sodium (Na^+) or chloride (Cl^-) ions to fill all exchange sites. Anion and cation membranes were stored in the 1 M NaCl at 4°C in separate containers until put into the field. Before being placed into the field, membranes were rinsed with DI H₂O. A pair of IEMs (anion and cation) were placed into each management plot at each block and were installed at a 45° angle in the soil, ensuring no overlaps or wrinkles. After 30 days in the field during the month of August, each membrane was placed into its own plastic bag and transported on ice back to the lab, and then stored at 4°C for <7 days. For the extraction of inorganic-N, soil

particles were rinsed off of the membrane surface using DI H₂O and then individually submerged in 1 M potassium chloride (KCl). They were then placed for 1 hour on a reciprocal shaker at 22 rev min⁻¹ (Subler, Blair & Edwards 1995; Hangs, Greer & Sulewski 2004). The analysis of extracts for NO₃⁻-N and NH₄⁺-N occurred on a TrAAcs 2000 Analytical Console (Bran + Luebbe, Analyser Division, Norderstedt, Germany) which was connected to an XY2 Auto sampler (SEAL Analytical, Mequan, Wisconsin).

To test for treatment effects on hardwood tree seedling survival, we conducted a logistic regression. To test for treatment effects on change in height and change in basal diameter of hardwood tree seedlings, we conducted a two-way analysis of variance (ANOVA) with tree species and treatment as fixed effects. A one-way ANOVA was conducted to test for the effect of autumn olive control treatments on quantities of nitrate (NO₃⁻) and ammonium (NH₄⁺). Differences in change in height and change in diameter among treatments and species, as well as differences in the amount of NO₃⁻ and NH₄⁺ among treatments, were determined using Tukey-Kramer HSD. Statistical analyses were conducted using JMP Pro 11.

Results

Survival

Survival was high (>80%), and did not differ among the three native tree species (p=0.1685) or the interaction between species and treatments (p=0.3155), but survival did vary among the treatments (p=0.0392). Hardwood tree seedlings had the highest survival in the autumn olive cut/spray management plots (p=0.0392, Figure 1).

Hardwood Growth

Autumn olive management produced no significant differences in tree seedling height or basal diameter growth ($p=0.1788$; $p=0.4393$), nor were there significant differences in the interactions between autumn olive management and species height growth or basal diameter growth ($p=0.5054$; $p=0.4807$). However, during peak growing season (August), there was a significant difference between autumn olive management treatments, with the seedlings in autumn olive plots on average growing more (40 cm) than seedlings in the plots without autumn olive present (34 cm) ($p=0.0187$, Figure 2).

Across all treatments and tree species, new growth was the same, 4.4 cm ($p=0.3460$). Pin oak seedlings had almost four times as much basal diameter growth across all treatments than did red maple seedlings, and almost double the amount of basal diameter growth than cherry seedlings ($p<0.0001$, Figure 3).

IEMs

The amount of plant available nitrate (NO_3^-) was over two times greater where autumn olive was still intact than in plots where there was no autumn olive ($p=0.0202$, Figure 4). Removing autumn olive had no impact on nitrate availability. There was no significant difference among autumn olive management plots in the amount of ammonium (NH_4^+) available (0.9351).

Discussion

Overall, native tree survival was high, and the presence or absence of autumn olive had little effect on tree survival or growth. Despite higher plant available nitrate levels when autumn olive is present, in this study the trees did not appear to be affected

by increased nutrient availability. The lack of overall differences most likely reflects the short duration of the study, and we would expect differences, if they did exist, to manifest with more time as tree growth is relatively slow.

While hardwood seedling survival was highest in the autumn olive management plots where autumn olive had been cut and sprayed, there were no differences in seedling growth among the autumn olive management plots at the end of the growing season. This was somewhat surprising, considering the almost double amount of nitrate (NO_3^-) available in the management plots with standing autumn olive compared to the plots without autumn olive present.

Nitrogen (N) is one of the most important components of ecosystems worldwide, influencing function (e.g., nutrient cycling) and structure (e.g., diversity) (Vitousek *et al.* 2002). Disturbance can often lead to pulses in plant nutrients, which can lead to invasion and change successional trajectories (Davis, Grime & Thompson 2000). However, in highly disturbed sites such as coalmines, which commonly use fractured bedrock as the growing medium, nitrogen and other important plant nutrients are lacking. In many cases, disturbed nutrient-poor systems are commonly colonized by nitrogen-fixing exotic plants (Vitousek *et al.* 1987; Vitousek & Walker 1989).

Autumn olive creates root nodules that have a symbiosis with actinomycetes (*Frankia* spp.) in the soil, giving them the ability to fix and utilize atmospheric nitrogen (Paschke, Dawson & David 1989; Kim, Cardenas & Chennupati 1993). This symbiosis, combined with rapid growth rates and endozoochory have allowed autumn olive to thrive on degraded lands. In fact, there is evidence that underneath autumn olive, there are higher nitrification rates, and nitrate leaching underneath autumn olive is comparable to a

fertilized agricultural field (Baer *et al.* 2006). Therefore, we expected the tree seedlings to grow better in plots with autumn olive, managed or unmanaged, than in the plots where there is no autumn olive present due to the increased concentration of plant available nitrogen. Instead, we observed no influence of the varying amounts of NO_3^- concentration on tree seedling growth at the end of the growing season (October) among autumn olive management plots. While nitrogen availability did not appear to affect tree growth, our study may have been too brief to observe differences among relatively slow growing trees, or competition from other species could be mediating the impact of nitrogen. In the first year, the tree seedlings are spending resources establishing a root system, which may not be fully developed enough to exploit differences in plant available nutrients.

Across all treatments, the pin oaks had more growth in basal diameter than the black cherry or red maple over the course of the first growing season. Anecdotally, we noticed poor drainage and standing water at some of our experimental blocks after rainfall. Even though red maple, black cherry, and pin oak are all woody species with rapid growth rates and ability to tolerate varying pH levels (Davis *et al.* 2012), pin oaks are better suited for wet sites than red maples and black cherries (Davis *et al.* 2012). Though we did not take data pertaining to the standing water we observed in some areas, this could help to explain the larger basal diameter growth of pin oaks than the other two species.

Once autumn olive becomes established in an area, eradication requires tremendous effort and expense. In fact, for most invasive species, eradication, or the complete elimination of the species in the area, grows proportionally more difficult with the size of the invasion (Rejmánek & Pitcairn 2002). The difficulties of autumn olive

eradication stem from prolific seed production, dispersal via wildlife, and the ability to re-sprout after cutting or damage (Kohri *et al.* 2002), resulting in aggressive colonization which hinders native woody species dispersal on reclaimed mine areas (Zipper *et al.* 2011b). Often, a single mechanical removal is used to control dense groves of autumn olive on abandoned mines that are hindering the establishment and growth of hardwood tree species (Byrd *et al.* 2012). However, mature autumn olive is known to aggressively resprout when cut (Campbell, Dawson & Gregory 1989), thus requiring an additional chemical component.

Autumn olive also has the potential to impact ecosystem processes, through nitrogen fixation and alteration of nutrient cycling (Schlesinger & Williams 1984), giving it the potential to impact the long-term development of forests (Moore *et al.* 2013). Successful management of autumn olive with all techniques requires continual management (Byrd *et al.* 2012). Despite the known effects of autumn olive, our results have not shown autumn olive to be detrimental to the success of tree seedlings, at least in the first year. However, it is important to maintain this experiment to continue following the effects autumn olive management (or lack thereof) may have on hardwood tree seedlings as they mature, further our understanding of forest succession on reclaimed mine sites.

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Figures

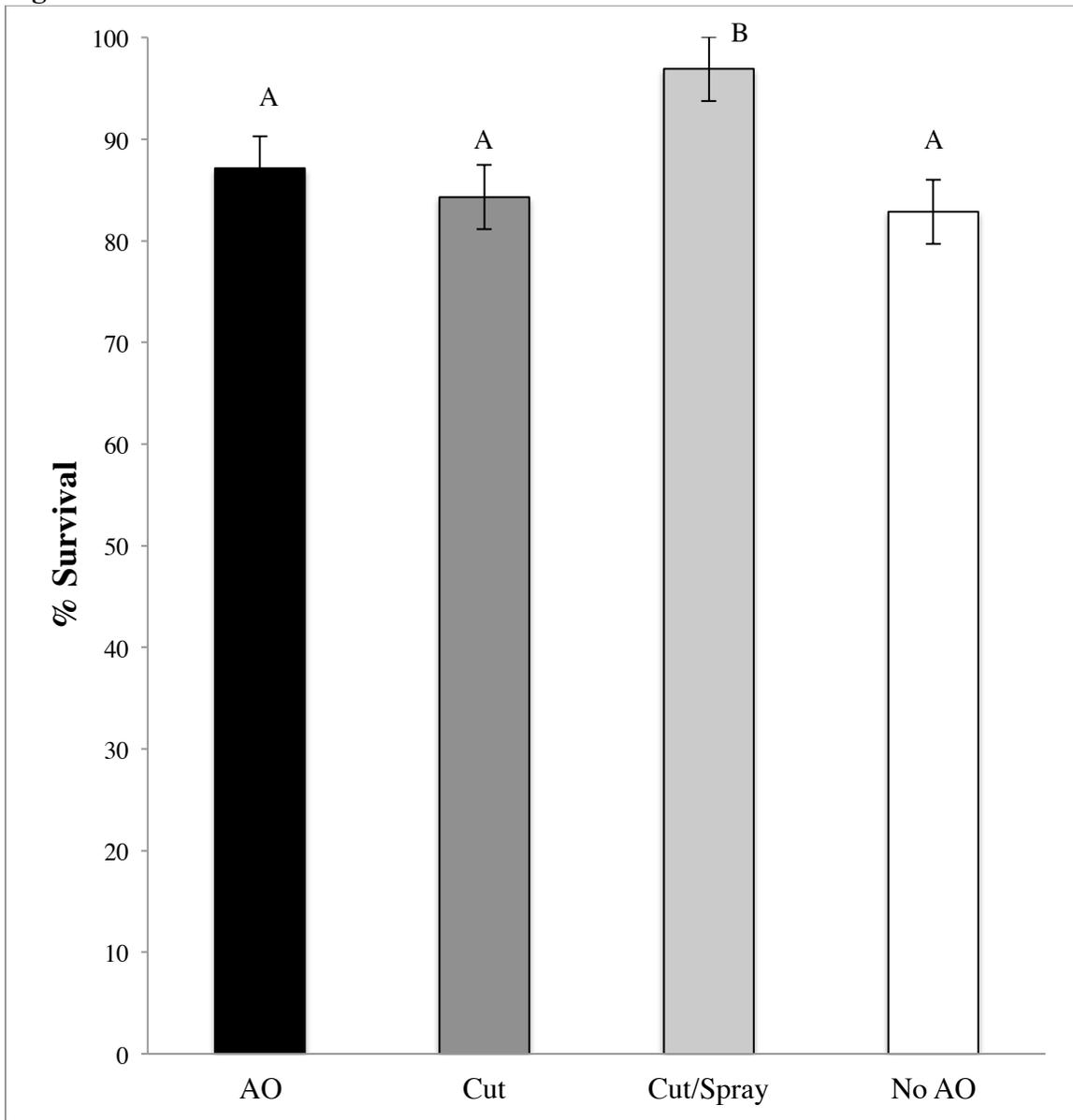


Figure 1. Percent survival of hardwood tree seedlings by management plots with standard error.

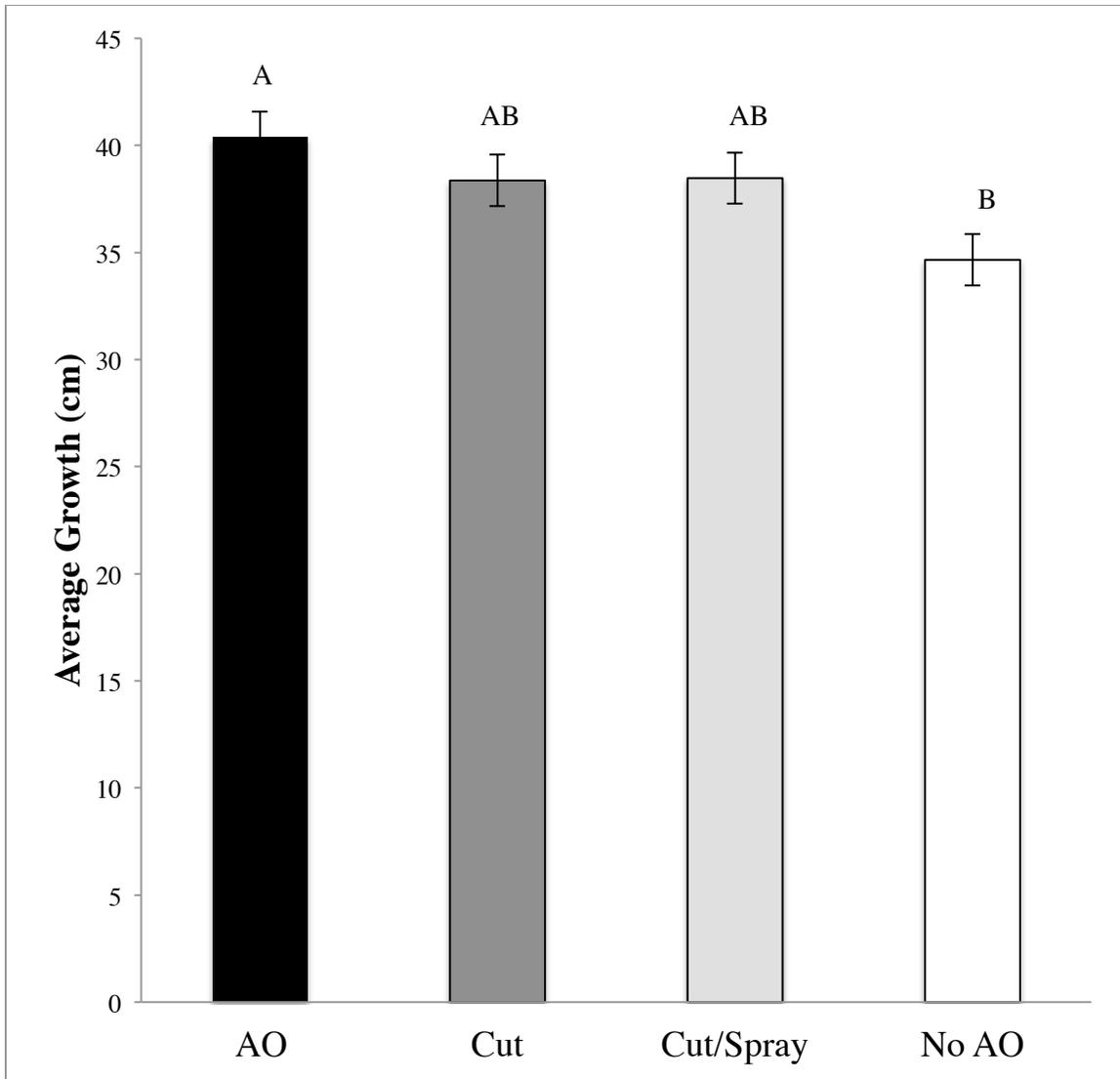


Figure 2. Mean change in height of planted tree seedlings by treatment with standard error.

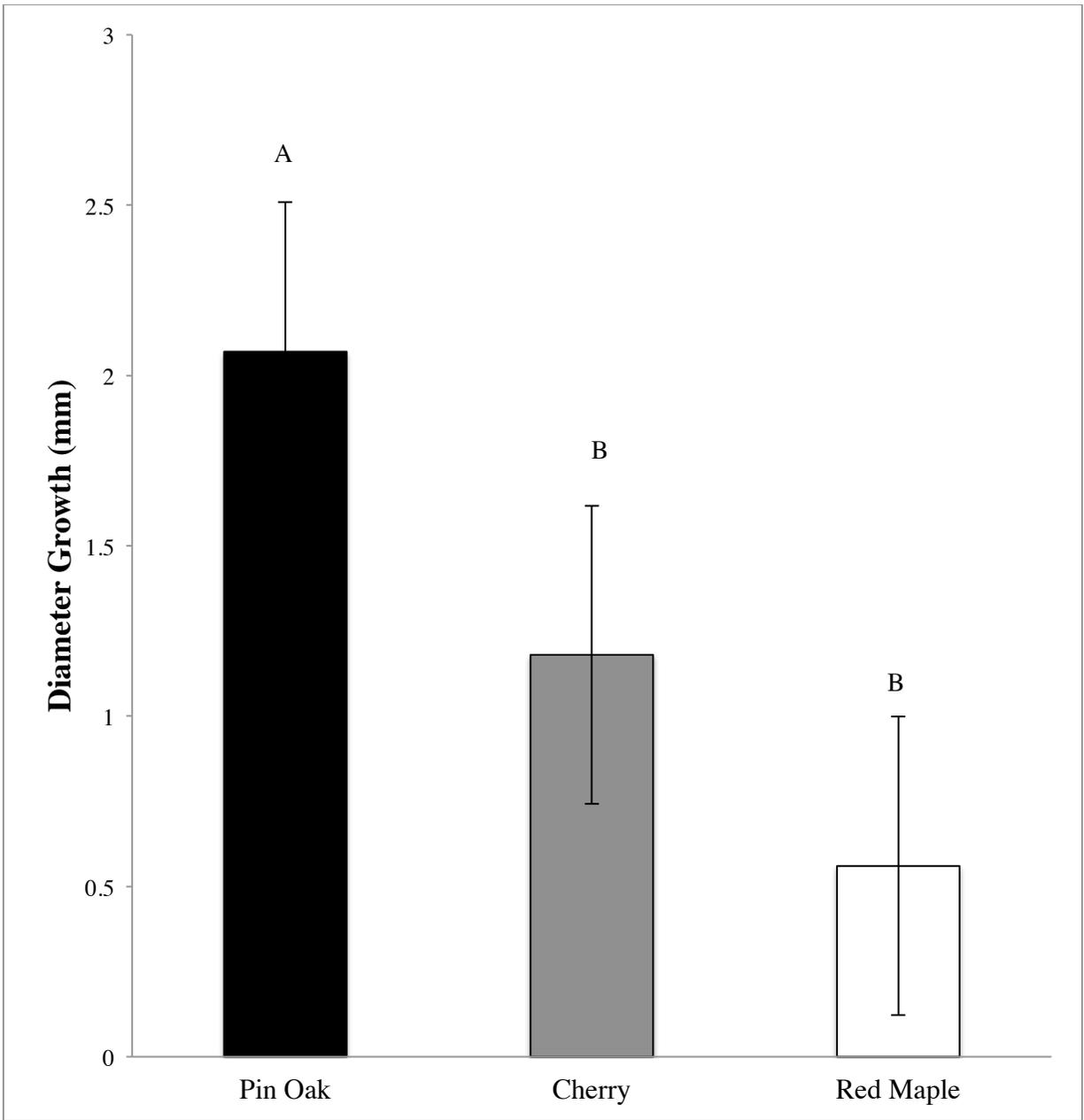


Figure 3. Mean change in basal diameter of tree seedlings by species with standard error.

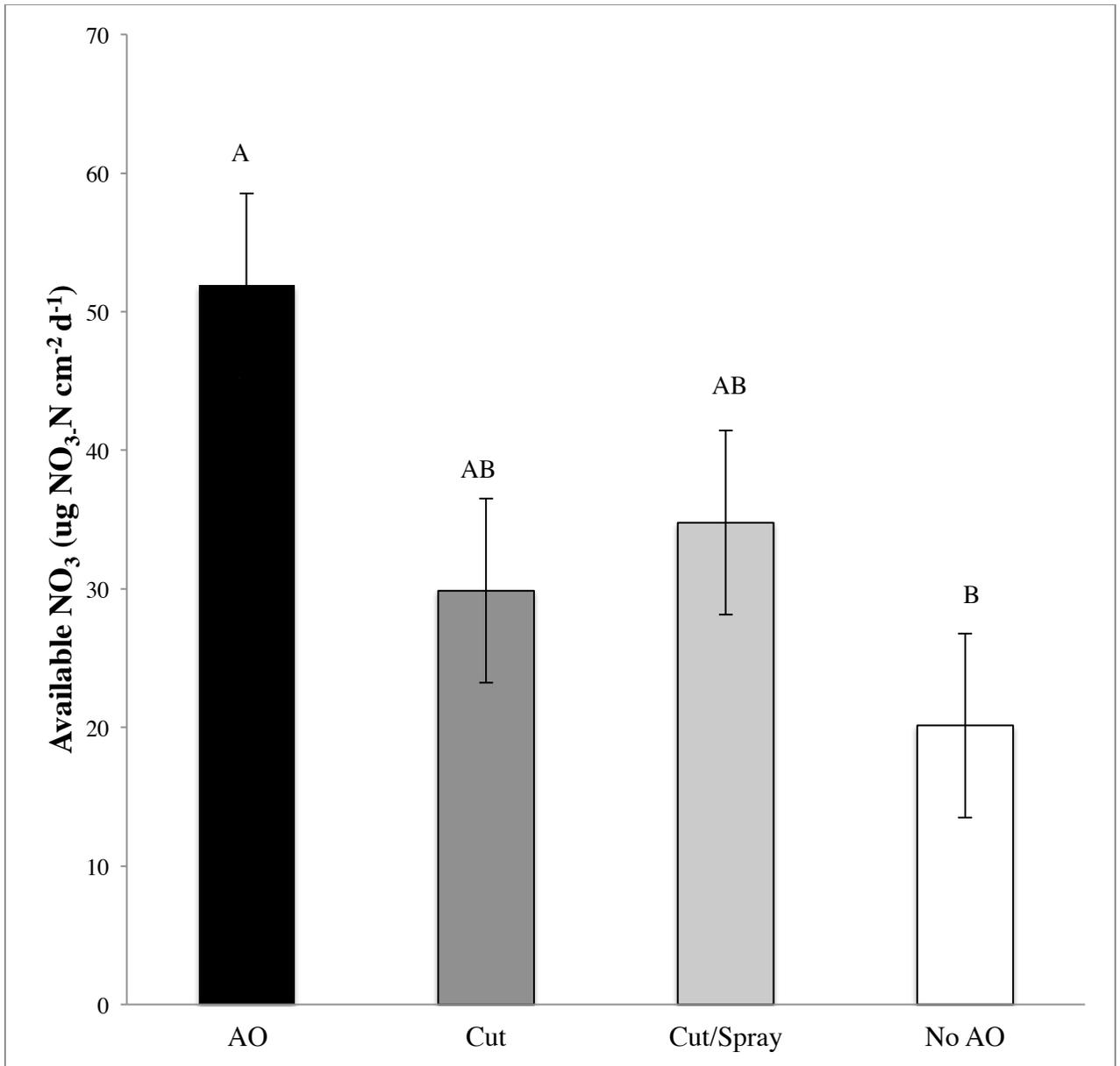


Figure 4. Nitrate (NO_3^-) availability ($\text{mg N cm}^{-2} \text{d}^{-1}$) within each management plot with standard error.

Chapter 4

The Appalachian region of the United States is home to some of the most biologically diverse areas in the non-tropical world, but has undergone significant disturbance, as well as forest fragmentation and loss, due to the prevalence of coal surface mining. Since most of the land was left altered and unmanaged, a majority of these areas have been invaded by non-native species. One of the most prevalent and problematic invaders on reclaimed sites is autumn olive (*Elaeagnus umbellata*), which has been viewed as one of the main hindrances to forest reclamation. However, little empirical research exists specifically looking at the impacts of autumn olive on forest success. This research is one of the first steps at trying to better understand what reclamation conditions (substrate and vegetation mix) influence the performance of autumn olive, and what kind of impact autumn olive could be having on the growth and success of hardwood tree seedlings.

While only having one year of data to examine, there are still some points of discussion and directions for future research. One of the more interesting and encouraging results was the decreased growth of autumn olive in the tree-compatible vegetation mix compared to the conventional mix. The tree-compatible mix is composed of species intentionally selected for their slower growth and lower competitive ability on these sites in order to allow spaces for native plant colonization. Thus, the fact that autumn olive actually grew significantly better in the conventional mix (composed of more aggressive, fast-growing species) than the tree-compatible mix could suggest that once these mixes have been able to establish and have time to allow for native colonization to occur, the tree-compatible mix could actually be better at resisting the rapid growth and invasion of autumn olive compared to other seeding mixes. With the

continuation of these experiments, we can speak to these ideas more conclusively, but for now, this is exhibiting one more advantage of the use of the tree-compatible mix being favored by reclamation practitioners who developed the Forestry Reclamation Approach.

Regarding the impact autumn olive could be having on native hardwood tree seedlings and their success on these sites, we did not find any evidence that different management techniques have an impact on hardwood seedlings growth, at least not after the first growing season. On the other hand, we also did not see any evidence that, because autumn olive is a nitrogen fixer, it could be acting as a nurse plant for these hardwood seedlings and facilitating a higher growth rate for these seedlings. However, we did see slight suppression of autumn olive on hardwood survival, due to the fact that we had a significantly higher hardwood survival rate in the plots where we cut and sprayed autumn olive (effectively killing it) compared to any of the other treatments. Continuing to follow this experiment will be vital to better understanding and having quantitative evidence of the impact autumn olive can have on hardwood tree seedlings success on these reclaimed mine sites.