A CHRONOSEQUENCE OF SCALE-DEPENDENT VEGETATION AND SOIL PROPERTIES ON A SURFACE COAL MINE OVER 40 YEARS OF RECLAMATION

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Stefanie Lynn Bohrer

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The Supervisory Committee certifies that this disquisition complies with North Dakota State University's regulations and meets the accepted standards for the degree of

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SUPERVISORY COMMITTEE:

Ryan Limb
Chair

Aaron Daigh

Abbey Wick

Jay Volk

Approved:

April 11, 2016
Date

Francis Casey
Department Chair

ABSTRACT

Surface coal mining has taken place in North Dakota for many decades. Upon the mining process, the mined lands need to be reclaimed to a better state than pre-mining. The reclamation process is a timely and costly procedure. Currently, most reclamation strategies focus only on above ground biomass. Our research entailed two different studies, the first looking into vegetative species composition and canopy cover of reclaimed mine lands, and the second focuses on belowground properties affected by soil compaction over a 40 year reclamation gradient. Species composition and canopy cover did not increase over 40 years ($p > 0.05$). Soil compaction did not decrease, and rooting depths and soil water content range did not increase over the reclamation gradient ($p > 0.05$). Relative plant community patch size and soil health on reclaimed lands over four decades indicate the landscape-level success of the current ecosystem-based reclamation strategy.

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DEDICATION

I dedicate this work to my Grandpa Howard. No one has had more influence on the person I have become than you. You taught me life-long lessons such as responsibility, respect, hard work, take pride in your own work, and follow your dreams that I will cherish and carry with me for the rest of my life. I hope I can instill your work ethic into my life and career and continue to make you proud. Grandpa you will forever be in my heart. ♡

PREFACE

Chapter one is written as a general literature review for both chapters two and three. Chapters two and three are written as independent manuscripts to be submitted to peer-reviewed journals. Chapter two, “Small and large-scale patterns of diversity on reclaimed prairie over a 40-year chronosequence” is written following the style and formatting guidelines *Environmental Management Journal*. Chapter three “A 40 Year Chronosequence of Soil Properties on Reclaimed Surface Coal Mine Lands” is written following the style and formatted for the *Journal of Land Degradation and Development.*

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CHAPTER 1: INTRODUCTION AND LITERATURE REVIEW

Rangeland Disturbance and Species Diversity Indices

Historical Disturbance

Grassland dominance globally began during the end of the Cenozoic era (65 million years ago) (Retallack 2001). Expansion of the grassland biome in North America occurred in the Miocene-Pleistocene transition (7–5 million years ago) and was associated with an increase in herbivores such as camels, horses, rhinoceroses, antelopes, bison and elephants (Axelrod 1985). The climate during the Pleistocene (2.6-.01 million years ago) continued to provide the environmental conditions needed to promote grasslands, fire, and herbivores (Saur 1950). The Pleistocene-Holocene transition (0.01 million years ago) was a time of important change in climate and anthropogenic influences that coincided with glacial retreat in North America. Following glacial retreat, stabilization of the prairie vegetation was largely due to disturbances such as anthropogenic fire, occasional lightning strikes, and herbivory (Curtis 1971; Anderson 1990, 1998, 2006). The Great Plains vegetation, particularly the central grasslands, is relatively new in terms of geologic time. Currently, only remnants of the previously diverse fauna (bison, elk, and deer) remain (Flores 1996).

Disturbance is a temporary change in an environmental condition that causes a pronounced change in an ecosystem process (Hobbs & Huenneke 1992). Typically, disturbance removes biomass and quickly changes the previous environment as a result of wind and water erosion, mega fauna trampling, fire, grazing, and many other abiotic and biotic factors. Grassland plant communities are dependent on disturbance (Pickett & White 1985; Hobbs & Huenneke 1992) because it initiates succession through altered light and nutrient availability (Hobbs & Huenneke 1992). Alteration of the historical grazing and fire disturbance regime will be detrimental to the health of the ecosystem.

Disturbance regimes influence vegetation composition by modifying the abundance of existing species and providing establishment opportunities for other species with different resource requirements (Li et al. 2006; McIntrye et al.1995; Moloney & Levin 1996). The spatial scale of disturbance plays an important role in impeding the process of vegetation succession following disturbance by increasing sunlight, nutrient availability, and distribution area for recruitment. Attention needs to be given when making inferences from small-scale observations to the large scale management of vegetation because disturbances may have severe impacts on ecosystems that cross successional thresholds, preventing recovery of the ecosystem (i.e fire, grazing and fossorial animals) (Li et al. 2006). This role of scale gives rise to the health of ecosystems by enhancing heterogeneous diversity on the landscape.

Natural Disturbance on Rangelands

Rangelands are heterogeneous because of their high diversity composition and productivity across multiple scales (Ludwig & Tongway 1995; Pattern & Ellis 1995; Fuhlendorf & Smeins 1999; Fuhlendorf & Engle 2001). Heterogeneity is the variability in vegetation stature, composition, density and biomass resulting from different timings of disturbances, topographic and edaphic patterns, competitive interactions, and succession among vegetative patches (Kola & Pickett 1991; Fuhlendorf & Engle 2001). Heterogeneity may be the cause of biological diversity (Fuhlendorf & Engle 2004; Christensen 1997; Ostfeld et al. 1997; Weins 1997), influencing species diversity, wildlife habitat, and ecosystem function (Christensen 1997; Ostfeld et al. 1997; Wiens 1974; Fuhlendorf & Engle 1998; Fuhlendorf & Engle 2001). Therefore, rangeland management should focus more on restoring heterogeneity within the landscape than on restoring the previous successional composition of grasslands (Fuhlendorf & Engle 1998).

Fire is recognized as an important component of a historical disturbance regime (Wright & Bailey 1980) because it restricts woody encroachment and promotes grass dominance in the Great Plains (Wight 1877; Axelrod 1985; Anderson 2006). Climate, along with fire, is highly responsible for the continued success of grass species; large herbivores aid to promote the continued expansion of grasses by trampling seedlings and destroying older trees (Axelrod 1985).

Plant communities in the Northern Great Plains (NGP) have a long evolutionary history with herbivores (Milchunas et al. 1988). Grazing animals react to their environment through a hierarchy of natural responses and behavioral actions causing variable distributions at the landscape, community, and patch level (Senft et al. 1987; Stuth 1991). Grazing can be sustainable on most grasslands, but overgrazing at focal grazing points can create lawns which facilitate ecosystem degradation (Martin & Ward 1970; Foran & Bastin 1985; Fuls 1992; Fuhlendorf & Engle 2001; Landsberg et al. 2003; Tobler et al. 2003). Consequently, grazing management frequently promotes uniform herbivory with even distribution of grazing animals at moderate levels, therefore, no areas are heavily grazed or non-grazed

(Holechek et al. 2003; Fuhlendorf & Engle 2004). Nested within the larger matrix of fire and grazing disturbance, focal soil disturbance by fossorial animals is another disturbance regime important to the health of grassland ecosystems.

Burrowing animals often act as disturbance agents (Sousa 1984; Pickett & White 1985) as their actions destroy or displace mature plants, modify resource availability, and the physical environment. Consequently, these modifications allow new individual plant species to establish. Excavation activities bring soil to the surface, re-distribute nutrients, aerate the soil, alter soil moisture (Eldridge 2004; Eldridge & Whitford 2009), and alter local plant community composition (Borchard & Eldridge 2012). Fossorial animals can modify community-level characteristics, such as species diversity, if plant populations are differentially prone to the created disturbances. The manner in which co-occurring animal taxa interact to alter vegetation is a function of their respective behaviors that shape the characteristics of disturbances (Lynn & Detling 2008). With these natural disturbances, native prairies are host to diverse vegetative communities.

Alpha and Beta Diversity

Alpha diversity can be defined as “the number of species present at a single site or sampling plot” (Cingolani et al. 2010). Alpha diversity consists of two components: species richness and species evenness (Stirling & Wilsey 2001). Species richness is the number of species present, and evenness is how species abundance is distributed (Cingolani et al 2010). High species evenness can increase total productivity, resistance of invasive species, and can reduce plant loss (Wilsey & Potvin 2000; Wilsey & Polley 2002, 2004; Smith et al. 2004). Proper alpha diversity represents species recruitment and stability on disturbed sites, enhances the development of the plant community, creates a self-sustaining ecosystems, and measures packing within a community. This reflects how species divide ecological resources (Sepkoski 1988; Alday et al. 2011).

A second component of species diversity is beta diversity, which is the variation in species composition among sites, or how species numbers and identities differ between communities (Magurran 1988; Wilsey et al. 2005; Martin et al. 2012). Beta diversity may arise when a disturbance changes the vegetative community, creating dissimilar patches (Martin & Wilsey 2012). Due to anthropogenic disturbances assisting the establishment of invasive species, creating beta diversity may be unattainable in restorations (Martin & Wilsey 2012). Invasive species often become troublesome in highly disturbed areas due to their ability to colonize and form monocultures (Pritekel et al. 2006).

Anthropogenic Alterations of Historic Disturbances

Novel ecosystems result when species combinations, species abundances, and changes in ecosystem functioning that have not historically occurred within a biome (Hobbs et al. 2006). Anthropogenic disturbances often result in different disturbance regimes (e.g. changes in fire, grazing and fossorial animal distributions) that can play a significant role in shaping alpha and beta diversity (Kirkpatrick 1994; Conacher & Conacher 1995). With these different disturbance regimes, it can be difficult to fully achieve proper native diversity within ecosystems due to the lack of resistance to exotic species invasion (Grubb 1977; Li et al. 2006; McIntyre et al. 1995; Moloney & Levin 1996).

Indigenous Tribes were the first to use fire on grasslands as a management tool to encourage plant regrowth, reduce woody species, manipulate prey species, and increase travelling ease (Stewart 1956; Curtis 1971; Pyne 1983, 1997; Anderson 1990, 1997; Bragg 1995). A reduction in anthropogenic fire occurred around 1875, and is currently inconsistently used to manage native prairies. Recently, prescribed burning has increased due to interest in restoring historic disturbances regimes (Brockway et al. 2002) and applications for wildlife habitat management (Augustine et al. 2007, 2010; Thompson et al., 2008). However, the use of fire is limited by habitat fragmentation from agriculture and urban development (Higgins 1986; Anderson & Bowles 1999; Anderson 2006).

Historically, large, grazing herbivores covered the prairies of the NGP. Yet, after European settlement, widespread livestock grazing became prominent on native ecosystems of western North America (Wagner 1978; Crumpacker 1984; Fleischner 1994). Most rangeland management practices were developed to increase livestock production and promote forage species dominance, reducing landscape heterogeneity (Fuhlendorf & Engle 1998). Rangeland management traditionally promotes homogeneity by even distribution of livestock grazing across the landscape (Fuhlendorf & Engle 1998). Traditional grazing management creates uniform use of plants with some areas resulting in overgrazing and a decrease in landscape heterogeneity (Fuhlendorf & Engle 1998). Overgrazing was a common practice in early European settlement, which lead to degradation of rangelands (van der Westhuizen et al.

2005; Liang et al. 2009; Sch€onbach et al. 2011). Severe livestock grazing makes it difficult for other natural disturbance regimes such as fossorial animals to compete and survive in rangelands.

Burrowing animal habitat is 90% reduced due to agricultural practices and urban development during the past 200 years (Anderson et al. 1986; Strapp 1998; Miller & Cully 2001; Vermeire et al. 2004; Ramirez & Keller 2010). As habitat becomes increasingly fragmented, burrowing animal colonies are becoming smaller and more isolated, causing a higher risk of extinction (Miller et al. 1994; Lomolino & Smith 2001; Ramirez & Keller 2010). With a decreasing population of keystone species, conservation of soil health and heterogeneity of grassland habitat is at stake. To maintain the health of prairie ecosystems, understanding the impacts of disturbance by human beings is necessary (Ramirez & Keller 2010).

Invasive vegetative species are often well suited to take advantage and establish with environmental changes and prosper on rangelands due to increasing human disturbances (Christian & Wilson 1999; Hobbs et al 2006; Wilsey et al. 2011). Some invasive species are highly adapted which allow them to compete and succeed with the native species in disturbed areas (Hobbs & Huenneke 1992; Pritekel et al. 2005), while other species are generalist which can occupy broad niches within an ecosystem. Exotic plants species can become photosynthetically active several weeks before comparable native species (Martin & Wilsey 2012), creating a lack of nutrients, sunlight, and space for native vegetation. Establishing native plant species before exotic plant species in the restoration process is essential for restoring diverse native prairie communities where perennial invasive species may exist (Martin & Wilsey 2012). A better understanding of the restoration process is needed to restore successful native plant diversity on highly disturbed soils. (Martin & Wilsey 2012; Martin et al. 2012; Cavender 2014).

Novel Rangeland Disturbances on Soil Characteristics

Soil compaction is an increase in soil density through a decrease in available pore space and a packing together of soil particles by forces exerted at the surface (Vomocil 1957; Thompson et al 1987). Anthropogenic and animal disturbance forces are known as the principal causes of soil compaction (Mulholland & Fullen 1991; Davies et al. 1992; Milne and Haynes 2004; Batey 2009). However, compact soils can also be found under other conditions without human or animal involvement. Soil compaction is a major issue in soil management (Batey 2009) due to increasing bulk density, and cascading many edaphic problems. Soil compaction decreases root growth and plant productivity by decreasing the storage and supply of water, air and nutrients (Thompson et al 1987; Andersen et al. 2013). In low soil organic matter content soils, compaction can be intensified by grazing, or use of heavy equipment during times of high soil water contents. These factors can limit the plant rooting depth.

Soil resistance to penetration is a measure of soil strength, and therefore, also a metric of compaction (Taylor 1971; Mason et al. 1988; Panayiotopoulos et al. 1994; Hamza & Anderson 2001, 2003). Root growth and penetration of soil can be limited and even prevented if soil compaction is sufficiently high (Barley et al. 1965; Thompson 1987). Soil compaction impedes the movement of water and air through the soil by reducing the pore size, level of aeration, and consequently, inhibit root and plant growth (Gifford et al 1977; Thompson et al 1987). Soil compaction processes influence soil water (Lipiec et al. 2002; Hamza & Anderson 2005), and increasing soil moisture content reduces the load of pressure found in soil (Medvedev & Cybulko 1995; Kondo & Dias Junior 1999; Hamza & Anderson 2005). Soil compaction and soil water content are tied at the moment of which compaction initially occurs. The reduction in soil pores by compaction event can change how soil water is retained and moves in the soil (Batey 2009).

With the continued increase of technology, axle weight of machinery has been increasing and causing major concerns for soil health across all types of landscapes (Hakansson 1994; Van den Akker et al. 2003; Godwin et al 2008; Batey 2009). Severe compaction is often associated with industrial activities where heavy machinery is being used, such as the extraction of minerals (Sinnett et al. 2006; Batey 2009), pipeline installation, and reclaimed landscapes (Batey & McKenzie 2006). In agricultural situations, compacted layers vary in thickness from a few millimeters to 100 millimeters (Hatley et al. 2005; Batey 2009). Compaction can occur on the surface, within a tilled layer, just below the tillage zone, or at greater depths. Industrial activities may have more severe impacts and can cause soil compaction at depths greater than one meter; sometimes persisting up to 30 years (Spoor 2006; Batey 2009). Severe compaction is a current issue on surface coal mining reclamation sites that the industry is trying to resolve with highly regulated reclamation techniques.

Surface coal mining has been an anthropogenic process for several decades. Most mines begin with filing for a mining permit of the area disturbed. Once a permit is accepted, dams and diversions are built to control runoff, followed by diesel-powered equipment removing the top-soil and sub-soil. Electric-powered draglines follow, removing the overburden that may range anywhere from 25-120 feet exposing the coal seams. Soil horizons are removed and placed into separate piles until needed for reclamation purposes, or is directly re-spread into an open pit that is ready for reclamation to begin. Once the soil has been removed, the coal seam is removed, and the reclamation process may begin.

Reclamation

Reclamation includes efforts that improve the quality of the land by restoring the remnant ecosystem (Bradshaw 1984; Holl 2002). Ecological restoration holds great promise for the recovery of degraded and destroyed ecosystems (Dobson et al.1997; Hobbs & Harris 2001). The process of natural succession reveals that nature can achieve restoration independently and develop functioning soils (Bradshaw 1996). The goal of reclamation is not to re-construct the natural species composition but to provide a healthy standing establishment of native species. In surface mining, the original vegetation is damaged, and the soil structure is usually lost or buried. To achieve a successful reclamation, soil needs to be replaced and the vegetation re-established (Bradshaw 1996). As legal requirements to reclaim highly disturbed areas are becoming more stringent, reclamation is focusing on both species and ecosystems (Holl 2002).

Federally, surface mine reclamation was highly unregulated prior to the early 1970s (Cavendar et al. 2014), however, some states had state mining regulations implemented. Topsoil, subsoil, and overburden were removed to expose coal seams. The overburden and soils were stockpiled adjacent to strip-mined areas and returned in reverse order of removal (subsoil on top of topsoil), or not returned at all (Cavendar et al. 2014). Ecosystem development and plant succession was slow on the stockpiled sites because they lacked ideal aerobic conditions for plant establishment and growth (Smyth 1997). Even with natural recovery of insufficient vegetation, dust, erosion and safety issues associated with non-reclaimed lands remain (Burger 2011). Due to an increase in environmental safety in the 1970s, a new federal law was implemented that made it mandatory for surface coal mines to restore mined areas back to as good or better than pre-mining condition.

The Surface Mining Control and Reclamation Act (SMCRA) of 1977 mandated mining operations to restore land back to the same or better vegetative state than pre-mining. Reshaping all areas affected by surface coal mining operations to a topography consistent with adjacent unmined landscape elements is essential to develop a post-mining landscape that will provide healthy soils and sufficient vegetative characteristics. “Re-establish a diverse, effective and permanent vegetative cover of the same seasonal variety and native to the area and capable of self-regeneration of plant succession…” is necessary to achieve the stated post-mining land use (Holl 2002; ND Public Service Commission 2015). With the implementation of this law, the new reclamation processes are mandatory beginning with the re-spreading of soil.

Reclamation enhances soil quality by re-establishing mine soils by terms of improving physical and chemical properties (Shukla et al. 2004). Reclaimed mine soils are developing on anthropogenically-altered landscapes and are pedologically young soils (Sencindiver & Ammons 2000). Reclaimed mine soils tend to have higher bulk density, lower porosity, poorer structure, lower water holding capacity, lower infiltration rates, and shallower rooting depths than that of undisturbed sites (Indorante et al 1981; Thurman & Sencindiver 1986; Dunker & Barnhisel 2000; Shukla et al 2004a). Due to heavy machinery and compaction, water moves slowly through the soil (Pedersen et al. 1980; Chong & Cowsert 1997), resulting in a higher resistance to root penetration and reduced root elongation (Fehrenbacher et al. 1982; Thompson et al. 1987; Chong & Cowsert 1997). Re-established soil at restored sites can suffer from severe compaction that may hinder future reclamation efforts such as native vegetation and healthy soils (Sinnett et al 2006). Studies report (e.g. Potter et al. 1988; Thomas & Jansen 1985) that reclaimed surface mined lands have slow soil structure development and profile characteristics that are not comparable to that of non-mined soils (Chong & Cowsert 1997). Therefore, the reclaimed mine soils need to be combined with various land-use practices to improve soil properties over time. The goal is to reconstruct reclaimed sites in a manner that maximizes the rate of soil improvement over time (Jansen 1981; Shukla 2004b).

The reclamation process currently begins with the re-spreading of the overburden, subsoil and topsoil respectively to homogeneous depths (24’,36’, or 48”) and contoured to a specific elevation stated in the mining permit using large equipment such as Caterpillar D11 dozers, 657 scrapers, 16M and 24M patrol blades, 789 end dump trucks and Hitachi 2500 excavators. Large rocks are removed, and a seedbed is prepared prior to seeding the newly replaced soil. The seeding mix used, usually consists of native cool and warm season grass species found naturally in the NGP.

Reclamation success is determined by methods approved for re-vegetation comparisons of vegetative ground cover, species diversity, and productivity between reclaimed and undisturbed reference areas (Schumann et al. 1999; Ries & Nilson 2000). In 1989, an extension to the SMCRA law of 1977 requires that all land disturbed by mining must remain under performance bond for a minimum of ten years from the date of seeding to insure that the land will be successfully reclaimed and is up to or above productive standards of pre-mining. Once the land meets these standards, the land is then released back to the previous landowner or bond returned to mining company. Bonds allow the establishment of vegetative stands and benefit from new species that move into the area, expanding the plant diversity on the site.

Achieving reclamation success in the western United States may be challenging due to diversity of plant species, high temperatures, and high evapotranspiration rates (Ries & Nilson 2000; Schladweiler et al. 2005). The fundamental objectives of surface mine reclamation are to assist vegetation establishment, improve the development of the plant community, and produce a self-sustaining ecosystem (Holl 2002; Hobbs et al. 2006; Alday et al. 2011). Plant succession commonly takes 50-100 years to recover before a satisfactory vegetation develops, especially on reclaimed mine sites (Bradshaw 1996; Dobson et al. 1997). Studies report the number of plant species increased with time since reclamation, and the composition of the oldest reclaimed sites approached that of adjacent, less-disturbed sites (Holl 2002). Native plant species typically colonize reclaimed coal surface mined sites after 10-15 years after reclamation (Thompson et al. 1984; Skousen et al. 1994; Thompson et al. 1996; Rodrigue 2001; Holl 2002). However, monitoring vegetation at different spatial scales is necessary to implement comprehensive management strategies over time (Cingolani et al. 2010) and assess re-vegetation.

Objectives

The objectives of this study is to (1) evaluate small- and large-scale vegetation patterns across a 40-year reclamation gradient, (2) to estimate spatial cover and vegetation diversity on the reclamation gradient to understand species response and seedling establishment on the re-spread soil, (3) determine the effect of soil compaction over a 40-year reclamation gradient, (4) evaluate the effect of rooting depth on compacted soils found on reclaimed mine sites, and (5) distinguish the level of soil water content range with regards to compacted soil. We hypothesis that alpha and beta diversity will increase with time since reclamation. We also hypothesis that soil compaction and soil water content range will decrease with time since reclamation, and rooting depth of vegetation will increase over the 40 year reclamation gradient.

Literature Cited

Alday JG, Pallavicini GY, Marrs RH and Martínez-Ruiz C. 2011. Functional groups and dispersal strategies as guides for predicting vegetation dynamics on reclaimed mines. *Plant Ecology* 212: 1759-1775.

Andersen MN, Munkholm LJ and Nielsen AL. 2013. Soil Compaction limits root development, radiation-use efficiency and yield of three winter wheat (*Triticum aestivum* L.) cultivars. *Soil and Plant Science* 63: 406-419.

Anderson E, Forrest S, Clark MTW and Richardson L. 1986. Paleobiology, biogeography, and systematics of the black-footed ferret, *Mustela nigripes* (Audubon and Buchman), *Great Basin Naturalist Memoirs* 8:11-62.

Anderson RC. 1990. The historic role of fire in the North American Grassland, Fire in tallgrass prairie ecosystem. University of Oklahoma Press, Norman, OK.

Anderson RC. 1997. Response of the Midwestern tallgrass prairies to summer burns, The Tallgrass Restoration Handbook. Island Press, Washing-ton, DC.

Anderson RC. 1998. Overview of Midwestern Oak Savanna. *Transactions of the Wisconsin Academy of Sciences and Arts and Letter.* 86: 1–18.

Anderson RC and Bowles ML. 1999. Deep soil savannas and barrens of the Midwestern United States, Savannas, barrens, and rock outcrop plant communities of North America. Cambridge University Press, New York, NY.

Anderson RC. 2006. Evolution and origin of the Central Grassland of North America: climate, fire, and mammalian grazers. *Journal of the Torrey Botanical Society* 133 (4): 626-647

Augustine DJ, Cully JF Jr and Johnson TL. 2007. Influence of Fire on Black-tailed Prairie Dog Colony Expansion in Shortgrass Steppe. *Rangeland Ecology and Management* 60: 538-542.

Augustine DJ, Derner JD and Milchunas DG. 2010. Prescribed Fire, Grazing, and Herbaceous Plant Production in Shortgrass Steppe. *Rangeland Ecology and Management* 63: 317-323.

Axelrod DI. 1985. Rise of the grassland biome, central North America. *Botanical Review* 51: 163-201

Barley KP, Farrell DA and Greacan EL. 1965. The influence of soil strength on the penetration of loam by plant roots. *Australia Soil Resource* 3: 69-79.

Batey T and McKenzie DC. 2006. Soil compaction: identification directly in the field. *Soil Use and Management* 22, 123-131.

Batey T. 2009. Soil compaction and soil management- a review. *Soil Use and Management* 25: 335-345.

Borchard P and Eldridge DJ. 2012. Vegetation changes associated with cattle (*Bos tauraus*) and wombat (*Vombatus ursinus*) activity in a riparian forest. *Applied Vegetation Science* 15: 62-70.

Bradshaw AD. 1984. Land restoration: now and in the future. *Proceedings of the Royal Society of London B* 223: 1-23.

Bradshaw AD. 1996. Restoration of mined lands- using natural processes. *Ecological Engineering* 8: 255-269

Bragg TB. 1995. The physical environment of Great Plains Grassland. The changing prairie North America grassland. Oxford University Press, New York, NY.

Burger JA. 2011. Sustainable mined land reclamation in the eastern US coalfields: A case for an ecosystem reclamation approach. *The American Society of Mining and Reclamation Proceedings:* Sciences Leading to Success. Lexington, Ky.

Cavendar N, Byrd S, Bechtoldt CL and Bauman JM. 2014. Vegetation Communities of a Coal Reclamation Site in Southeastern Ohio. *Northeastern Naturalist* 21 (1):31-46

Chong SK and Cowsert PT. 1997. Infiltration in reclaimed mined land ameliorated with deep tillage treatments. *Soil and Tillage Research* 44: 255-264.

Christian JM and Wilson SD.1999 Long-term ecosystem impacts of an introduced grass in the Northern Great Plains*. Ecology* 80: 2397-2407.

Christensen NL. 1997. Managing for heterogeneity and complexity on dynamic landscapes. The Ecological Basis for Conservation: Heterogeneity, Ecosystems, and Biodiversity*.* New York: Chapman and Hall.

Cingolani AM, Vaieretti MV, Gurvich DE, Giorgis MA and Cabido M. 2010. Predicting alpha, beta, and gamma plant diversity from physiognomic and physical indicators as a tool for ecosystem monitoring. *Biological Conservation* 143: 2570-2577.

Conacher A and Conacher J. 1995. Rural land degradation in Australia. Oxford University Press, Melbourne.

Crumpacker DW. 1984. Regional riparian research and a multi-university approach to the special problem of livestock grazing in the Rocky Mountains and Great Plains. California riparian systems: Ecology, conservation, and productive management. University of California Press, Berkeley, California.

Curtis JT. 1971. The Vegetation of Wisconsin. University of Wisconsin Press, Madison, WI. p.657

Daubenmire R. 1959. A canopy-coverage method of vegetational analysis. Northwest Science 33 (1): 43-64

Davies DB, Finney JB and Eagle DJ. 1992. *Soil management*, 5, Farming Press Ltd, Ipswich, England.

Dobson AP, Brashaw AD and Baker AJM. 1997. Hopes for the future: Restoration ecology and conservation biology. *Science* 277: 515-522.

Dunker RE and Barnhisel RI. 2000. Cropland reclamation. *Reclamation of Drastically Disturbed Lands.* 323-369.

Eldridge DJ. 2004. Mounds of the American Badger (*Taxidea taxus*): significant features of North American shrub-steppe ecosystems. *Journal of Mammal* 85:1060–1067.

Eldridge DJ and Whitford W G. 2009. Badger (*Taxidea taxus)* disturbances increase soil heterogeneity in a degraded shrub-steppe ecosystem. *Journal of Arid Environment* 73:66-73.

Fehrenbacher D, Jansen IJ and Fehrenbacher JB. 1982. Corn root development in constructed soils on surface-mined land in Western Illinois. *Soil Science Society of American* 46: 353-359.

Fleischner TL. 1994. Ecological Costs of Livestock Grazing in Western North America. *Society for Conservation Biology* 8 (3): 629-644.

Flores D. 1996. A long love affair with an uncommon country: environmental history and the Great Plains. Prairie conservation: preserving North America's most endangered ecosystem. *Island Press*, Washington, DC.

Foran BD, Bastin G and Hill B.1985. The pasture dynamics and management of two rangeland communities in the Victoria River District of the Northern Territory. *The Rangeland Journal* 7: 107-113.

Fuhlendorf SD and Engle DM. 1998. Soil heterogeneity influence on plant species response to grazing within a semi-arid savanna. *Plant Ecology* 138: 89-96.

Fuhlendorf SD and Smeins FE. 1999. Scaling effects of grazing in a semi-arid grassland. *Journal of Vegetation Science* 10: 731-738.

Fuhlendorf SD and Smeins FE. 2001. Restoring Heterogeneity on Rangelands: Ecosystem Management Bases on Evolutionary Grazing Patterns. *BioScience* 51(8): 625-632.

Fuhlendorf SD and Engle DM. 2004. Application of the Fire-Grazing Interaction to Restore a Shifting Mosaic on Tallgrass Prairie. *Journal of Ecology* 41: 604-614.

Fuls ER. 1992. Semi-arid and arid rangelands: A resource under siege due to patch-selective grazing. *Journal of Arid Environments* 22: 191–193.

Gauch HG Jr. 1982 Multivariate Analysis in Community Ecology. *Cambridge University Press*, Cambridge, UK.

Gifford FG, Faust RH and Coltharp GB. 1977. Measuring Soil Compaction on Rangeland. *Journal of Range Management* 30: 457-460.

Godwin R, Spoor G, Finney B, Hann M and Davies B. 2008. The current status of soil and water management in England. *Royal Agricultural Society of England*, Stoneleigh, Warwick.

Grubb PJ. 1977. The maintenance of species richness in plant communities: the importance of the regeneration niche. *Biological Reviews* 52:107-145.

Hakansson I. 1994. Subsoil compaction caused by heavy vehicles –a long term threat to soil productivity. *Soil and Tillage Research* 29: 105-110.

Hamza MA, Anderson SH and Aylmore LAG. 2001. Studies of water drawdowns by single radish roots at decreasing soil water content using computer assisted tomography. *Australian Journal of Soil Research* 39: 1387–1396.

Hamza MA and Anderson WK. 2003. Responses of soil properties and grain yields to deep ripping and gypsum application in a compacted loamy sand soil contrasted with a sandy clay loam soil in Western Australia. *Australian Journal of Agricultural Research* 54: 273– 282.

Hamza MA and Anderson WK. 2005. Soil compaction in cropping systems- A review of the nature, causes and possible solutions. *Oil and Tillage Research* 82: 121-145.

Hatley D, Wiltshire J, Basford B, Royale S, Buckley D and Johnson P. 2005. Soil compaction and potato crops. *Research Review* R260, British Potato Council, Oxford, UK.

Higgins KF. 1986. Effects of Fire in the Northern Great Plains. U.S. Fish and Wildlife Service and Cooperative Extension Service, South Dakota State University, U.S. Department of Agriculture EC 761.

Hill MO. 1979. DECORANA- A FORTRAN program for detrended correspondence analysis and reciprocal averaging. *Ecology and Systematics*, Cornell University, Ithaca, New York 14850 52.

Hobbs RJ and Huenneke LF. 1992. Disturbance, Diversity, and Invasion: Implications for Conservation. *Society for Conservation Biology* 6 (3): 324-337.

Hobbs RJ and Harris JA. 2001. Restoration ecology: repairing the earth's ecosystem in the new millennium. *Restoration Ecology* 9: 239-246.

Hobbs RJ, Arico S, Aronson J, Baron JS, Bridgewater P, Cramer VA, Epstein PR, Ewel JJ, Klink CA, Lugo AE, Norton D, Ojima D, Richardson DM, Sanderson EW, Valladeres F, Vila´ M, Zomoro R and Zobel M. 2006 Novel ecosystems: theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography* 15: 1-7.

Holechek JL, Pieper RD and Herbel CH. 2003. Range Management Principles and Practices. Prentice Hall Inc., Upper Saddle River, NJ.

Holl KD. 2002. Long-term vegetation recovery on reclaimed coal surface mines in eastern USA. *Journal of Applied Ecolog*y 39: 960-670.

Indorante SJ, Jansen IJ and Boast CW. 1981. Surface mining and reclamation: Initial changes in soil character*. Journal of Soil Water Conservation* 36:347–351.

Jansen IJ. 1981. Reconstructing soils after surface mining of prime agriculture land. *Mining Engineering* 33:312–314.

Kirkpatrick J. 1994. A Continent transformed: human impact on the natural vegetation of Australia. Oxford University Press, Oxford.

Kola J and Pickett STA. 1991. Ecological Heterogeneity, New York: *Springer-Verlag*.

Kondo MK and Dias Junior MS. 1999. Soil compressibility of three latosols as a function of moisture and use. *Revista Brasileira de Ciencia do Solo* 23: 211–218.

Landsberg J, James CD, Morton SR, Muller WJ and Stol J. 2003. Abundance and composition of plant species along grazing gradients in Australia rangeland. Jo*urnal of Applied Ecology* 40:1008-1024.

Li J, Duggin JA, Loneragan WA and Grant CD. 2006. Grassland responses to multiple disturbances on the New England Tablelands in NSW, Australia. *Plant Ecology* 193:39-57

Liang Y, Han GD, Zhou H, Zhao ML, Snyman HA, Shan D and Havstad KM. 2009. Grazing intensity on vegetation dynamics of a typical steppe in northeast Inner Mongolia. *Rangeland Ecology and Management* 62: 328–336.

Lipiec J, Ferrero Giovanetti AV, Nosalewicz A and Turski M. 2002. Response of structure to simulated trampling of woodland soil. *Advanced Geoecology* 35: 133–140.

Lomolino MV and Smith GA. 2001. Dynamic biogeography of prairie dog (Cynomys ludovicianus) towns near the edge of their range. *Journal of Mammalogy* 82:937-945.

Ludwig JA and Tongway DJ. 1995. Spatial organization of landscapes and its function in semi-arid woodlands, Australia. *Landscape Ecology* 10:51-63.

Lynn CA and Detling JK. 2008. Interactive Disturbance Effects of Two Disparate Ecosystem Engineers in North American shortgrass Steppe. *International Association for Ecology* 157:2: 269-278.

Magurran AE. 1988. Ecological Diversity and Its Measurement. Princeton University Press 1-175.

Martin SC and Ward DE. 1970. Rotating access to water to improve semi-desert cattle range near water. *Journal of Range Management* 22-26.

Martin LM, Moloney KA and Wilsey BJ. 2005. As assessment of grassland restoration success using species diversity components. *Journal of Applied Ecology* 42: 327-336.

Martin LM and Wilsey BJ. 2012. Assembly history alters alpha and beta diversity, exotic-native proportions and functioning of restored prairie plant communities. *Journal of Applied Ecology* 49: 1436-1445.

Mason EG, Cullen AWJ and Rijkse WC. 1988. Growth of two Pinus radiata stock types on ripped and ripped/bedded plots at Karioi forest. N. Zeal. *Journal of Forestry Sci*ence 18: 287–296.

McCune B, Grace JB and Urban DL. 2002. Analysis of ecological communities. Gleneden Beach, Oregon: MjM software design. 28.

McIntyre S, Lavorel S and Tremont PM. 1995. Plant life history attributes: Their relationship to disturbance response in herbaceous vegetation. *Journal of Ecology* 83: 31-44.

Medvedev VV and Cybulko WG. 1995. Soil criteria for assessing the maximum permissible ground pressure of agricultural vehicles on Chernozem soils. *Soil Tillage Research* 36: 153–164.

Milchunas DG, Sala OE and Lauenroth WK. 1988. A generalized model of the effects of grazing by large herbivores on grassland community structure. *American Midland Naturalist* 132:87-106.

Miller B, Ceballos G and Reading R. 1994. The prairie dog and biotic diversity. *Conservation Biology* 8:677–681.

Miller SD and Cully JFJR. 2001. Conservation of black-tailed prairie dogs (Cynomys ludovicianus). *Journal of Mammalogy* 82:889-893.

Milne RM and Haynes RJ. 2004. Comparative effects of annual and permanent dairy pastures on soil physical properties in the Tsitsikamma region of South Africa. *Soil Use and Management* 20: 81-88.

Moloney HA and Levin SA. 1996. The effects of disturbance architecture on landscape-level population dynamics*. Ecology* 77:375-394.

Mulholland B and Fullen MA. 1991. Cattle trampling and soil compaction on loamy sands. S*oil Use and Management* 7: 189-193.

North Dakota Public Service Commission. 2015. Surface Mining and Reclamation Operations. Chapter 38-14.1. <http://www.legis.nd.gov/cencode/t38c14-1.pdf?20150501100154> Date accessed 1 May 2015.

Ostfeld RS, Pickett STA, Shachek M and Likens GE. 1997. Defining scientific issues. *The Ecological Basis for Conservation: Geterogeneity, Ecosystems, and Biodiversity*. New York: Chapman and Hall.

Panayiotopoulos KP, Papadopoulou CP and Hatjiioannidou A. 1994. Compaction and penetration resistance of an Alfisol and Entisol and their influence on root growth of maize seedlings. S*oil Tillage Res*earch 31: 323–337.

Pattern RS and Ellis JE. 1995. Patterns of species and community distribution related to environmental gradients in an arid tropical ecosystems. *Vegetation* 117: 69-79.

Pedersen TA, Rogowski AS and Pennock R Jr. 1980. Physical characteristics of some minesoils. *Soil Science Society of America* 44: 321-328.

Pickett STA and White P.1985. The ecology of natural disturbance and patch dynamics. Academic Press, Orlando.

Potter KN, Carter FS and Doll EC. 1988. Physical properties of constructed and undisturbed soils. *Soil Science Society of American* 52: 1435-1438.

Pritekel C, Whittemore-Olson A, Snow N and Moor, JC. 2005. Impacts from invasive plant species and their control on the plant community and belowground ecosystem at Rocky Mountain National Park, USA. *Applied Soil Ecology* 32: 132-141.

Pyne SJ. 1983. Indian fires. *Natural History* 2:6-11.

Pyne S. 1997. Fire in America: A cultural history of wildland and rural fire. University of Washing-ton Press, Seattle, WA. 654.

Ramirez JE and Keller GS. 2010. Effects of Landscape on Behavior of Black-Tailed Prairie Dogs (*Cynomys ludovicianus*) in Rural and Urban Habitats. *Southwestern Association of Nautralists* 55(2):167-171.

Retallack GJ. 2001. Cenozoic expansion of grasslands and climatic cooling. *Journal of Geology* 109: 407-426.

Ries RE and Nilson DJ. 2000. Reclamation considerations for range, pasture, and hay lands receiving twenty-five to sixty-six centimeters annual precipitation. *American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America* 41:273-302.

Rodrigue JA. 2001, Woody species diversity, forest and site productivity, stumpage value, and carbon sequestration of forests on mined lands reclaimed prior to the passage of the Surface Mining Control and Reclamation Act of 1977. MS Thesis. Virginia Polytechnic Institute and State University, Blacksburg, VA.

Saur CO. 1950. Grassland climax, fire, and man. *Journal of Range Management* 3: 16-21.

Sch€onbach P, Wan HW, Gierus M, Baiy F, Mǜller K, Lin LJ, Susenbeth A and Taube F. 2011. Grassland responses to grazing: effects of grazing intensity and management system in an Inner Mongolian steppe ecosystem. *Plant and Soil* 340: 103–115.

Schladweiler BK, Vance GF, Legg DE, Munn LC, and Haroian R. 2005. Topsoil Depth Effects on Reclaimed Coal Mine and Native Area Vegetation in Northeastern Wyoming. *Society for Range Management* 58(2):167-176

Schumann GE. 1999. Assessment of mined land rehabilitation success. *In: Current challenges in land management*. Fremantle, Western Australia: *Environmental Consultants Association* 1-6.

Sencindiver JC and Ammons JT. 2000. Minesoil genesis and classification. Agronomy monograph no. 41. *Reclamation of Drastically Disturbed Lands,* Madison, WI, 595-613.

Senft RL, Coughenour MB, Bailey DW, Rittenhouse LB, Sala OE and Swift DM. 1987. Large herbivores' foraging and ecological hierarchies. *BioScience* 37: 789-799.

Sepkoski J Jr. 1988. Alpha, Beta and Gamma: Where does all the diversity go? *Paleobiology* 14:221-234.

Shukla MK, Lal R and Ebinger M. 2004a. Soil quality indicators for reclaimed mine soils in southeastern Ohio. *Soil Science* 169:133-142.

Shukla MK, Lal R, Underwood J and Ebinge M. 2004b. Physical and Hydrological Characteristics of Reclaimed Minesoils in Southeastern Ohio. *Soil Science Society of America* 68:1352-1359.

Sinnett JD, Poole J and Hutchings TR. 2006. The efficacy of three techniques to alleviate soil compaction at a restored sand and gravel quary. *Soil Use and Management* 22: 362-371.

Skousen JG, Johnson CD and Garbutt K. 1994. Natural revegetation of 15 abandoned mine land sites in West Virginia. *Journal of Environmental Qualit*y 23:1224-1230.

Smith MD, Wilcox JC, Kelly T and Knapp AK. 2004. Dominance not richness determines invisibility of tallgrass prairie. *Oikos* 106: 253-262.

Smyth CR. 1997. Early Succession Patterns with a Native Species Seed Mix on Amended and Un-amended Coal Mine Spoil in the Rocky Mountains of Southeastern British Columbia, Canada. *Arctic and Alpine Research* 29(2): 184-195.

Sousa WP. 1984. The role of disturbance in natural communities. *Annual Review Ecology and Systematics* 15:353-391.

Spoor G. 2006. Alleviation of soil compaction: requirements equipment and techniques. *Soil Use and Management* 22: 113-122.

Stewart OC. 1956. Fire as the first great force employed by man, University of Chicago Press, Chicago, IL. 115-133.

Strapp P. 1998. A Re-evaluation of the Role of Prairie Dogs in Great Plains Grasslands. *Conservation Biology* 12(6):1253-1259.

Stirling G and Wilsey B. 2001. Empirical relationships between species richness, evenness, and proportional diversity. *American Naturalist* 158, 286-299.

Stuth JW. 1991. Foraging Behavior. *Grazing Management: An Ecological Perspective*. Portland Oregon: *Timber Press* 65-83.

Taylor HM. 1971. Effect of soil strength on seedling emergence, root growth and crop yield. Compaction of Agricultural Soils, *American Society of Agricultural Engineering* 292-305.

Thomas D, and Jansen IJ. 1985. Soil development in coal mine spoils. *Soil Water Conservation* 40: 439-442.

Thompson RL, Vogel WG and Taylor DD. 1984. Vegetation and flora of a coal surface-mined area in Laurel County, Kentucky. *Castanea* 49: 111-126.

Thompson PJ, Jansen IJ and Hooks CL. 1987. Penetrometer resistance and bulk density as parameters for predicting root system performance in mine soils. *Soil Science Society of America* 51: 1288-1293.

Thompson RL, Wade GL and Straw RA. 1996. Natural and planted flora of the Log Mountain surface mined. Virginia Tech Research Division, Blacksburg, VA. 484-503.

Thompson CM, Augustine DJ and Mayers DM. 2008. Swift Fox Response to Prescribed Fire in Shortgrass Steppe. *Western North American Naturalist* 68: 251-256.

Thurman NC, Sencindiver JC. 1986. Properties, classification, and interpretations of mine soils at two sites in West Virginia. *Soil Science of America* 50:181-185.

Tobler MW, Cochard R and Edwards PJ. 2003. The impact of cattle ranching on large-scale vegetation patterns in a coastal savanna in Tanzania. *Journal of Applied Ecology* 40: 430-444.

Van den Akker JHH, Arvidsson J and Horn R. 2003. Introduction to the special issue on experiences with the impact and prevention of subsoil compaction in the European Union. *Soil and Tillage Research* 73: 1-8.

van der Westhuizen HC, Snyman HA and Fouche HJ. 2005. A degradation gradient for the assessment of rangeland condition of a semi-arid sourveld in southern Africa. *African Journal of Range and Forage Science* 22: 47-58.

Vermeire LT, Heitschmidt RK, Johnson PS and Sowell BF. 2004. The prairie dog story: do we have it right? *BioScience* 54:689-695.

Vomocil JA. 1957. Measurement of soil bulk density and penetrability: A review of method. *Advance Agronomy* 9: 159-176.

Wagner FH. 1978. Livestock grazing and the livestock industry. *Wildlife and America*. Council on Environmental Quality Washington, D.C. 121-145.

Wiens JA. 1974. Habitat heterogeneity and avian community structure in North American grassland birds. *American Midland Naturalist* 91: 195-213.

Wight OW. 1877. Report of progress and results for the year 1875*. Geology of Wisconsin* 2:67-89.

Wilsey BJ, and Potvin C. 2000. Biodiversity and ecosystem functioning: importance of species evenness in an old field. *Ecology* 81: 887-892.

Wilsey BJ, and Polley HW. 2002 Reductions in species evenness increase dicot seedling invasion and spittle bug infestation. *Ecology Letters* 5: 676-684.

Wilsey BJ, and Polley HW. 2004. Realistically low species evenness does not alter grassland species richness-productivity relationships*. Ecology* 85:2693-2700.

Wilsey BJ, Martin LM and Polley HW. 2005 Predicting plant extinction based on species- area curves in prairie fragments with high beta richness. *Conservation Biology* 19: 1835-1841.

Wilsey BJ, Daneshgar PP and Polley HW. 2011 Biodiversity, phenology and temporal niche differences between native- and novel-exotic dominated grasslands. Perspectives in Plant Ecology, *Evolution and Systematics* 13: 256-276.

Wright HA, and Bailey AW. 1980. Fire ecology: United States and southern Canada. John Wiley and Sons, Inc, New York 501.

CHAPTER 2: SMALL AND LARGE-SCALE VEGETATION PATTERNS OF DIVERSITY ON RECLAIMED SURFACE COAL MINELAND OVER A 40- YEAR CHRONOSEQUENCE

Abstract

Rangelands are described as heterogeneous, due to diversity species assemblages and productivity, resulting from disturbances across multiple scales. Reclaiming rangelands often focus on facilitating vegetation establishment, enhancing the development of the plant community and creating a self-sustaining ecosystem. However, reclamation efforts following anthropogenic disturbances focused on reclaiming native biodiversity, but largely overlooked the need to reclaim heterogeneous patterns within landscapes. The objectives were to evaluate the small- and large-scale vegetation patterns across a 40-year reclamation gradient on reclaimed surface coalmine lands. We hypothesized that both alpha and beta diversity would increase and species dissimilarity to reference sites would decrease over the reclamation gradient. Plant communities were surveyed on 19 post-coalmine reclaimed and four native reference sites in central North Dakota mixed-grass prairie. Our results showed no differences in alpha or beta diversity and plant community patch size over the 40-year reclamation gradient. However, both alpha and beta diversity on reclaimed sites mimicked reference sites. Native species recruitment was limited due to an influx of invasive species such as Kentucky bluegrass (*Poa pratensis*) on both the reclaimed and reference sites. Species composition was different between reclaimed and reference sites. Plant composition dissimilarity on reclaimed sites increased with over 40 years. Plant communities resulting from reclamation followed non-equilibrium succession, even with consistent seeds mixes established across all reclaimed years. This suggests that post-reclamation management strategies are critical to determine species composition outcomes and that homogeneous land management may lead to decreased landscape-level diversity.

Introduction

Heterogeneity is an innate characteristic of rangelands due in part to disturbances of plant species diversity, productivity across multiple scales (Pattern and Ellis 1995; Fuhlendorf and Smeins 1999; Fuhlendorf and Engle 2001), and often credited as the root of biological diversity (Christensen

1997; Fuhlendorf and Engle 2004). Heterogeneity should serve as the foundation for ecosystem management and be a priority in reclaimed landscapes (Wiens 1974; Christensen 1997; Fuhlendorf et al. 2012). However, the importance of heterogeneity and spatial patterns is not historically acknowledged in disturbance and reclamation ecology (Fuhlendorf and Engle 2001). However, reclamation activities should enhance the development of the plant community by reclaiming a stable and functional plant community (Holl 2002; Hobbs et al. 2007; Alday et al. 2011) using alpha and beta diversity metrics.

A common goal of reclamation is to reconstruct the pattern of species richness found in remnant sites (Polley et al. 2005), and to arrive at a desired vegetative community by speeding up succession (Hobbes et al. 2007). However, reclamation sites are monitored primarily for the number of native/exotic species that are present, and less attention is given to measuring species evenness and spatial patterns. Recreating the patterns of plant species richness and abundances found in reference sites is difficult to achieve (Howe 1994) due to a poor understanding of the underlying mechanisms driving diversity in reclaimed communities (Polley et al. 2005). Research on patterns of both species richness and abundances in reclamation is limited (Kindscher and Tieszen 1998) and even fewer investigations focus on diversity changes across spatial scales. To begin and fully understand the factors controlling vegetation dynamics during reclamation, we first need to describe the patterns of richness and abundances that emerge through time (Pickett et al. 1985).

The diversity/stability hypothesis suggests that increased diversity will enhance the development of the plant community, and increase the likelihood of a self-sustaining ecosystem (Hurd et al. 1971). Common metrics to gauge reclamation are alpha diversity and vegetation composition. The two components of alpha diversity (richness and evenness) may respond differently to ecological factors or across spatial scales (Wilson et al. 1996; Stirling and Wilsey 2001), resulting in a wider variety of vegetation to the ecosystem. In order to increase or maintain species richness over the long-term, a better understanding of the processes that promote equitability in species abundance is required (Polley et al. 2005).

Understanding all factors driving diversity is necessary in heterogeneous native landscapes (Polley et. al 2005). Seasonal timing of disturbances could enhance beta diversity if species exploit disturbances at different times of the year (Questad and Foster 2008). Grazing disturbances can promote homogeneous or heterogeneous landscapes depending on management strategies (Holechek et al. 2003; Fuhlendorf and Engle 2004). Therefore, rangeland managers should focus more on reclaiming the heterogeneity characteristic within the landscape than on reclaiming the previous successional composition of grasslands post disturbance (Fuhlendorf and Smeins 1998).

Many reclamation projects result from anthropogenic disturbances that cause a change in an ecological state in which disrupts the natural successional trajectories. Novel or anthropogenic communities result when species combinations, abundances, and changes in ecosystem functions exist, but do not occur naturally within a biome (Kirkpatrick 1994; Conacher and Conacher 1995; Hobbs et al. 2007). This can often be a result of different disturbance regimes (i.e. fire, grazing and fossorial animals), which play significant roles in shaping patterns of alpha and beta diversity (Limb et al. 2010). Native diversity can be difficult to achieve due to the ecosystem's lack of resistance to exotic species invasion with these different disturbance regimes (McIntyre et al. 1995; Moloney and Levin 1996; Li et al. 2006).

Surface coal mining is a large-scale anthropogenic disturbance found on rangelands in the western United States. In surface mining, the original vegetation and soil structure is damaged. To achieve a successful reclamation, the soil needs to be remediated and the vegetation re-established (Bradshaw 1996); in order to focus on species composition and ecosystem sustainability (Holl 2002). The process of natural succession reveals that nature can achieve reclamation independently and develop desirable soil characteristics (Bradshaw 1996). The goal of mine-land reclamation is not necessarily to re-construct the natural species composition, but rather to provide a stable and productive stand of perennial vegetation (Holl 2002). However, many reclaimed surface mine-land plant communities are found to be more productive and less diverse than the nearby undisturbed lands (Dangi et al 2011).

The working assumption on reclaimed mine-land is that diversity will increase with time-since-reclamation due to successional processes and species immigration from soil seedbanks and seed rain. Species may group according to variation in abiotic conditions, dispersal constraints, and community assembly history (Whittaker 1960; Belyea and Lancaster 1999; Chase 2003; Martin and Wilsey 2012). This suggests that a single stable equilibrium may be reached by species composition within a uniform environment, but show high beta diversity among environments (Chase 2003; Martin and Wilsey 2012). If the historical order of species establishment differs, then multiple stable equilibriums can form within uniform environments, also generating high beta diversity (Drake 1991; Chase 2003; Martin and Wilsey 2012). Differences in species dispersal may generate beta diversity where some species fail to reach all appropriate locations (Martin and Wilsey 2012).

In this study, we used a chronosequence approach to characterize the change in plant community patterns that developed during a reclamation period of approximately 40 years on a surface coalmine. The objectives of this study were to evaluate: 1) small-scale vegetation patterns, 2) species composition, and 3) large-scale vegetation patterns across 40 years of reclamation. The working hypothesis is that both alpha and beta diversity, and species similarity to reference sites will increase with time since reclamation.

Methods

Experimental Design, Field and Lab Methods

Research was conducted at BNI Coal in central North Dakota approximately 5 km southeast of Center in Oliver county, North Dakota (lat 47°6'54 "N long 101°18'1 "W). The site is a mixed-grass prairie in the Northern Great Plains ecoregion at 602-m elevation. The climate is semi-arid with a 30-year mean annual precipitation of 406 mm and the majority of precipitation occurs between mid-April to mid-September (NDAWN 2015). The 30-year mean daily air temperatures range between 24°C in June to -16 °C in January with a 120-day frost-free growing season (NDAWN 2015; WRCC 2015). The 2014 growing season had above-average precipitation (466 mm) with below-normal temperatures (17.2°C) (NDAWN 2015; USDA NRCS 2015).

We sampled plant communities among a chronosequence of 19 post-mine reclaimed and four intact native reference sites (reclamation year 1975, 1985, 1986, 1988, 1993, 1994 and 1997- 2010) in the summer of 2014 creating a time-since-reclamation gradient spanning nearly 40 years. The four native reference sites are classified as loamy, sandy loamy, thin loamy, and shallow loamy ecological sites (USDA NRCS 2015). Mine reclamation sites were leveled prior to seeding, contain minimal micro-topography (<2-10% slope), and range from 2.4 to 34.8 ha in area. Soil was re-spread using a variety of Caterpillar equipment, such as D11 dozers, 657 scrapers, 16M and 24M patrol blades, 789 end dump trucks and Hitachi 2500 excavators. Re-spread depths of topsoil, and subsoil ranged from 61-121.9 cm.

Initial seed mix used in the reclamation process includes: western wheatgrass (*Pascopyrum smithii*), littlebluestem (*Schizachyrium scoparium*), big bluestem (*Andropogon gerardii*), sideoats grama (*Bouteloua curtipendula*), bluegrama grass (*Bouteloua gracilis*), and switchgrass (*Panicum virgatum*). Management strategies on the sites are a combination of season-long grazing and haying.

We established two 70-m transects in each reclamation year and reference site with a minimum of 10-m between the two transects. Species composition and abundance were estimated using a modified Daubenmire frame at two-meter intervals along each transect in mid-summer (Daubenmire 1959). Canopy cover estimates were utilized using a cover class (trace-1%, 1-2%, 2-5%, 5-10%, 10-20%, 20-30% 30-40% 40-50% 50-60% 60-70% 70-80% 80-90% 90-95% 95-98% 98-99% 99-100%). Species composition from each transect was compiled into a composite sample for the reclamation year. The midpoint values for each class were used for analysis.

Statistical analysis

Species richness, evenness, and alpha diversity (Simpson D) were subject to regression analysis (IBM SPSS 21) to determine relationships with time-since-reclamation. We complied data from the 36 quadrats in each transect and averaged across the two transects in each reclamation year. The mean species composition among the two 70-m transects for each site were analyzed with nonmetric multi-dimensional scaling (NMS) in PC-ORD 6.0. We used the Relative Sørensen distance measure in PC-ORD autopilot mode, which conducts 250 runs with real data and 250 runs with randomized data. In the autopilot mode, PC-ORD selects the best-fit solution (lowest stress) in a possible one–six dimensional solution (McCune and Grace 2002). A Monte Carlo test comparing the real data to the randomized data was used to determine significance (McCune and Grace 2002). Additionally, community dissimilarity was determined using the Sørenson dissimilarity index.

We calculated β-diversity (Whittaker 1972) (Equ. 1) for each transect and used regression procedures to determine the mean β-diversity for each reclamation year and time-since-reclamation.

$$\beta = \gamma/\alpha \tag{2.1}$$

Where γ is the total number of species, and α is the mean number of species in alpha sample. The inherent small-scale dissimilarity of vegetation along with the average patch size and the highest mean dissimilarity between patches was determined using dissimilograms based on the Relative Sørensen dissimilarity index in PC-ORD 6.0. The dissimilogram measures the dissimilarity of the quadrats along transects rather than the variance among the quadrats. We used the Gompertz asymptotic equation (Brownstein et al. 2012) (Equ. 2) for the dissimilogram analysis to fit a relation of the Relative Sørenson dissimilarities between quadrats as the dependent y-variable and their spatial distance apart (quadrat centers) as the independent x-variable. Each parameter *a, b* and *c* are fitted and *e* is Euler's number.

$$D_{\text{Sørenson}} = a \cdot e^{-b \cdot e^{(-c \cdot \text{distance})}} \tag{2.2}$$

The dissimilarity is calculated for the various lag distances and increases with increased lag distance to an asymptote at which no additional plant species variation exist.

Similar to spatial patterns in vegetation composition, individual plants in close proximity are more likely to be similar than distant ones. Therefore, patterns of plant species composition represented with the dissimilogram can be interpreted similar to a spatial semivariogram (Mistral et al. 2000). We substituted the semi-variance with the dissimilarity and the range with patch size. The sill (a) is the measure of the maximum dissimilarity between quadrats (asymptote).

The 90% dissimilarity (Lawrence Lodge et al. 2007; Roe et al. 2011; Meyers et al. 2014) value is 90% of the theoretical lag distance that an asymptote is predicted and indicates the distance where dissimilarity becomes independent in space. The number of comparable locations diminishes with increased lags. Therefore, we used the 90% asymptote to estimate the lag distance to ensure adequate sample size. The corresponding lag distance with the 90% dissimilarity was used as an indication of mean plant community patch size. Low patchiness in the plot would be indicted by low mean dissimilarity, low inherent variability and large mean patch size. Relative plant community patch size on reclaimed lands over four decades indicates the landscape-level success of the current ecosystem-based reclamation strategy.

Results

Species Richness, Evenness, and Composition

We recorded a total of 65 plant species across the 19 reclaimed and four reference sites. Species richness averaged 16 (12 min-23 max) per reclamation year and 17 (14 min - 22 max) per reference site. The data show that exotic species had a higher abundance than the native species over the 40 year reclamation gradient (Fig. 1). Invasive species canopy cover dominated the overall plant community, ranging from 52% to 97%, with a mean of 83%, across all sites. In contrast, native species canopy cover ranged from < 1% to 56%, with a mean of 16%, across all sites. Grass cover was generally higher than forbs in the reclaimed areas; however, of those grasses, the majority of species are exotic. Exotic grasses dominated the reclaimed areas with an average of 72%, and reference areas also exhibited more exotic species compared to native grasses with the averages of 53% and 42% respectively. Plant species not in initial grass seed mix immigrated into the reclaimed sites and accounted for 94% of plant abundance as opposed to the initial grass seed mix that accounted for only 6% (Fig. 1). These relative plant abundances of immigrated and seeded plants were generally stationary across the chronosequence from 5 to 40 years since reclamation. Vegetative species evenness did not differ over the 40-year reclamation gradient. Similarly, alpha diversity was stationary along the chronosequence averaging 0.67 (0.55 min - 0.84 max).

The NMS analysis indicated a three dimensional solution with a final stress of 7.15. Axis one is the strongest driver at 77% of the cumulative variability with all axes explaining 94% of the total variation. While Kentucky bluegrass was present in all reclamation years and reference locations, it was not a driving factor with values of 0.06, -0.04, and -0.02 along axis 1, 2 and 3 respectively (Fig. 2). Negative values along the primary axis in the NMS bi-plot (Fig. 2) show a strong relationship with scarlet globemallow (*Sphaeralcea coccinea*), and prairie rose (*Rosa arkansana*) contrary to positive values with a strong influence of cudweed sagewort (*Artemisia Ludoviciana*), blue grama (*Bouteloua gracilis)*, other native grasses and reference sites. Overall, species composition on reclaimed sites was different than reference locations for all years. Further, dissimilarity did not decrease on older reclaimed sites, but rather increased with time since reclamation (Fig. 3).

Beta Diversity and 90% Patch Size

The data show no change ($P > 0.05$) in beta diversity over the reclamation gradient (Fig. 4a). Beta diversity averaged 1.8, and ranged between 0.9 and 2.7. Similarly, 90% patch size (0.66 - 2465.2) showed no relationship with time-since-reclamation (Fig. 4b).

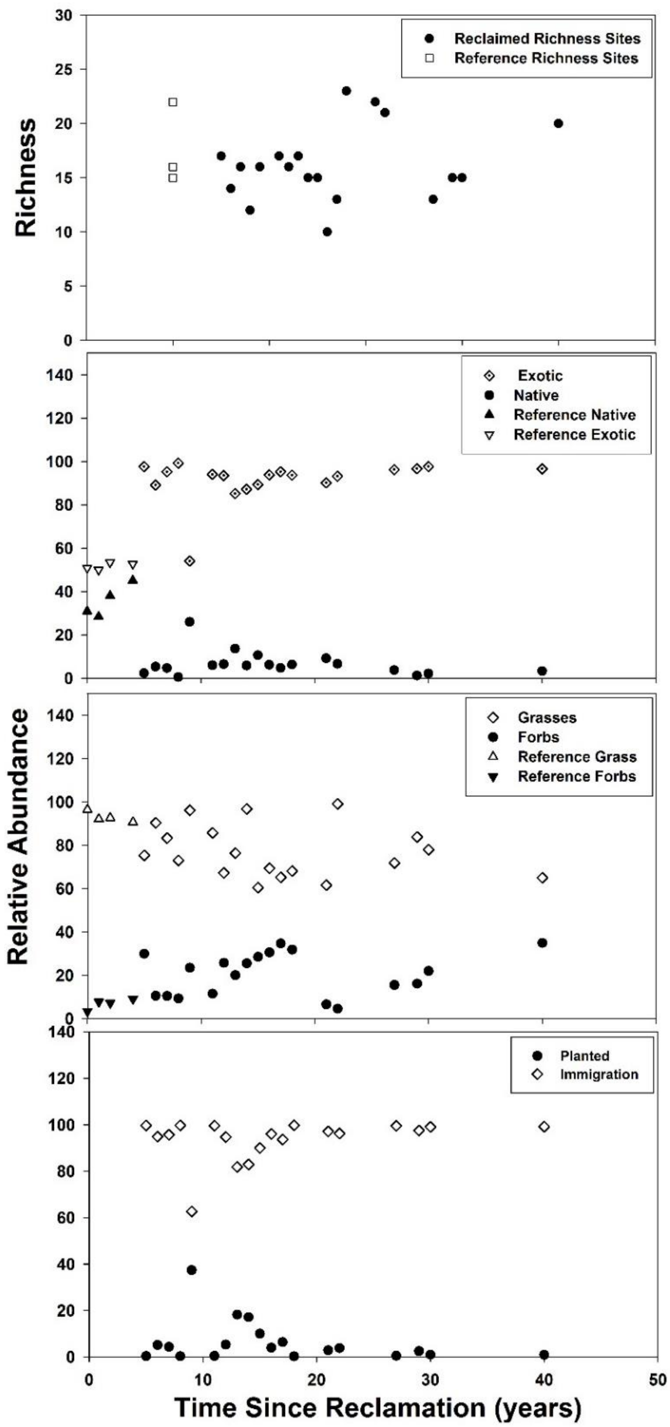

Figure 2.1: Represents species richness (a) and relative abundances over a 40-year reclamation gradient. Exotic species are largely abundant compared to native in all sites (b). Grasses are dominant and reclaimed forbs show to be more abundant than reference forbs (c). Immigrated species show higher abundance than species initially planted in reclamation seed mix (d). Data collected from BNI Coal mine, Center, ND, USA (2014).

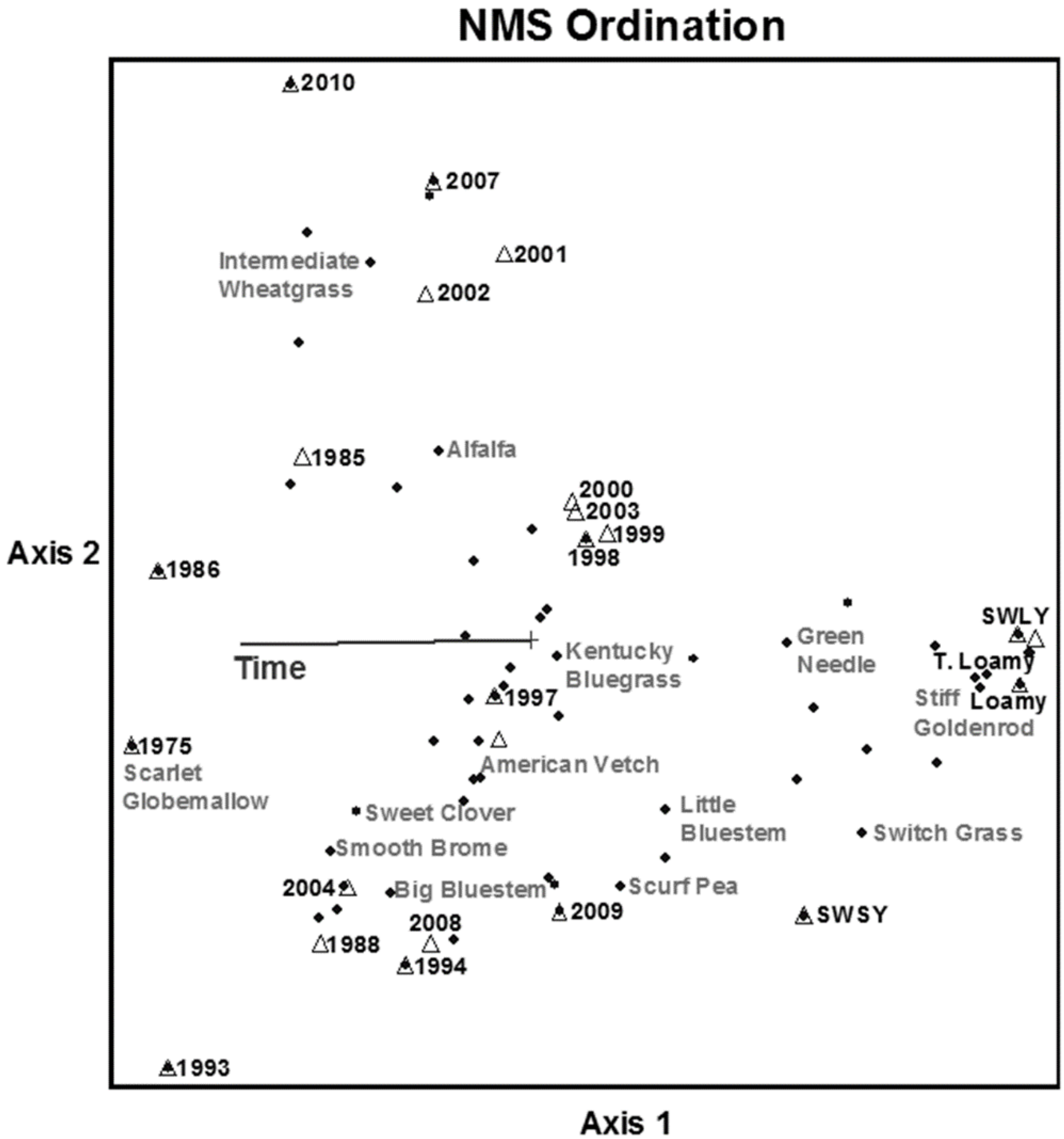

Figure 2.2: Non-metric multidimensional scaling ordination (NMS) of axis 1 and 2 with regards to time, sites, and vegetation from BNI Coal mine, Center, ND, USA (2014).

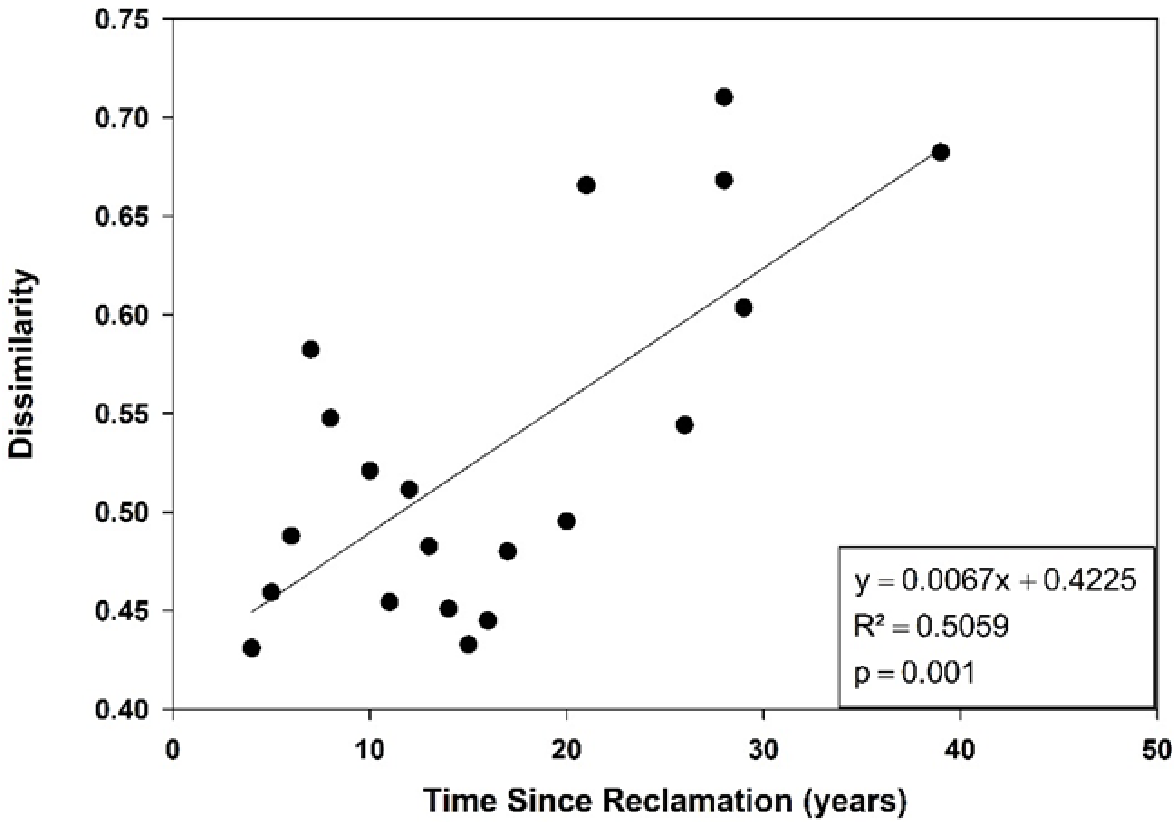

Figure 2.3: Mean dissimilarity to reference sites at BNI Coal near Center, ND, USA. The increasing dissimilarity over the 40-year reclamation gradient suggests that the restored plant communities are displaying patterns of non-equilibrium succession.

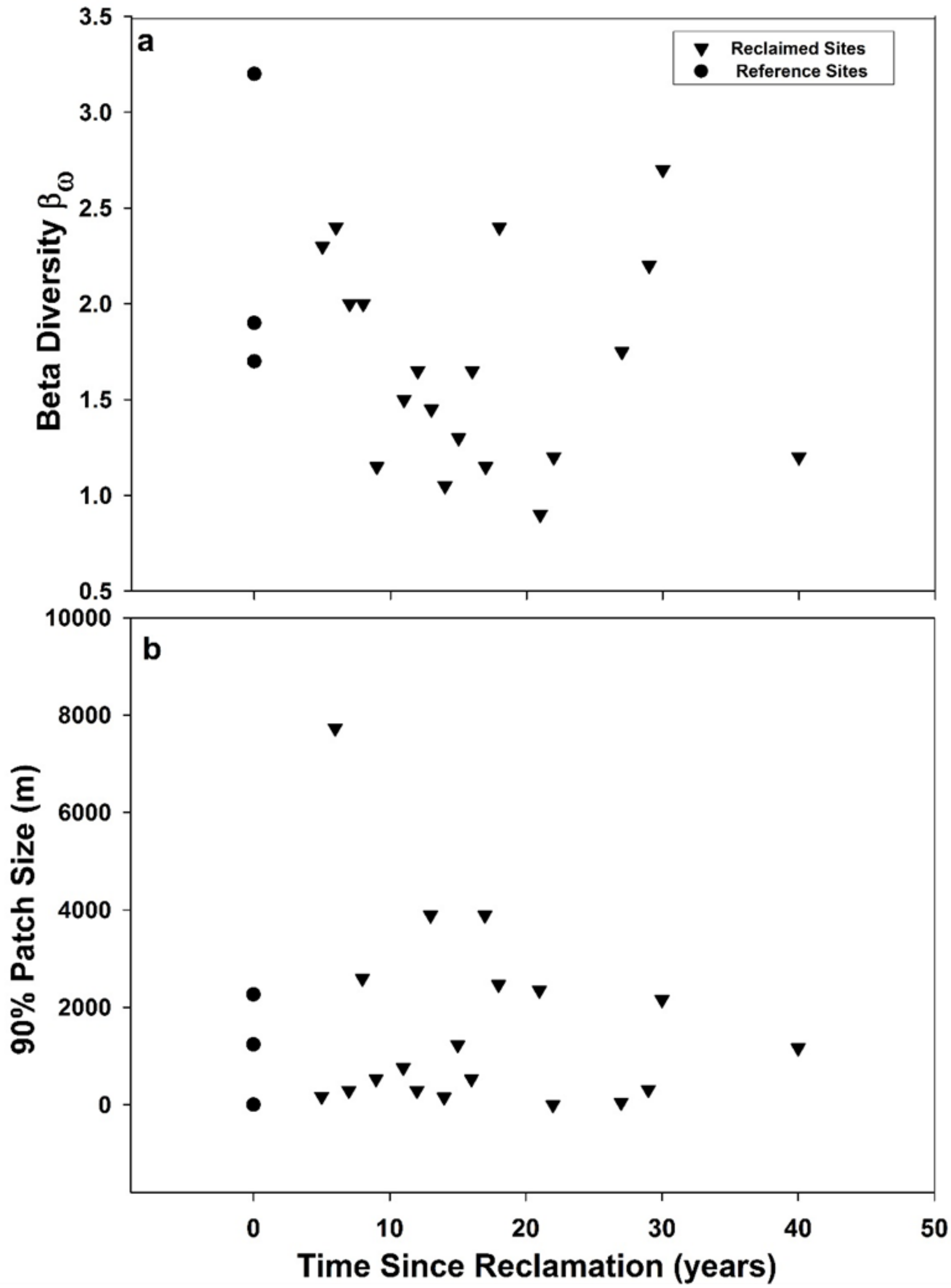

Figure 2.4: Represents site beta diversity (a), and patch size (b) over a 40-year reclamation gradient and four reference sites taken from BNI Coal, ND (2014). (Two reference sites overlap in graphs)

Discussion

Species Richness, Evenness, and Composition

Increasing biodiversity is a common goal following anthropogenic disturbance. However, monitoring diversity is often limited to fine scales without consideration to larger-scale patterns. Vegetation monitoring at different spatial scales is needed for comprehensive management strategies and practices that will thrive over time (Cingolani et al. 2010) in order to decrease the potential for the establishment and success of invasive species. Plant species composition was different between reclaimed and reference sites, both fine- and large-scale diversity patterns on reclaimed sites closely mimicked reference sites across all years. However, the reference sites are not representative of native Northern Mixed Grass Prairie due abundance of exotic species.

Time following reclamation (i.e., 40 years) showed minimal increases in species richness and native species recruitment. The most abundant plant species are those used in the original native seed mix, and Kentucky bluegrass, an exotic and invasive species present in all reclamation sites and the reference locations. Kentucky bluegrass can create a dominant herbaceous layer of both senesced and live stems that restricts native recruitment by altering nutrient cycles, water and sunlight availability, soil microbial processes from historic conditions (Vitousek 1990; Pritekel et al. 2006; Cavendar 2014) and increases in abundance with the lack of fire and herbivory (Grant et al. 2009). In past studies, plant community shifts were found due to the invasion of exotic aggressive species, and the plant community properties changed over 25 years since reclamation (Newman and Redente 2001; Bowen et al. 2005; Wick et al. 2011). Many surface mine reclamation efforts in the past focused on establishing rapid-growing, non-native species that control erosion but may out compete shade intolerant native species (Holl 2002).

A common concept with primary succession is that species richness will increase with time, particularly with anthropogenic assisted succession. Several reports state that native species re-colonize on surface mine sites in the eastern United States after 10 to 15 years of reclamation (Skousen et al. 1994; Thompson et al. 1996; Rodrigue 2001). However, the time required to achieve a reclaimed native site varied widely across all studies. In a forested system, the species composition of the oldest reclaimed sites approached that of the non-disturbed forest, but some species are not present on reclaimed sites

(Holl 2002). In contrast to these studies, we found that both native and exotic species richness decreased with time-since-reclamation and remained steady across the 40-year reclamation gradient. Yet, dominance of non-native species increased with time as illustrated by an increase in plant community dissimilarity in older sites. A study conducted in central North Dakota found similar results stating that invasive species increased over time, resulting in increased diversity, and a decrease in production values on the studied site (Wick et. al 2011).

Past research is highly variable with regards to species composition and vegetative diversity. A coalmine in southeastern Ohio showed results similar to our study, stating that little native recruitment had taken place after three decades since reclamation, and the most abundant species were invasive (Cavender 2014). Unlike community composition, the data collected in our study found plant species and alpha diversity on the reclaimed sites to be similar to those in the reference sites. Results from other research are conflicting with some reporting that plant species diversity is higher in reference sites than in restored prairies at all spatial scales studied (Polley 2005). Others are stating that the average number of plant species per transect is significantly lower in the reclaimed area than the native reference sites (Schladweiler et al. 2005). Studies show that alpha diversity decreased with time since reclamation (Kindscher and Tieszen 1998; Sluis 2002; Schladweiler et al. 2005), due to instances where native and annual species are overrun by exotic and perennial species.

Beta Diversity and 90% Patch Size

The surface coalmine disturbance began after vegetative species reached maturity. However, due to the extreme nature of mining disturbances, the study was homogenously re-spread with soil and re-seeded with a native vegetation mix. This resulted in homogeneous soil depths and plant composition on the landscape over a 40-year reclamation gradient. Creating beta diversity may be unattainable in reclamation if there is exposure to an exotic species dominance (Martin and Wilsey 2012) and homogeneous landscapes. Though, establishing native species before the exotics emerge is critical for reclaiming diverse native prairie communities where perennial exotics are present (Martin and Wilsey 2012). Multiple stable states can rise within uniform environments generating high beta diversity, if the species historical arrival order differs (Drake 1991; Chase 2003) and beta diversity could increase if the seasonal timing of disturbance is varied over the growing season (Questad and Foster 2008; Limb et al.

2010; Martin and Wilsey 2012). Differences in early-emerging species could increase beta diversity if species establish from other functional groups more readily than their own (Diamond 1975; Fox 1987; Gotelli and McCabe 2002; Martin and Wilsey 2012). It is hypothesized that early establishing species can adjust local abiotic conditions and affect community composition (Bazzaz 1996; Martin and Wilsey 2012). Natural succession is commonly reported to take 50 to 100 years to recover before a satisfactory vegetation develops especially on reclaimed coalmine sites (Bradshaw 1996; Dobson et al. 1997).

Management Implications

Proper post-reclamation management strategies play an important role in achieving succession, and stability on the newly developing plant community (Sindelar and Plantenberg 1978). To complete reclamation on mine-lands, species diversity requires practices and techniques that limit the distribution or abundances of dominant species (Howe 1994), increase species numbers, and decrease uniformity of the seed mixtures (Polley et al. 2005). Management strategies on the sites were season-long grazing or seasonal haying. No differences were observed between the management strategies, vegetation composition, and time. However, both management strategies on our sites are relatively homogeneous, which may also account for homogenous vegetation.

Conclusion

Our study found no change in species richness, alpha and beta diversity, and patch size over the 40-year reclamation gradient. The study area was dominated by invasive, exotic species even after 40-years of reclamation, illustrating how homogeneous reclamation efforts followed by homogeneous management practices may not result in heterogeneous native plant communities. Determining an outcome on reclaimed ecosystems is difficult to achieve, because of the species recruitment and the changes in vegetation composition is constantly changing on mine-lands (Wick et. al 2011). Following reclamation with proper management strategies emphasizing on minimizing invasive species may be the key to a successful ecosystem recovery.

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Literature Cited

Alday JG, Pallavicini Y, Marrs RH, Martínez-Ruiz C (2011) Functional groups and dispersal strategies as guides for predicting vegetation dynamics on reclaimed mines. *Plant Ecology* 212: 1759-1775. doi: 10.1007/s11258-011-9947-6

Bazzaz FA (1996) Plants in changing environments: Linking physiological, population, and community ecology. Cambridge University Press, Cambridge, UK.

Belyea LR, Lancaster J (1999) Assembly rules within a contingent ecology. *Oikos* 86: 402-416. doi: 10.2307/3546646

Bowen CK, Schuman GE, Olson RA, Ingram LJ (2005) Influence of topsoil depth on plant and soil attributes of 24-year old reclaimed mined lands. *Arid Land Research and Management* 19:267-284.

Bradshaw AD (1996) Restoration of mined lands- using natural processes. *Ecological Engineering* 8: 255-269. doi: 10.1016/S0925-8574(97)00022-0

Brownstein G, Steel JB, Porter S, Gray A, Wilson C, Wilson PG, Bastow-Wilson J (2012) Change in plant communities: a new approach to its measurement using the nugget from spatial autocorrelation. *Journal of Ecology* 100: 987-996.

Cavendar N, Byrd S, Bechtoldt CL, Bauman JM (2014) Vegetation communities of a coal reclamation site in southeastern Ohio. *Northeastern Naturalist* 21: 31-46. doi: 10.1656/045.021.0104

Chase JM (2003) Community assembly: when should history matter? *Oecologia* 136: 489-498. doi: 10.1007/s00442-003-1311-7

Christensen NL (1997) Managing for heterogeneity and complexity on dynamic landscapes. The Ecological Basis for Conservation: Heterogeneity, Ecosystems, and Biodiversity. New York: Chapman and Hall. doi: 10.1007/978-1-4615-6003-6_17

Cingolani AM, Vaieretti MV, Gurvich DE, Giorgis MA, Cabido M (2010) Predicting alpha, beta, and gamma plant diversity from physiognomic and physical indicators as a tool for ecosystem monitoring. *Biological Conservation* 143: 2570-2577. doi:10.1016/j.- biocon.2010.06.026

Conacher A, Conacher J (1995) Rural land degradation in Australia. Oxford University Press, Melbourne.

Dangi SR, Stahl PD, Wick AF, Ingram LJ, Buyer JS (2011) Soil Microbial Community Recovery in Reclaimed Soils on a Surface Coal Mine Site. *Soil Science Society of America Journal* 76:915-924.

Daubenmire R (1959) A canopy-coverage method of vegetational analysis. *Northwest Science* 33: 43-64.

Diamond JM (1975) Assembly of species communities. Ecology and Evolution of Communities. Harvard University Press, Cambridge, MA, USA 342-444.

Drake JA (1991) Community-assembly mechanisms and the structure of an experimental species ensemble. *American Naturalist* 137: 1-26.

Fox BJ (1987) Species assembly and the evolution of community structure. *Evolutionary Ecology* 1: 201–213. doi: 10.1007/BF02067551

Fuhlendorf SD, Engle DM (2001) Restoring heterogeneity on rangelands: Ecosystem Management Based on Evolutionary Grazing Patterns. *BioScience* 51: 625-632. doi: 10.1.641/0006-3568(2001)051[0625

Fuhlendorf SD, Engle DM (2004) Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie. *Journal of Ecology* 41: 604-614. doi: 10.1111/j.0021. -8901.2004.00937.x

Fuhlendorf SD, Engle DM, Elmore RD, Limb RF, Bidwell TG (2012) Conservation of pattern and process: developing an alternative paradigm of rangeland management. *Rangeland Ecology and Management* 65:579-589. doi: 10.2111/REM-D-11-00109.1

Fuhlendorf SD, Smeins FE (1998) Soil heterogeneity influence on plant species response to grazing within a semi-arid savanna. *Plant Ecology* 138: 89-96. doi: 10.1023/A:100970-4723526

Fuhlendorf SD, Smeins FE (1999) Scaling effects of grazing in a semi-arid grassland. *Journal of Vegetation Science* 10: 731-738. doi: 10.1023/A: 1009704723526

Gotelli NJ, McCabe DJ (2002) Species co-occurrence: a meta-analysis of J.M. Diamond’s assembly rules model. *Ecology* 83: 2091-2096. doi: 10.1890/0012-9658

Hobbs RJ, Walker LR, Walker J (2007) Integrating restoration and succession. Linking restoration and ecological succession. Springer, New York 168-179. doi:10.1111/j.1466-822x.2006.00212.x

Hobbs RJ, Jentsch A, Temperton VM (2007) Restoration as a process of assembly and succession mediated by disturbance. Linking restoration and ecological succession. Springer, New York. 150-167.

Holechek JL, Pieper RD, Herbal CH (2003) Range management principles and practices. Prentice Hall Inc., Upper Saddle River, NJ.

Holl KD (2002) Long-term vegetation recovery on reclaimed coal surface mines in eastern USA. *Journal of Applied Ecology* 39: 960-670. doi:10.1046/j.1365-2664.2002.00767

Howe HF (1994) Managing species diversity in tallgrass prairie: assumptions and implications. *Conservation Biology* 8: 691-704. doi: 10.1046/j.1523-1739.1994.08030691.x

Hurd LE, Mellinger MV, Wolf LL, McNaughton SJ (1971) Stability and diversity at three trophic levels in terrestrial successional ecosystems. *Science* 173: 1134-1136. doi:10.112 -6/science.173.4002.1134

Kindscher K, Tieszen LL (1998) Floristic and soil organic matter changes after five and thirty-five years of native tallgrass prairie restoration. *Restoration Ecology* 6: 181-196. doi: 10.1111/j.1526-100X.1998.06210

Kirkpatrick J (1994) A continent transformed: human impact on the natural vegetation of Australia. Oxford University Press, Oxford.

Lawrence Lodge RHE, Anderson BJ, de Groot A, Bill A, McQueen AAM, Steel JB, Mistral M, Mason NWH, Wilson JB (2007) Spatial autocorrelation in plant communities: vegetation texture versus species composition. *Ecography* 30: 801-811.

Li J, Duggin JA, Loneragan WA, Grant CD (2006) Grassland responses to multiple disturbances on the New England Tablelands in NSW, Australia. *Plant Ecology* 193: 39-57. doi: 10.1007/s11258-006-9247-8

Limb RF, Engle DM, Bidwell TG, Altoff DP, Anderson AB, Gipson PS, and Howard HR (2010) Restoring biopedturbation in grasslands with anthropogenic focal disturbance. Plant Ecology 210:331-342. doi: 10.1007/s11258-010-9760-7

Martin LM, Wilsey BJ (2012) Assembly history alters alpha and beta diversity, exotic-native proportions and functioning of restored prairie plant communities. *Journal of Applied Ecology* 49: 1436-1445. doi: 10.1111/j.1365-2664.2012.02202

McCune B, Grace JB, Urban DL (2002) Analysis of ecological communities. Gleneden Beach, OR: MjM software design. Vol. 28.

McIntyre S, Lavorel S, Tremont PM (1995) Plant life history attributes: Their relationship to disturbance response in herbaceous vegetation. *Journal of Ecology* 83: 31-44. doi: 10.2307/2261148

Meyers LM, DeKeyser ES, Norland JE (2014) Differences in spatial autocorreclation (Sac), plant species richness and diversity, and plant community composition in grazed and un-grazed grasslands along a moisture gradient, North Dakota, USA. Ap*plied Vegetation Science* 17: 53-62.

Mistral M, Buck O, Meier-Behrmann DC, Burnett DA, Barnfield TE, Scott AJ, Anderson BJ, Wilson JB (2000) Direct measurement of spatial autocorrelation at the community level in four plant communities. *Journal of Vegetation Science* 11: 911-916.

Moloney KA, Levin SA (1996) The effects of disturbance architecture on landscape-level population dynamics. *Ecology* 77: 375-394. doi: 10.2307/2265616

NDAWN (2015) North Dakota Agricultural Weather Network. NDAWN Station Hazen, ND. https://ndawn.ndsu.nodak.edu/station-info.html?station=28.

Newman GJ, Redente EF (2001) Long-term plant community development as influenced by revegetation techniques. *Journal of Range Management* 54:717-724.

Pattern RS, Ellis JE (1995) Patterns of species and community distribution related to environmental gradients in an arid tropical ecosystems. *Vegetation* 117: 69-79. doi: 10.1007/BF00033260

Pickett STATE, White P (1985) The ecology of natural disturbance and patch dynamics. Academic Press, Orlando.

Polley WH, Derner JD, Wilsey BJ (2005) Patterns of plant species diversity in remnant and restored tallgrass prairies. *Society of Ecological Restoration International* 13: 480-487. doi:10.1111/j.1526100X.2005.00060

Pritekel C, Whittemore-Olson A, Snow N, Moore JC (2006) Impacts from invasive plant species and their control on the plant community and belowground ecosystem at Rocky Mountain National Park, USA. *Applied Soil Ecology* 32: 132-141. doi:10.1016/j.apsoil. 2005.01.010

Questad EJ, Foster BJ (2008) Coexistence through spatio-temporal heterogeneity and specie sorting in grassland plant communities. *Ecology Letters* 11: 717-726. doi: 10.1111/j.1461-0248.2008.01186

Rodrigue JA (2001) Woody species diversity, forest and site productivity, stumpage value, and carbon sequestration of forests on mined lands reclaimed prior to the passage of the Surface Mining Control and Reclamation Act of 1977. MS Thesis. Virginia Polytechnic Institute and State University, Blacksburg, VA

Roe CM, Parker GC, Korsten AC, Lister CJ, Weatherall SB, Lawrence Lodge RHE, Wilson JB (2011) Small scale autocorrelation in plant communities: the effects of spatial grain and measure of abundance, with an improved sampling scheme. *Journal of Vegetation Science* 23: 471-482.

Schladweiler BK, Vance GF, Legg DE, Munn LC, Haroian R (2005) Topsoil depth effects on reclaimed coal mine and native area vegetation in northeastern Wyoming. *Society for Range Management* 58: 167-176. doi: 10.2111/1551-5028(2005)58<167: TDEORC>2.0.CO;2

Sindelar BW, Plantenberg PL (1978) Establishment, succession and stability of vegetation on surface mined lands in eastern Montana. Annual Program Representative., Montana Agricultural. Experiment. Station.

Skousen JG, Johnson CD, Garbutt K (1994) Natural revegetation of 15 abandoned mine land sites in West Virginia. *Journal of Environmental Qualit*y 23: 1224-1230.

Sluis WJ (2002) Patterns of species richness and composition in re-created grassland. *Restoration Ecology* 10: 677-684. doi: 10.1046/j.1526-100X.2002.01048.x

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at http://websoilsurvey.nrcs.usda.gov/. Accessed: October/29/2015.

Stirling G, Wilsey B (2001) Empirical relationships between species richness, evenness, and proportional diversity. *American Naturalist* 158: 286-299. doi: 10.1086/321317

Thompson RL, Wade GL, Straw RA (1996) Natural and planted flora of the Log Mountain surface mined. *Journal of Applied Ecology* 39: 960-970.

Vitousek PM (1990) Biological invasions and ecosystem processes: Towards an integration of population biology and ecosystem studies. *Oikos* 57: 7-13. doi: 10.1007/978. -1-4612-4018-1_17

WRCC (2015) Western regional climate historical summaries. Available at http://www.wrcc.dri.edu/cgi bin/cliMAIN.plnd1456. Accessed 22 October 2015.

Whittaker RH (1960) Vegetation of the Siskiyou Mountains, Oregon and California. *Ecological Monographs* 30: 279-338. doi: 10.2307/1943563

Whittaker RH (1972) Evolution and measurement of species diversity. *Taxon* 21: 213-251. doi:10.2307/1218190

Wick AF, Merrill SD, Toy TJ, Liebig MA (2011) Effect of soil depth and topographic position on plant productivity and community development on 28- year –old reclaimed mine lands. *Journal of Soil and Water Conservation* 66:201-211. doi:10.2489/jswc.66.3.-201

Wiens, JA (1974) Habitat heterogeneity and avian community structure in North American grassland birds. *American Midland Naturalist* 91: 195-213. doi: 10.2307/2424522

Wilson JB, Wells TCE, Trueman IC, Jones GJ, Atkinsons MD, Crawley MJ, Dodd ME, Silvertown J (1996) Are there assembly rules for plant species abundance? An investigation in relation to soil resources and successional trends. *Journal of Ecology* 84: 527-538. doi: 10.2307/2261

CHAPTER 3: A 40-YEAR CHRONOSEQUENCE OF SOIL PROPERTIES ON RECLAIMED SURFACE COAL MINELANDS

Abstract

Reclamation following anthropogenic disturbance often aims to restore stable soils that support productive and diverse native plant communities. Land reclamation regulations dictate the re-spread soil depths and grades as well as seed mixes, vegetation production standards and timelines. The soil re-spread process increases soil compaction, which may alter soil water, plant composition, rooting depths and soil organic matter. This may have a direct impact on vegetation establishment and species recruitment over time. Our objectives were to 1) quantify changes in soil compaction with regards to vegetation rooting depths, and 2) evaluate patterns of soil water across a 40-year reclamation gradient. We hypothesized that soil compaction would decrease with time since reclamation, vegetative rooting depth and soil water heterogeneity would increase over the 40-year reclamation gradient. Rooting depth, soil compaction, water content, and organic matter were recorded at 19 reclaimed and one native reference site in central North Dakota mixed-grass prairie. We determined soil compaction, rooting depths and organic matter data using a non-linear regression model. Soil water content range was analyzed using the SAS and GS+ programs to determine the range of an isotropic variogram model for each reclamation year. We rejected our hypothesis as soil compaction properties stayed steady over the 40 year reclamation gradient. This indicates that there is a lack of natural processes taking place in the soil to decrease compaction within 40 years. Soil sustainability on reclaimed lands over four decades will indicate the landscape-level success of the current ecosystem-based current reclamation strategy.

Keywords: Surface coalmine, compaction, rangelands, reclaimed soils

Introduction

Surface coalmine reclaimed soils are developing on anthropogenically-altered landscapes and are pedologically young due to the excessive removal and re-spread of the soil horizons (Sencindiver & Ammons 2000). The mining and reclamation process is an extensive procedure, and reclamation success is largely determined by comparisons of above ground vegetative cover, species diversity, and productivity between reclaimed and undisturbed reference areas (Schumann et al. 1999; Ries & Nilson 2000; Wick 2007). Following coal extraction and the reclamation process, reclaimed lands in the western United States of America are placed into a performance bond for a minimum of 10 years (SMCA 1977) until predetermined soil and vegetation parameters are satisfied. Once the land succeeds limitations, the land can be released back to previous land owners (SMCRA 1977). Even though the reclaimed land meets expectations, this does not mean the success of the ecosystem is performing at full potential. Below ground structure and function is largely overlooked in reclamation (Dangi et al. 2011), and therefore research is greatly needed in reclamation for a successful recovery of the ecosystem.

Following the mining process, heavy machinery such as Caterpillar (CAT) D11 dozers, 657 scrapers, 16M and 24M patrol blades, 789 end dump trucks and a Hitachi 2500 excavator are used to transport soil, applying this severe pressure during the reclamation process causes soil compaction (McSweeney & Jansen 1984, Chong et al. 1986; Chong & Cowsert 1997). The principal causes of compaction on reclaimed soils are from the applied normal and shear stresses of industrial equipment and trafficking (Mulholland & Fullen 1991; Davies et al. 1992; Milne & Haynes 2004; Batey & McKenzie 2006; Sinnett et al. 2006; Batey 2009). Compaction can occur on the surface or at greater depths (Batey 2009). These industrial activities severely impacts the soil health and productivity causing soil compaction at depths greater than one meter that persist up to 30 years (Spoor 2006; Batey 2009). The compaction manifests as higher bulk density, little to no soil structure, lower porosity, water holding capacity, infiltration rates, and shallower rooting depths than that of undisturbed, unmined reference sites (Indorante et al 1981; Thurman and Sencindiver 1986; Dunker & Barnhisel 2000; Shukla et al 2004a). This results in higher resistance to root penetration and reduced root elongation (Fehrenbacher et al. 1982; Thompson et al. 1987; Bradshaw 1996; Chong & Cowsert 1997). Compaction is intensified by low soil organic matter content, which can limit the rooting depth of the vegetation covering the soil by the use heavy equipment at high soil moisture contents (Batey 2009).

Soil on an intact native range is usually very heterogeneous and considered to have strong and distinct structure (Taylor & Brar 1991), but in surface coal mining the vegetation cover and soil structure is damaged by the mining process. Soil is then replaced homogeneously during the reclamation process. The goal of reclamation is to reconstruct reclaimed sites in a manner that maximizes the rate of soil development (Jansen 1981; Shukla et al. 2004b), and establishes adequate vegetation over time, returning the land back to the original heterogeneous state. Vegetation on mine sites play a vital role in improving the physical, chemical and biological properties of reclaimed sites (Bradshaw 1987). Above ground vegetation mimics below ground characteristics with regards to soil water regimes and conditions (Shukla et al. 2004b). Different plant species require different water supply rates, so having spatially heterogeneous soil water conditions can aid in producing heterogeneous plant communities (Bowen 2005). Additionally, adequate vegetation cover and production increases water infiltration rates and soil structure by plant roots, further developing a heterogeneous landscape (Bowen 2005).

Reclamation of soil compaction and initiating soil development can be left to natural processes such as vegetation establishment, and freeze-thaw cycles. However, the natural succession process is time consuming, and can take many decades before an acceptable vegetation cover establishes (Bradshaw 1996). Time since reclamation, antecedent soil properties, vegetation, and management may also affect the outcome of the reclamation process (Merrill et al. 1998; Ussirri et al. 2006). Therefore, our study evaluated soil compaction and ecosystem functions within reclaimed mine sites following 40 years of reclamation. The objectives for this study are to (1) determine the effect of soil compaction over a 40-year reclamation gradient, (2) evaluate the effect of rooting depth on compacted soils found on reclaimed mine sites, and (3) distinguish the spatial patterns of soil water contents. Hypotheses for this study are soil compaction and spatial patterns of soil water content will decrease, and vegetation rooting depths will increase over the 40 year reclamation gradient.

Methods

Experimental Design, Field and Lab Methods

Research was conducted at BNI Coal in central North Dakota approximately 5 km southeast of the town of Center in Oliver county (lat. 47°6’54 “N long 101°18’1 “W). The site is characterized as a mixed-grass prairie in the Northern Great Plains ecoregion at 602m elevation. The pre-mining soils found on the study sites fall into the great groups of Argiustolls, Haplustolls, and Haplustalfs (USDA Soil Web Survey 2016), and are considered to be loamy ecological sites after reclamation. The climate is semi-arid with annual average precipitation at 406.4 mm, the majority of which occurs from mid-April to mid-September. Average daily temperatures range between 22.8°C in early June to -16.7°C in January with a mean temperature of 21.1°C. The frost-free growing season generally ranges from 110-130 days (NDAWN 2015; WRCC 2015).

Soil data was collected in the summer of 2015 among a chronosequence of 19 post-mine reclaimed and one intact native reference site (reclamation year 1975, 1985, 1986, 1988, 1993, 1994 and 1997- 2010) creating a time-since-reclamation gradient spanning 40 years. The native reference site used in this experiment is classified as a loamy ecological site (USDA NRCS 2015). Reclaimed topsoil and subsoil were individually stockpiled until respread in reclamation. The reclamation process took place in each respective year, and each reclaimed year had top and subsoil respreads depths ranging between 61 and 121.9 cm. Mine reclamation sites were mechanically leveled prior to seeding using large CAT and Hitachi equipment, contain minimal micro-topography (<2-10% slope), and range from 2.4 to 34.8 ha in area. Post-reclamation management strategies on the sites are a combination of season-long grazing and haying.

Two 70-m transects were established in each reclamation year and reference site with a minimum of 10-m between the two transects. Soil penetrometer resistance (a measure of soil compaction and soil shear strength) was measured with a dynamic cone penetrometer at six regular intervals along one of the 70-meter transects. Penetration resistance was taken to a depth of 77 cm at each interval. Three, 154 cm deep soil cores were collected at every 14th meter interval along one 70-m transect using a Geoprobe 9800E to measure root biomass. Soil core length of 154cm was used to insure all re-spread depths were exceeded in all sites. Roots were then washed and oven dried (50°C) to a constant weight before being weighed in grams. Soil organic matter samples were collected at the same 14 meter intervals along one 70 m transect, at a depth of 0 to 15 cm. These soil samples were evaluated using the loss of ignition method (NDSU Soils Testing Lab). Lastly, soil water content range was measured at two meter regular intervals along both transects in each reclamation year. Data was recorded at two separate time intervals during the growing season (near field capacity and moderately dry conditions) using a Decegon G.S.3 sensor operating at 70 mh (Decagon).

Statistical Analysis

Penetrometer, rooting biomass and organic matter was evaluated across years using regression procedures (IMB SPSS 21). Penetration resistance is calculated as the soils resistance to halt work being done by the penetrometer divided by the distance traveled by penetrometer:

$$R_s = \frac{W_s}{P_d} \tag{3.1}$$

where R_s is the soil resistance (N), W_s is the work done by the penetrometer (J), and P_d is the distance the penetrometer traveled through the soil (equation 1) (Herrick & Jones 2001). Work equals the kinetic energy that is transferred to the cone from when the hammer strikes the strike plate. The mass of the penetrometer's drop weight is traveling at a velocity (*v*) 3.13 m s^{-1} when hitting strike plate.

$$v = \sqrt{v_0^2 + 2a(x)} = 3.13ms^{-1} \tag{3.2}$$

Where v_0 is the velocity at time 0 (*0m s* $^{-1}$), *a* is the acceleration due to gravity (9.81 ms^{-2}) and x is the change in height (0.5 m)(equation 2). Kinetic energy (KE) is then measured for the hammer of a 6.80 kg falling 50 cm is 3.4 (J) (equation 3). Kinetic energy *(KE)* was then substituted for W_s in equation 1.

$$KE = W_s = \frac{1}{2}mv^2 = 12.5\,J \tag{3.3}$$

In addition to evaluation across reclamation years using regression analysis, root biomass was analyzed by depth within year using the general linear model ANOVA with Tukey's B means separation and assuming measurement locations within a reclamation years were not spatially independent (IBM SPSS 21). Soil water content range was first averaged across both sample interval and then analyzed using the Statistical Analysis Software (SAS), and Geostatistics for the Environmental Science programs (GS+ 10) to determine the range of an isotropic variogram model for each reclamation year. Semivariograms were constructed and the nugget, sill, and range parameters were inversely parameterized using a nonlinear least squares procedure (Burnham & Anderson 2011). Semivariograms were fitted with either exponential, Gaussian, spherical, sine whole effect, cubic, or matern models based on best fit using Akaike's Information Criterion (AIC). The nugget is described as the y-intercept, sill is the model asymptote for bound models and 95% of the sample variance as the limit approaches infinity for transitional models, and the range is the separation, or lag distance over which spatial dependence is apparent (Beauchemin 2013).

Results

Soil Compaction

Penetrometer resistance did not change over the 40-year reclamation gradient (P>0.05) (Figure 1). Soil compaction slightly decreased in the top 15 cm, but stayed stationary across times in depths between 15 and 45 cm. Compaction then tends to increase with time at depths greater than 45 cm within the soil profile (Figure 2).

Root Biomass and Organic Matter

Root biomass data showed no positive relationship (P>0.05) by rooting density per year. However, there was a substantial inverse relationship of rooting density and biomass constant over all reclamation years. The reference site root biomass is the highest compared to the reclaimed sites (Figure 3). Soil organic matter (SOM) was abundant on sites ranging from 3.5% to 5.4% on reclaimed sites and 5.1% to 6.8% on reference sites however, SOM showed no differences with time-since-reclamation (Figure 3). Root biomass in the top 15 cm is substantially larger than root biomass after 15 cm depth (Figure 4). No relationship between rooting biomass and compaction over the reclamation gradient existed, indicating soil compaction is not decreasing and vegetative roots are not penetrating the compaction layers well as time progresses (Figure 5).

Soil Water Content

Soil water content (SWC) range is increasing with time (Figure 6) indicating that surface soil homogeneity is increasing as time progresses from initial reclamation. The majority of the variograms fit the sine-hole effect using SAS. The smaller the SWC range, the more heterogeneous the vegetation and soil properties of the landscape will be until a pure nugget effect is evident. The larger the SWC range the more homogeneous the landscape and dependent on topographic or rainfall patterns; not vegetative patterns. The benefit of having a smaller SWC range is that the soil properties and conditions can consume heavy rain events more efficiently.

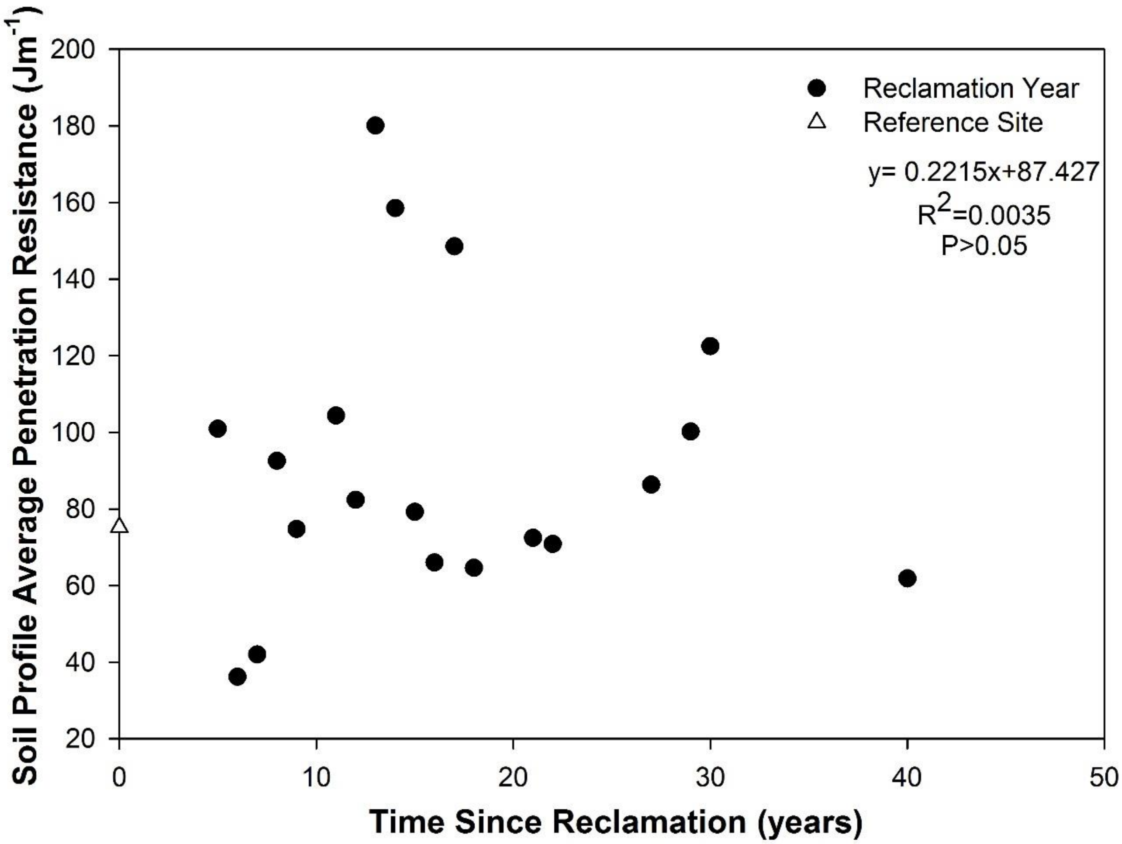

Figure 3.1: Soil profile averaged penetration resistance as a metric of soil compaction over a 40-year reclamation gradient. Older sites would be expected to have less compaction due to physical processes breaking down compaction over time. No trend found. Data collected in the summer of 2015 at BNI Coal, Center, ND.

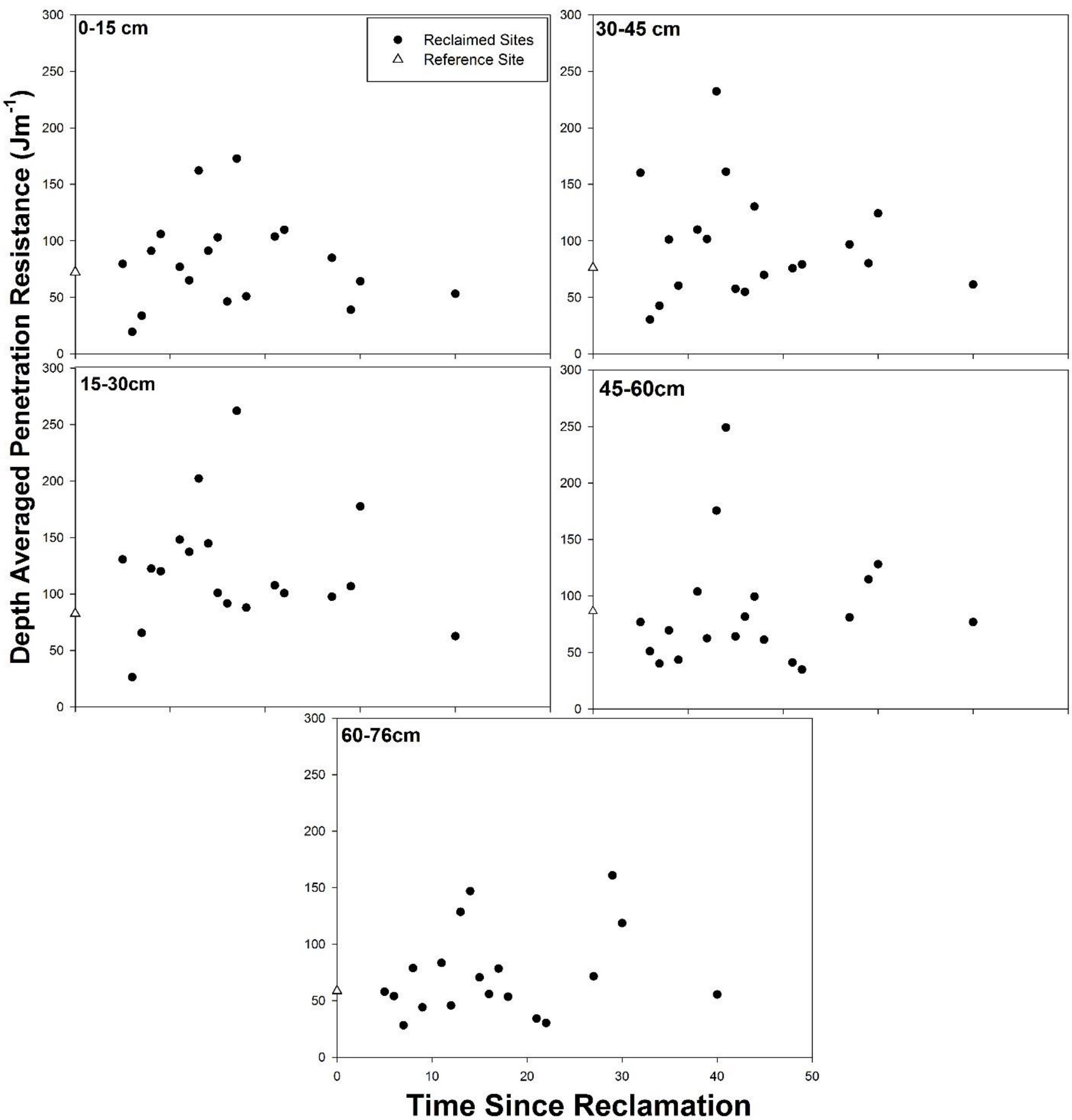

Figure 3.2: Depth averaged penetration resistance as a metric of soil compaction over the 40 year reclamation gradient. We would expect compaction to be decreasing overtime, yet data shows no trend. Data collected at BNI Coal, Center, ND.

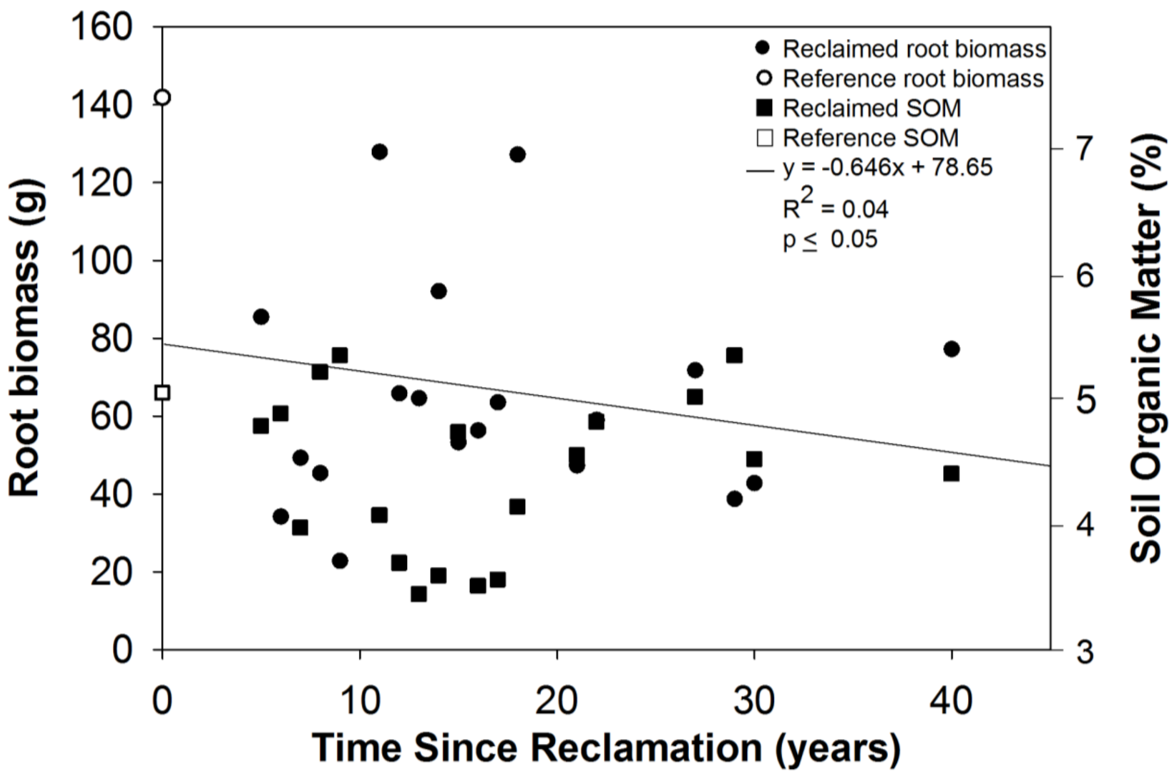

Figure 3.3: Total root biomass and organic matter (0-15 cm) composite sample per reclamation year. Older sites expected to have more root biomass and organic matter, however, root biomass seem to be decreasing over 40 years. Data collected at BNI Coal, Center, ND.

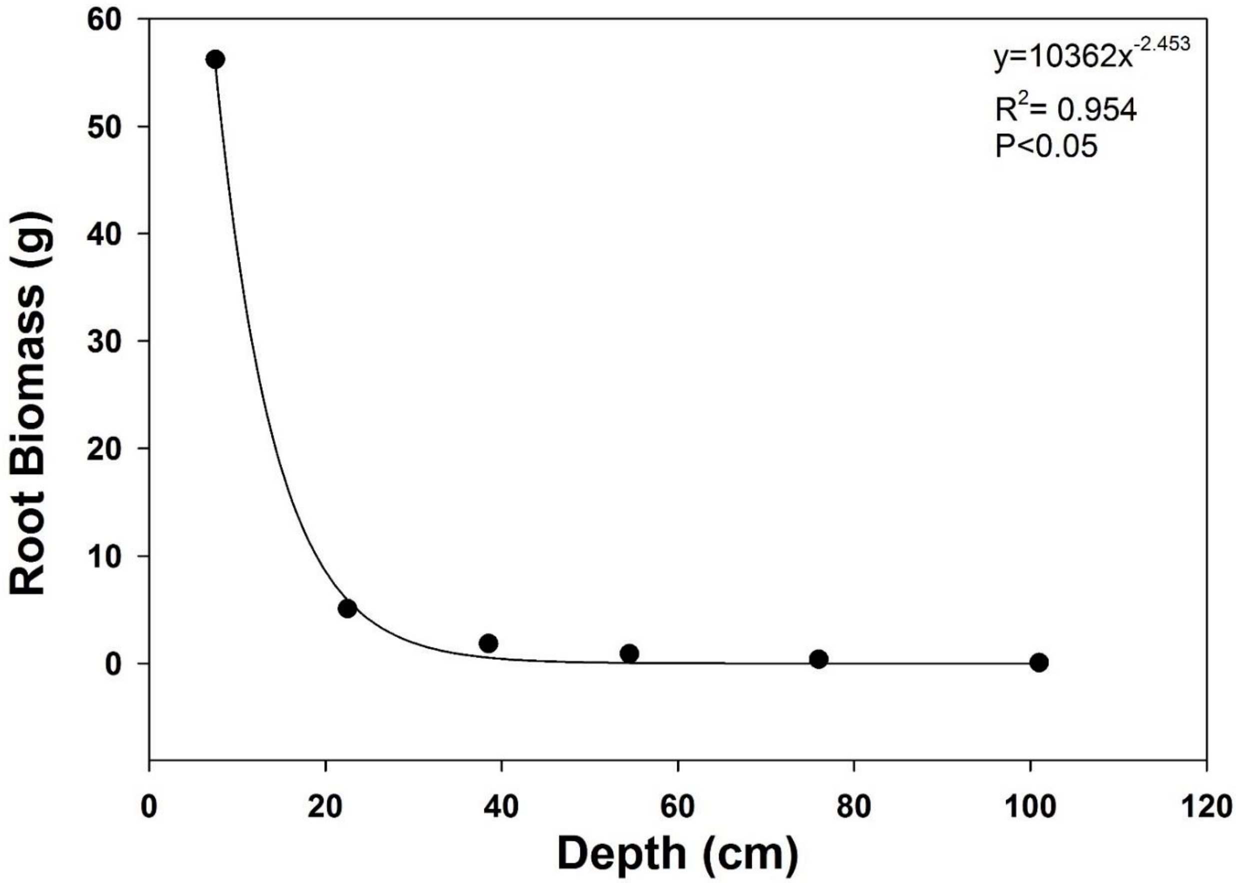

Figure 3.4: Root biomass (g) by soil depth. The farther down the soil profile, the less root biomass found. Data collected at BNI Coal, Center, ND.

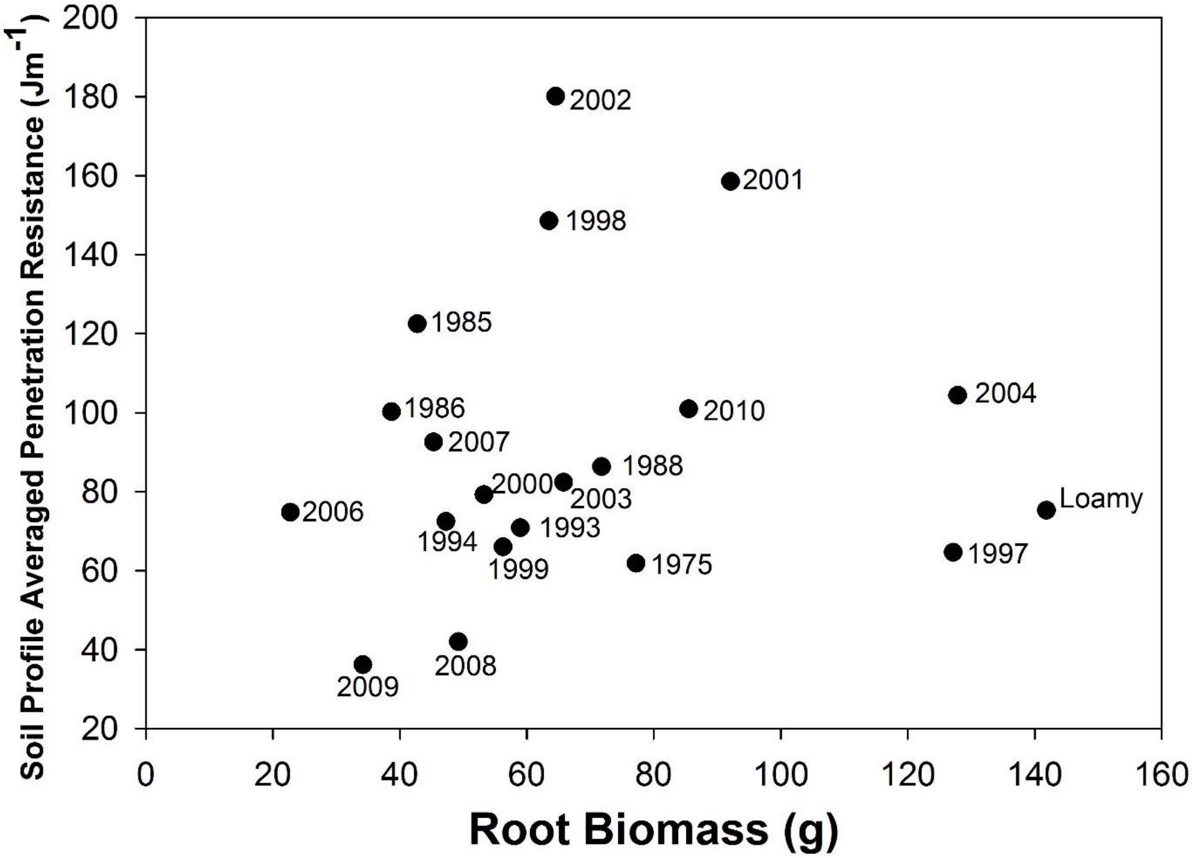

Figure 3.5: Root biomass (g) with regards to soil profile averaged penetration resistance. Older sites expected to have higher root biomass, and less soil strength. No trend found. Data collected at BNI Coal, Center, ND.

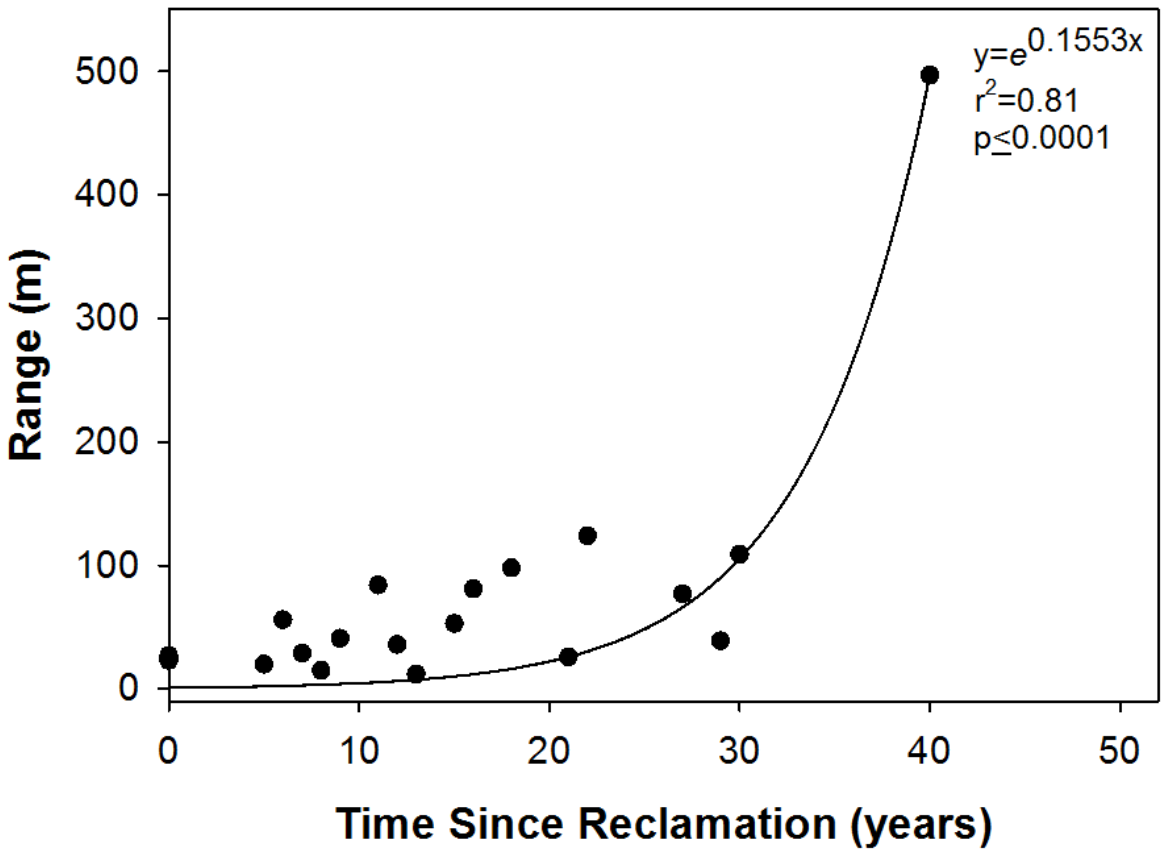

Figure 3.6: Soil water content (SWC) range over 40 years of reclamation. Expect graph to decrease over time, however, due to homogeneity of landscape vegetation, management and soil properties, SWC range is increasing. Data collected in 2015 at BNI Coal, Center, ND.

Discussion

Soil Compaction

Soil compaction is commonly found following anthropogenic disturbances that can inhibit hydrologic activity and vegetation growth. However, finding economical and feasible procedures to reduce compaction is limited in reclamation due to large equipment needed to re-spread soil. This study evaluated metrics of soil compaction, plant rooting, and the presence of soil water patterns across a 40-year reclamation gradient to determine if biotic process will improve soil conditions. Soil compaction did not decrease over time, but rather steadily persisted across the 40-year reclamation gradient. Slow soil development due to limited soil structure is common on reclaimed site with reports of soil compaction being a major limitation to post-mining productivity 28 years after reclamation (Akala & Lal 2000) with the use of heavy equipment.

Root Biomass

No significance was found for rooting depth over the chronosequence, meaning roots are not penetrating the compaction layers as time proceeds from initial reclamation. This is possibly due to the smaller diameter and type of roots in which are penetrating the soil. Soil resistance to root penetration is a measure of soil strength by compaction (Taylor 1971; Mason et al. 1988; Panayiotopoulos et al. 1994; Hamza & Anderson 2001, 2003). Root growth can be prevented if the soil strength is sufficient (Barley et al. 1965, Thompson 1987), but planting strong, thick diameter root vegetation and accumulating organic matter can reduce severe soil compaction over time (Bradshaw 1997).

Root biomass at depths vary in the literature with prior research suggesting that compacted reclaimed soils have significantly higher penetration resistance than the reference soils throughout the profile indicating the potential for restricted root growth (Bengough et al. 2011). However, some roots in a compacted layer may still be able to elongate, but at a reduced rate due to the mechanical resistance and navigation through small pores (Targieu 1994). Because little anthropogenic disturbance occurred 20 years after reclamation, root biomass improved soil structure, decreased compaction, and increased porosity of the reclaimed soils (Thompson et al. 1987; Thomas et al 2000). Data show reclaimed soils over 70 years old have deeper rooting depths, weaker soil structures, and stronger compaction than native soils (Smith et al 1971; Thomas et al. 2000) because of accelerated physical weathering and increase of organic materials during reclamation (Sencindiver & Ammons in press). Loose zones, cracks and soil macro fauna channels are preferential to roots because they provide areas of low resistance for roots to elongate (Targieu 1994), though, changing the root system will not necessarily cause a shift in above-ground growth (Taylor & Brar 1991). Much of this previous data is found in the northern hemisphere, where prominent freeze thaw cycles and adequate rainfall occur for native rangelands.

Soil Water Content

The SWC range increases over time, suggesting that the near-surface hydrology of the reclaimed landscape is becoming more homogeneous. Unpublished data by Bohrer et al. found that the vegetation on these sites are also becoming more homogeneous over time, which would explain the homogenous soil water content range. Soil compaction restricts hydrologic processes that inhibit water and air movement through the soil, impeding root and plant growth (Gifford et al 1977; Thompson et al 1987; Lipiec et al. 2002; Hamza & Anderson 2005). Infiltration rates and vegetation cover or standing biomass are typically are positively correlated (Rauzi et al. 1968; Thurow et al. 1988a; Blackburn et al. 1992). Adverse soil conditions such as poor soil structure, compaction, low soil organic matter content, and low water retention often limit vegetation establishment, and constrain reclaimed sites to a homogeneous landscape (Ussiri et al 2006).

Organic Matter

High infiltration rates and water retention in undisturbed forest compared to reclaimed mine sites may be accredited to the low soil compaction, high porosity, high soil organic matter and larger quantities of macro soil fauna (Wuest 2001). Organic matter additions are known to increase infiltration rate, and water retention (Benbi et al. 1998), while decreasing bulk density and compaction (Schjonning et al. 1994; Watts & Dexter 1998; Hatley et al. 2005). Organic matter levels observed in the analysis were abundant and not considered a limiting factor for plant productivity and soil health. An earlier study found that during a 20-year reclamation period, reclaimed mine soils accumulated large amounts of organic matter that account for the structural development in the mine soils. (Underwood & Smeck 2002). Reclaimed plant communities that are more productive than native communities (Wick et al. 2007) could result in greater contributions of organic matter to the soil (Stahl et al. 2003; Wick et al. 2008).

Management Implications

Management strategies that aim to increase soil function, sustainability, native vegetation stands and overall ecosystem health should be the main focus in post-reclamation. Previously there has not been a practical way of re-aggregating soil particles for better rooting depths, except by biological means. However, a mechanical management strategy that may help decrease soil compaction and increase rooting depth on mine reclamation sites is soil ripping. Soil ripping uses deep tines to an approximate depth of 75 cm, however, the operation is costly (Bradshaw 1997) and can lead to the reversal of soil structure in non-reclamation sites absent of severe soil compaction. Ensuring vigorous plant growth for proper organic matter is an important concept for suitable soil structure. Root penetration in a variety of vegetative species has been shown to correlate with an increase in root diameter (Materechera et al. 1991; Targieu 1994). Planting a variety of native species, forbs and grasses, may help with soil compaction on reclaimed sites. Physical processes such as freeze thaw cycles, wetting-drying cycles will gradually alleviate soil compaction on reclaimed lands, but on many sites the severity of the compaction may be a limiting factor (Bradshaw 1997). Using cultivation, deep ripping, or adding a mulch mixture to prepare a proper seed bed for vegetation may be essential to kick start the successful reclamation of respreads mine soils.

Conclusion

Soil compaction remained prominent in soils even after 40 years since reclamation. Vegetative roots are not thick, or effectively penetrate the compaction layers to increase rooting depths of above ground vegetation and create heterogeneous soil water content regimes on the landscape. Incorporating soil ripping and mulching mechanisms within the soil profile may help speed up the alleviation of soil compaction and development of a heterogeneous functional landscape. Increasing vegetative diversity above ground may also play a key role in changing the below ground properties to decrease compaction and return these mined lands back to native production rangelands within a timely manner.

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Literature Cited

Andersen MN, Munkholm LJ, Nielsen AL. 2013. Soil Compaction limits root development, radiation-use efficiency and yield of three winter wheat (*Triticum aestivum* L.) cultivars. *Soil and Plant Science* 63: 406-419. DOI: 10.1080/09064710.2013.789125

Batey T, McKenzie DC. 2006. Soil compaction: identification directly in the field. *Soil Use and Management* 22: 123–131. DOI: 10.1111/j.1475-2743.2006.00017.x

Batey T. 2009. Soil compaction and soil management- a review. *Soil Use and Management.* 25: 335-345. DOI: 10.1111/j.1475-2743.2009.00236.x

Beauchemin M. 2013. Image thresholding based on semivariance. *Pattern Recognition Letters* 34: 456-462. DOI:10.1016/j.patrec.2012.11.017

Benbi DK, Biswas CR, Bawa SS, Kumar K. 1998. Influence of farmyard manure, inorganic fertilizers and weed control practices on some soil physical properties in a long-term experiment. *Soil Use and Management* 14: 52-54. DOI: 10.1111/j.14752743.1998.tb00610.x

Blackburn WH, Pierson FB, Hanson CL, Thurow TL, Hanson AL. 1992. The Spatial and Temporal Influence of Vegetation on Surface Soil Factors in Semiarid Rangelands. *Transactions of the American Society of Agricultural and Biological Engineers* 35(2): 479- 486.

Bowen CK, Schuman GE, Olson RA, Ingram J. 2007. Influence of Topsoil Depth on Plant and Soil Attributes of 24- Year Old Reclaimed Mined Lands. *Arid Land Research and Management* 19: 267-284. DOI: 10.1080/15324980590951441

Bradshaw AD. 1984. Land restoration: now and in the future. *Proceedings of the Royal Society of London* 223: 1-23. DOI: 10.1098/rspb.1984.0079

Bradshaw AD. 1997. Restoration of mined lands- using natural processes. *Ecological Engineering* 8: 255-269. DOI: 10.1016/S09258574(97)000220

Burnham KP, Anderson DR. 2011. AIC model selection and multimodel inference in behavioral ecology: some background, observations and comparisons. *Behavioral Ecology and* Sociobiology 65:23-35. DOI: 10.1007/s00265-010-1029-6

Cavendar NS, Byrd SB, Bechtoldt CL, Bauman JM. 2014. Vegetation Communities of a Coal Reclamation Site in Southeastern Ohio. *Northeastern Naturalist* 21(1):31-46. DOI: 10.1656/045.021.0104

Chong SK, Becker MA, Morre SM, Weaver GT. 1986. Characterization of mined land with and without topsoil. *Journal Environmental Quality* 15:157-160. DOI: 10.2134/jeq1986.00 472425001500020014x

Chong SK, Cowsert PT. 1997. Infiltration in reclaimed mined land ameliorated with deep tillage treatments. *Soil and Tillage Research* 44: 255-264. DOI: 10.1016/S0167-1987(97)00050-0

Dangi SR, Stahl PD, Wick AF, Ingram LJ, Buyer JS. 2011. Soil Microbial Community Recovery in Reclaimed Soils on a Surface Coal Mine Site. *Soil Science Society of America Journal* 76: 915-924. DOI:10.2136/sssaj2011.0288

Davies DB, Finney JB, Eagle DJ. 1992. *Soil management*, 5th edn. Farming Press Ltd, Ipswich, England.

Dobson AP, Brashaw AD, Baker AJM. 1997. Hopes for the future: Restoration ecology and conservation biology. *Science* 277:515-522. DOI: 10.1126/science.277.5325.515

Dunker RE, Barnhisel RI. 2000. Cropland reclamation. In: Barnhisel RI, Darmody RG, Daniels WL (eds) Reclamation of drastically disturbed lands. *Soil Science Society of America*, Madison, WI, USA, 323–369

Fehrenbacher DJ, Jansen IJ, Fehrenbacher JB. 1982. Corn root development in constructed soils on surface-mined land in Western Illinois. *Soil Science Society of America* 46: 353-359. DOI:10.2136/sssaj1982.03615995004600020028x

Gifford FG, Faust RH, Coltharp GB. 1977. Measuring Soil Compaction on Rangeland. *Journal of Range Management* 30: 457-460. DOI: 10.2307/3897812

Godwin R, Spoor G, Finney B, Hann M, Davies B. 2008. The current status of soil and water management in England. Royal Agricultural Society of England, Stoneleigh, Warwick.

Hakansson I. 1994. Subsoil compaction caused by heavy vehicles –a long term threat to soil productivity. *Soil and Tillage Research* 29:105–110.

Hamza MA, Anderson WK. 2005. Soil compaction in cropping systems- A review of the nature, causes and possible solutions. *Oil and Tillage Research* 82: 121-145. DOI: 10.1016/ j.still.2004.08.009

Hatley D, Wiltshire J, Basford B, Royale S, Buckley D, Johnson P. 2005. Soil compaction and potato crops. Research Review R260, British Potato Council, Oxford, UK.

Hobbs RJ, Harris JA. 2001. Restoration ecology: repairing the earth’s ecosystem in the new millennium. *Restoration Ecology* 9:239-246. DOI: 10.1046/j.1526100x.2001.009002239.x

Holl KD. 2002. Long-term vegetation recovery on reclaimed coal surface mines in eastern USA. *Journal of Applied Ecolog*y 39: 960-670. DOI: 10.1046/j.1365-2664.2002.00767.x

Indorante SJ, Jansen IJ, Boast CW. 1981. Surface mining and reclamation: Initial changes in soil character*. Journal of Soil Water Conservation* 36:347–351

Jansen IJ. 1981. Reconstructing soils after surface mining of prime agriculture land. *Mining Engineering* 33:312–314.

Kondo MK, Dias MS Jr. 1999. Soil compressibility of three latosols as a function of moisture and use. *Revista Brasileira de Ciencia do Solo* 23: 211–218. DOI: 10.1590/S0100-068319 99000200004

Laws, Regulations and Guidance, Office of Surface Mining Reclamation and Enforcement, Available online at http://www.osmre.gov/lrg.shtm. Accessed [01/20/2016]

Lipiec J, Ferrero A, Giovanetti V, Nosalewicz A, Turski M. 2002. Response of structure to simulated trampling of woodland soil. *Advance Geoecol*ogy 35:133–140.

Materechera DA, Dexter AR, Alston AM. 1991. Penetration of very strong soils by seedling roots of different plant species. *Plant Soil* 135: 31-41.

McSweeney K, Jansen IJ. 1984. Soil structure and associated rooting behavior in minesoils. *Soil Science Society of America Journal* 48: 607-612. DOI:10.2136/sssaj1984.036159950048 00030028x

Medvedev VV, Cybulko WG. 1995. Soil criteria for assessing the maximum permissible ground pressure of agricultural vehicles on Chernozem soils. *Soil Tillage Research* 36: 153–164. DOI: 10.1016/0 167-1987(95)00501-3

Merrill SD, Power JF, Ries RE. 1998. Subsoil characteristics and landscape position affect productivity of reconstructed minesoils. *Soil Science Society of America* 62:263–271. DOI:10.2136/sssaj1998.03615995006200010034x

Milne RM, Haynes RJ. 2004. Comparative effects of annual and permanent dairy pastures on soil physical properties in the Tsitsikamma region of South Africa. S*oil Use and Management* 20:81–88. DOI: 10.1111/j.1475-2743.2004.tb00340.x

Mulholland B, Fullen MA. 1991. Cattle trampling and soil compaction on loamy sands. S*oil Use and Management* 7:189–193. DOI: 10.1111/j.1475-2743.1991.tb00873.x

North Dakota State University Soils Testing Lab,Available online at https://www.ndsu.edu/soils/services /soil_testing_lab/. Accessed [11/05/2015]

North Dakota State Public Service Commission, Available online at http://www.psc.nd.gov/. Accessed [12/18/2015]

Pedersen TA, Rogowski AS, Pennock R Jr. 1980. Physical characteristics of some minesoils. *Soil Science Society of America* 44: 321-328. DOI:10.2136/sssaj1980.036159950044000200 24x

Perry TO. 1982. The ecology of tree roots and the practical significance thereof. *Journal of Arboriculture* 8: 197-211.

Ross CW. 1986. The effect of subsoiling and irrigation on potato production. *Soil and Tillage Research* 7: 315-325. DOI: 10.1016/0167-1987(86)90019-X

Rauzi F, Fly DL, Dyksterhuist EJ. 1968. Water intake on midcontinental rangelands as influenced by soil and plant cover. USDA Technical Bulletin 1390.

Schjonning P, Christensen BT, Carsten B. 1994. Physical and chemical properties of a sandy loam receiving animal manure, mineral fertilizer or no fertilizer for 90 years. *European Journal of Soil Science* 45: 257-268. DOI: 10.1111/j.1365-2389.1994.tb00508.x

Sencindiver JC, Ammons JT. 2000. Minesoil genesis and classification. In: Barnhisel RI, Daniels WL (eds) Reclamation of drastically disturbed lands. Agronomy monograph no. 41. *Soil Science Society of America*, Madison, WI, 595–613.

Shukla MK, Lal R, Ebinger M. 2004a. Soil quality indicators for reclaimed mine soils in southeastern Ohio. *Soil Science* 169:133–142

Shukla MK, Lal R, Underwood J, Ebinge M. 2004b. Physical and Hydrological Characteristics of Reclaimed Minesoils in Southeastern Ohio. *Soil Science Society of America* 68:1352-1359. DOI:10.2136/sssaj2004.1352

Sinnett JD, Poole J, Hutchings TR. 2006. The efficacy of three techniques to alleviate soil compaction at a restored sand and gravel quary. *Soil Use and Management* 22: 362-371. DOI: 10.1111/j.1475-2743.2006.00053.x

Smith RM, Tyron EH, Tyner EH. 1971. Soil development on mine spoil. West Virginia Agricultural Experiment Station Bulletin 604T

Soil Moisture Sensors, Decagon Devices Incorporated, https://www.decagon.com/en/soils/volumetric-water-content-sensors/gs3-vwc-temp-ec/. [Accessed on 04/13/2015]

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Web Soil Survey. Available online at http://websoilsurvey.nrcs.usda.gov/. Accessed [01/15/2016].

Spoor G. 2006. Alleviation of soil compaction: requirements equipment and techniques. *Soil Use and Management* 22:113–122. DOI: 10.1111/j.1475-2743.2006.00015.x

Tardieu F. 1994. Growth and Function of roots and of root systems subjected to soil compaction. Towards a system with multiple signaling? *Soil and Tillage Research* 30: 217-243. DOI: 10.1016/0167-1987(94)90006-X

Taylor HM, Brar GS. 1991. Effect of soil compaction on root development. *Soil and Tillage Research* 19: 111-119. DOI: 10.1016/0167-1987(91)90080-H

Thomas KA, Sencindiver JC, Skousen JG, Gorman JM. 2000. Soil Horizon Development on a Mountaintop Surface Mine in Southern West Virginia. *Green Lands* 41-52.

Thompson PJ, Jansen IJ, Hooks CL. 1987. Penetrometer resistance and bulk density as parameters for predicting root system performance in mine soils. *Soil Science Society of America* 51: 1288-1293. DOI:10.2136/sssaj1987.03615995005100050035x

Thurman NC, Sencindiver JC. 1986. Properties, classification, and interpretations of mine soils at two sites in West Virginia. *Soil Science of America* 50:181–185. DOI:10.2136/sssaj1986.0 3615995005000010034x

Thurow TL, Blackburn WH, Taylor CA Jr. 1986. Hydrologic characteristics of vegetation types as affected by livestock grazing systems, Edwards Plateau, Texas. *Journal of Range Management* 39:505-509. DOI: 10.2307/3898758

Ussiri DAN, Lal R, Jacinthe PA. 2006. Soil Properties and Carbon Sequestration of Afforested Pastures in Reclaimed Minesoils of Ohio. *Soil Science Society of America* 1797-1806. DOI:10.2136/sssaj2005.0352

Van den Akker JHH, Arvidsson J, Horn R. 2003. Introduction to the special issue on experiences with the impact and prevention of subsoil compaction in the European Union. *Soil and Tillage Research* 73:1–8. DOI: 10.1016/S0167-1987(03)00094-1

Vomocil JA. 1957. Measurement of soil bulk density and penetrability: A review of method. *Advance Agronomy* 9: 159-176.

Wuest SB. 2001. Earthworm, infiltration, and tillage relationships in a dry land pea–wheat rotation. *Applied Soil Ecology* 18:187-192. DOI: 10.1016/S0929-1393(01)00145-7

Wick AF, Ingram LJ, Stahl PD. 2009. Aggregate and organic matter dynamics in reclaimed soils as indicated by stable carbon isotopes. *Soil Biology and Biochemistry* 41:2 201-209.

Wick AF, Stahl PD, Ingram LJ. 2009. Aggregate-Associated Carbon and Nitrogen in Reclaimed Sandy Loam Soils. *Soil Science Society of America Journal* 73:6: 1852-1860

Wilhelm WW, Johnson JMF, Hatfield JL, Voorhees WB, Linden DR. 2004. Crop and soil productivity response to corn residue removal: A literature review. *Agronomy Journal* 96**:** 1-17. DOI:10.2134/agronj2004.1000