Assessing the use of biotic and abiotic soil remediation for the restoration of temperate meadow ecosystems

by

Martin Kastner

A thesis
presented to the University of Waterloo
in fulfilment of the
thesis requirement for the degree of
Master of Environmental Studies
in
Environment and Resource Studies

Waterloo, Ontario, Canada, 2014

Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

While the extent of grasslands in Southern Ontario has been greatly reduced, urban and suburban areas provide numerous potential sites for their restoration. Grassland restoration in cities can provide ecological and cultural benefits, but soil conditions may be less than optimal for native species recovery. This thesis explores the use of soil amendments in order to address nutrient deficiency on old-field meadow restoration sites. Five treatments were tested, namely the addition of (1) nitrogenous fertilizer, (2) native legume species, (3) biochar, (4) a combination of the previous three, and (5) an unaltered control. Each treatment was replicated four times on two different test plots in Waterloo Region, Ontario, Canada (Huron Natural Area and Springbank Farm), for a total of 40 subplots. The experimental plots were tilled in the fall of 2011, a randomly selected treatment was applied to each subplot, and then all were seeded with a mix of five native meadow species (2 grass, 2 forb, 1 sedge). Soil samples were taken from each subplot both before treatment application and also at the end of the growing season in 2012, and tested for nutrient levels (N, P, K), pH and organic matter. Species richness, as well as soil temperature and moisture, were regularly monitored over the growing season. In the fall of 2012, above-ground vegetation was harvested to assess accrued biomass. In order to detect differences in means, results were tested using one-way and repeated-measures ANOVAs, where appropriate. Pearson's product-moment correlations were also employed to test for linear dependence between variables.

There were no significant differences between treatments in terms of soil nutrients or pH at either site. At Huron Natural Area, post-treatment biochar-treated subplots had slightly higher levels of organic matter than controls ($p=0.095$). Values for species richness, above-ground biomass, soil temperature and soil moisture did not vary significantly between treatments. Species richness at Huron Natural Area was positively correlated with 2011 N ($r=0.42$; $p=0.07$) and organic matter ($r=0.52$; $p=0.02$) levels, while at Springbank Farm it was negatively correlated with 2012 N levels ($r=-0.67$; $p<0.001$). Above-ground biomass at Huron Natural Area was positively correlated with 2011 and 2012 P levels (both $r=0.52$; $p=0.02$), while at Springbank Farm it was positively correlated with 2011 N, P, K and organic matter, and 2012 N, P and K (all $r>0.44$; $p<0.05$). At Huron Natural Area, above-ground biomass was negatively correlated with soil temperature ($r=-0.64$; $p<0.0001$) and positively correlated with soil moisture ($r=0.38$; $p=0.1$). This study uncovered a strong, but variable, relationship between N concentration and species richness in old-field meadows. Furthermore, productivity was tightly correlated with different soil nutrient concentrations at each study site. The results demonstrate the need for restoration approaches to address local soil conditions on order to be effective. To date, there have been very few studies on meadow restoration, particularly in North America. More, and longer-term, multivariate studies are needed in order to test the effectiveness of different techniques.

Acknowledgements

I was lucky to have so many colleagues willing to lend me a hand in the field: Justin Hick, Natasha Lukey, Andrea Harrington, Amanda Bichel, Steve Yessie, Cristóbal Pizzaro, Jeff Sutherland, as well as undergraduate volunteers Sonya Oetterich and Monica Kilic. Lee Voisin also played an important part in collecting data over the summer of 2012.

Thank you to my friend Matt Quick, who taught me the rudiments of R.

The co-op students at the Ecology Lab were always enthusiastic in helping me with my labwork. Thanks to Marcus Maddalena, Calida de Jong, Ben Watson, Bennett Hannam, Sonya Cleland and Mathure Sivananthan.

I appreciate the advice of Prof. Maren Olbermann, who met with me early in the planning process. I am also indebted to Josh Shea, at the City of Kitchener, and Amanda Newell, at *rare*, who helped coordinate the logistics of the project. Thank you to Jeff Thompson at Native Plant Source for providing the seeds for the plantings, and Peter Fransham at Abri Tech for the biochar.

Many thanks to Prof. Merrin Macrae, who kindly agreed to provide feedback on this manuscript on short notice.

Finally, this project would have been impossible without the guidance of Prof. Steven Murphy, an ecological genius in his own right. He was always generous with his time and counsel, while allowing me the space and freedom to learn from my own mistakes and successes.

I was fortunate to receive funding from an Ontario Graduate Scholarship, a *rare* Scholarship in Graduate Research, and the University of Waterloo.

Dedication

- *To my grandfather,*

Antonio Ocaña Carmona,

a constant inspiration

who will always be remembered

fondly.

Table of Contents

List of Figures

List of Tables

Chapter 1
Introduction

1.1 Introduction

Conservation efforts have long focused on preserving the biodiversity of tracts of wilderness relatively free from human influence. The hope was that by protecting these pristine areas, people would be able to enjoy their scenery and species for perpetuity. As anthropogenic activities increasingly stretch to the furthest reaches of the globe, there has been a realization that such pristine areas are too few and far between, too small, and not adequately representative of the world's ecosystems to prevent the irreversible loss of species and ecosystem function (Newmark, 1987; Soulé and Terborgh, 1999). There is a pressing need to complement such traditional approaches with the conservation and restoration of natural landscapes in human-settled areas, whose expansion, often in ecologically significant areas, is a prevailing source of land-use change worldwide (Harris, 2010; Miller & Hobbs, 2002; Rudd, Vala & Schaefer, 2002). Indeed, the Convention for Biological Diversity's new strategic plan, arising from the Nagoya Conference in 2010, has among its targets not only to eliminate the loss of all natural habitats, but also to restore at least 15% of all degraded ecosystems (Rey Benayas & Bullock, 2012).

Restoration in urban and suburban areas can be fraught with difficulties: natural areas are isolated in expanses of asphalt, concrete and lawn; fast-moving and polluted stormwater can have detrimental impacts on waterways and wetlands; elevated levels of nutrients, heavy metals and toxic chemicals damage the soils (Craul, 1992). Despite these and other impediments, urban restoration has benefitted from the proximity to centres of learning, the existence of large pools of volunteer labour and the availability of funding (Ingram, 2008). Urban restoration is necessary for the promotion of landscape connectivity and numerous indispensible ecosystem services, such as flood control, temperature moderation, air purification and the protection of groundwater sources (Forman, 2008; Pavao-Zuckerman, 2008). It also has the potential to engender psychological wellbeing, greater social inclusion and environmental justice (Light & Higgs, 1995; Newman, 2003; Harris, 2010). Indeed, it has been suggested that urban ecological restoration is the key to a new "communion" between our species and the natural world (Jordan, 2003).

Any restoration project must take into consideration the history of the site to be restored. In urban areas, these histories can be formidably complex, adding layers of difficulty to both the planning and the execution of restoration projects (Craul, 1992). The "ecological memory" of such sites will have been lost to an extent, through the removal of the topsoil and its associated seed-bank, as well as changes in hydrology and nutrient cycling (Schaefer, 2009). While in denser urban areas, the ecosystems of sites to be restored may be entirely novel (Hobbs et al., 2013), as one moves towards the outskirts of cities they may increasingly resemble "traditional" successional landscapes, such as those seen on abandoned agricultural fields (Cramer, Hobbs & Standish, 2008).

Peri-urban and urban sites are increasingly becoming available for ecological restoration projects, as the trend of agricultural land abandonment continues to accelerate worldwide (see section

2.2.3; Cramer, Hobbs & Standish, 2008). It will be important to integrate natural areas into the fabric of growing urban areas through the opportunistic restoration of derelict sites, abandoned fields, lawns and architectural elements, such as green roofs and walls. In the American "rust belt", there has been a movement towards "shrinking cities", as population levels drop and the cost of maintaining infrastructure becomes prohibitive. Ecological restoration, given its relative cost-effectiveness and the many benefits it can provide, is being considered as one of the options for these newly available acreages (LaCroix, 2010). Indeed, as the concepts of resilience and green infrastructure – and the links between them – gain mainstream traction, cities are including major restoration projects into their long term planning (Miller, 2013).

The ecosystems of southern Ontario are the most diverse in the province, but they are also the most fragmented and degraded. The decline of the extent of grasslands in this region is remarkable: more than 97% of tallgrass prairie and 99% of savannahs have been lost to agricultural and urban development, and dozens of species have been extirpated (Bakowsky & Riley, 1994; Bell & DeMarco, 1999; Faber-Langendoen & Maycock, 1992). Ecological restoration in this part of the province is crucial in order to provide adequate habitat for plant and wildlife species. Meadows, as fertile, early-successional grasslands, are important habitats for many species of wildflowers, birds and pollinating insects, many of which are provincially and even nationally rare. In this region, meadows are generally located in abandoned fields that are transitioning towards either protection (and possible reforestation) or development (Milne & Bennett, 2007).

While old-fields do provide crucial habitat for meadow species, their soil conditions can be less than optimal for the establishment of a diverse native plant community. It is crucial to address soil biotic and abiotic conditions when planning to restore a site (see section 2.2.2; Bradshaw, 1997). Altered or depleted seed banks or soil faunal communities are biotic factors that can favour invasive exotic species and hamper the establishment of desired successional ecosystem, while compaction, nutrient excess or deficiency and altered hydrology are abiotic factors that can have the same effect (Cramer, 2007; Kardol, Bezemer & Van Der Putten, 2009).

Depending on the conditions of the site as well as those of its surroundings, an old-field may adopt any number of trajectories following abandonment, including reverting to its previous "natural" state or transitioning to any number of novel states (Suding & Hobbs, 2009). In some cases, especially when abiotic thresholds have been crossed, soil amendments can be used towards re-establishing the conditions that will allow the recovery of a desired plant community on old-fields (Cramer et al., 2008; Harris, 2009). The use of soil amendments in the restoration of grasslands is informed by their use in agriculture: in that context, products like mineral fertilizers, compost, manure and lime are used to enrich the soil or alter its acidity (Thompson & Troeh, 1973). In restoration, the goal is to attain the optimal soil conditions for the ecosystem of interest – in fact, the goal in many cases is to reverse a site's agricultural legacy. A multitude of products have been tested in the field: fertilizers, legume plants and sewage sludge can all boost site fertility (Andrés, 1999; Dancer, Handley, & Bradshaw, 1977b, 1979; Foster & Gross, 1998), while sugar or sawdust immobilize excess nutrients (Blumenthal, Jordan & Russelle, 2003). Mycorrhyzal inoculation has shown some positive effects on plant community establishment (Fischer et al., 2013*a*). Biochar – finely powdered pyrolized organic material – is gaining in popularity in the agricultural setting for its positive effects on water retention, nutrient cycling (Sohi et al., 2009) and soil biota, including symbiotic mycorrhizae

(Lehmann et al., 2011). Significant increases in crop yield are frequently reported in agricultural settings (Lehmann & Joseph, 2009). The use of biochar has not been extensively tested in the context of ecological restoration, but one experiment indicated positive effects on native plants establishment on ex-arable fields dominated by non-native species (Kulmatiski, 2011), possibly thanks to a capacity to inhibit allelopathy by sequestering allelochemicals.

In my research I investigate the effects of three promising soil amendments that target nutrient deficiency in a restoration context, namely 1) the application of chemical fertilizers, 2) the establishment of native legume species and 3) the addition of biochar, independently and in combination, on physical and chemical soil properties, as well as plant establishment.

1.2 Problem statement

Restoration projects in cities often focus mainly on a site's flora, aiming to eradicate exotic invasive species in order to then re-establish native plant communities. Such approaches are often ineffective as they disregard the soil conditions that can lead to a competitive imbalance between native and invasive plants. Evidence suggests that adopting an integrative approach that addresses soil conditions could be more successful (Li & Norland, 2001; Pavao-Zuckerman, 2008). Indeed, Bradshaw (1997) suggests that "the first problem to be addressed in restoration is the way in which the restoration of the soil is to be tackled." Soils in urban areas are variable and are often highly disturbed (Craul, 1992; de Kimpe and Morel, 2000; Heneghan et al., 2008).

Many decades of research into the dynamics of old-fields, which has informed succession theory and the field of restoration ecology to a great degree, demonstrate that when nutrient and water cycles are permanently altered and natural vegetation in the surrounding landscape is highly fragmented, historical plant communities are unlikely to recover (Cramer, Hobbs & Standish, 2008). For fields in an urban or peri-urban setting, the legacy of cultivation or industrial activity on the land is likely to be compounded by biotic and abiotic stresses inherent to their immediate environs.

There are a multitude of benefits to conducting ecological restoration in an urban setting (see sections 1.0 and 2.2.3). Historical and recent research demonstrates that soil amendments are important tools for use in restoration (Harris, 2009), and the use of biochar in particular has implications which stretch into the realms of climate change science and global economics, as it is known to act as an effective carbon sequestration tool (Lehmann & Joseph, 2009). My research sits at the intersection of the fields of restoration ecology, urban ecology and soil science (see Section 1.5). There is a sense of urgency inherent to the field of restoration ecology given its direct relevance towards practical application, which is compounded when the work is conducted in areas where it stands to benefit large numbers of people. Important gaps still exist in the literature regarding the practical application and comparative utility of different soil amendments in the establishment of native vegetation species. This is the problem that this experiment will address directly, within the restrictions inherent to its duration, breadth and context.

1.3 Reasoning for treatment selection

A preliminary study by Murphy et al. (2010) indicated that soils in the Huron Natural Area meadow might be nitrogen deficient. On the assumption that soil conditions would be similar at both field sites, it was decided that the same treatments would be replicated at both sites. Treatments were selected that would be expected to increase soil nitrogen concentration or availability, by different mechanisms. They were limited to four in number for the practical reasons of limited space and manpower.

- **Chemical fertilizer:** This treatment increases soil nitrogen concentration by providing it directly in plant available form (ammonium sulphate and urea). Soil nitrogen concentration at Huron Natural Area was ~12 kg/ha in 2010 (Murphy, unpublished), and ideal concentration in a temperate meadow ecosystem is ~80 kg/ha (Bradshaw, 1992). The quantity of fertilizer that would provide ~68 kg/ha per subplot was therefore computed (see Section 3.3).
- **Native legume planting:** This treatment should increase soil nitrogen concentration through the process of atmospheric fixation. Symbiotic bacteria in legume root nodules can fix between 50-200 kg N/ha/year, although this figure is variable and dependent on species, plant age, soil conditions and weather (Bradshaw, 2002). *Lupinus perennis* L. and *Astragalus canadensis* L. were selected as species appropriate to the biome, and treated with genus-appropriate rhizobia.
- **Biochar:** This treatment does not directly add nitrogen to the soil, but there is evidence that it can alter nitrogen retention and use-efficiency. It is has a complex pore structure and high surface area, and has been shown to alter soil cation exchange capacity (Sohi et al., 2009). The adsorption of ammonium ions by biochar would not prevent plant acquisition, but would greatly mitigate leaching loss. Another mechanism whereby biochar could prevent leaching loss is by increasing soil water retention, due to its small pore size. Increased nutrient cycling has also been reported in a number of studies (Sohi et al., 2009).
- **Combination treatment:** Theoretically, the combination treatment ought to increase nitrogen concentration in the treated subplots by the greatest amount. The one-time influx of chemical fertilizer would be supplemented by the gradual accretion of the legume plants. Furthermore, as mentioned above, biochar could prevent some leaching, and promote nutrient cycling.

1.4 Research Question and Objectives

1.4.1 Major question

Can soil treatments that increase nitrogen concentration or availability contribute to meadow restoration on nitrogen-limited old-field meadows in an urban context in Southern Ontario?

1.4.2 Supporting objectives

- To quantify and compare the effects of nitrogen (N) addition, native legume planting and biochar application in terms of indicators of plant establishment.
- To compare the combined effects of N addition, native legume planting and biochar application against each treatment individually.
- To assess soil physical, chemical and biological factors prior to treatment application as well as after one growing season.
- To produce results that will inform restoration methodology both on site and in similar ecosystems.

1.5 Hypotheses

- H1: If the amount of biochar added is sufficient to override local variability, then biochar and/or combination plots will have significantly higher organic matter than controls.
- H2: If nitrogen provided by chemical fertilizer stimulates plant growth, then above-ground biomass in fertilizer and combination subplots will be significantly higher than in controls.
- H3: If nitrogen fixed by native legume plants is sufficient to stimulate plant growth, then above-ground biomass in legume and combination subplots will be significantly higher than in controls.
- H4: If biochar increases plant-available nitrogen, then above-ground biomass in biochar and combination subplots will be significantly higher than in controls.
- H5: If the treatments are more effective in concert than individually, then combination plots will have significantly higher above-ground biomass than subplots treated with any individual treatment.
- H4: If increased plant productivity leads to increased inter-species competition, then subplots with higher biomass levels will be significantly negatively correlated with species richness.
- H5: If biochar-treated soil retains a greater amount of moisture than untreated soil, then biochar and combination plots will have significantly higher average soil moisture than controls.

1.6 Conceptual Framework

The field of restoration has evolved into a strong academic field over the past two decades, attracting ever-increasing levels of basic research and peer-reviewed publication, thanks to a long legacy of insight from a variety of fields, including erosion control, reforestation and habitat and range improvement (Young, Petersen & Clary, 2005). Restoration is inherently an interdisciplinary field. Undoubtedly, a great contribution to its knowledge base has been the integration of established ecological concepts. It is viewed by some academics as an "acid test" of our ecological understanding, a sounding board on which to assess our grasp of the structure and function of ecosystems (Bradshaw, 1987; Bradshaw, 1996). Beyond classic ecological science, restoration theory can, and indeed must, be supplemented by any number of other scientific disciplines (such as botany, wildlife biology and soil science), depending on the context of the work being done. It has also been argued that the field of restoration must also draw heavily from the humanities, politics, philosophy and traditional ecological knowledge in order to be capable of leaving a durable and positive legacy (Light & Higgs, 1995).

Inasmuch as it constitutes "typical" modern scientific investigation (concerned with hypothesis testing, analysis, etc.), my project falls within the boundaries of restoration ecology, as opposed to the broader field of ecological restoration (*sensu* Higgs, 1994). Given that I am largely concerned with the response of plant communities to soil conditions, it follows that I should borrow extensively from the theory and methodology of soil science (see Thompson and Troeh, 1973; White, 2006). As well, urban ecology provides an important contextual background to my work. I therefore propose that my research lies at the intersection of the fields of restoration ecology and soil science, and within the theoretical framework of urban ecology.

There are of course some important assumptions inherent to this research. The most important ones are:

- That the restoration of meadows is possible, at least to a certain extent (see Young, 2000).
- That addressing the soil conditions is an important first step in restoration (informed by Bradshaw, 1997).
- That the environmental effects of urban regions extend into relatively large urban natural areas.
- That studying a small subset of meadow species over a short period of time will produce relevant and useful data.

A variation of the first assumption is applicable to many if not most restoration experiments, and has been discussed in depth elsewhere (see Hilderbrand, Watts & Randle, 2005). It underlies much of the field of restoration ecology.

Chapter 2
Review of the Literature

2.1 Restoration ecology: an overview

In the early days of restoration ecology as an academic discipline, its major underlying principle was to return an ecosystem to some original state (Bradshaw, 1996). Then as now, this fixation on a historical (and seemingly static) ideal was contentious, and there has been a movement towards recalibrating the criteria of successful restoration towards more flexible and context-dependent ones (Choi, 2007; Hobbs & Harris, 2001). In any case, there is agreement that restoration consists of the transformation of a degraded area towards a relatively improved state (Hobbs & Norton, 1996). Clearly, any active restoration attempt will involve some element of human intervention and judgment. As such, some scientists have accused restorationists of interfering with some unadulterated form of nature (see Higgs, 2005).

Restoration has long been informed by succession theory, and has also significantly contributed to the development of that branch of ecology (Young et al., 2005; see section 2.2.4). As our understanding of ecosystem dynamics has evolved from a linear view to one more modulated by the fluxes inherent to complex systems (see Francis, 2005, for review), restoration scientists have taken note. Suding (Suding, Gross, & Houseman, 2004; Suding & Hobbs, 2009) has provided elegant reviews of the relevance of complex systems theory to our discipline.

Restoration ecology was, until recently, dominated by a “bottom-up” understanding of ecological processes, that is to say, the belief that ecosystems are driven by succession in plant communities. Concomitantly, much of the research in the field of restoration was concerned with the development of a vegetative community on contaminated or “derelict” plots (Soulé & Terborgh, 1999). Indeed, my own research fits neatly within this paradigm. Meanwhile, the field of conservation shifted its interest from spatially delimited protected areas to broad landscape-level processes such as migration and gene flow, as scientists came to the realization that some species or populations would inevitably go extinct if they remained isolated. As the utility of links between core habitats became recognized, the need to restore corridors on relatively large scales became obvious (Beier & Noss, 1998; Noss, 1987), especially in the face of range shifts due to climate change (Gilbert-Norton et al., 2010). As the size of restoration projects increased dramatically, the ability for wildlife species to act as agents of restoration on a regional scale began to be investigated and demonstrated (Griffiths et al., 2011; Ripple et al, 2010; Truett et al., 2001). This new “top-down” restoration paradigm has come to the fore in many large-scale restoration projects, such as rewilding endeavours and large mammal re-introductions. Restoration science will continue to benefit from an ever-broadening range of experimentation and theoretical input.

There is an important history of contributions to restoration science from a variety of other fields, from closely related conservation biology and population ecology to the more far-flung social sciences and indigenous studies (Diemont & Martin, 2009; Higgs, 2005; Newman, 2008; Shackelford et al., 2013; Young, 2000). There has been some debate regarding the relative importance of scientific rigour versus broad interdisciplinarity to the field of restoration (Cabin, 2007; Giardina et al., 2007). This underlies the supposed division between an experimental field of restoration ecology and the more holistic, applied ecological restoration. My work, being of experimental nature, certainly adopts the view that methodological exactitude is crucial to the advancement of knowledge. That is not to say, however, that experimental restoration research cannot be informed and complemented both theoretically and practically by other disciplines.

2.2 Soil science and restoration

Much of the early work that contributed directly to the field of restoration was conducted on the reclamation of “derelict” land such as mining spoils. Dr. Anthony Bradshaw recognized the basic connection between soil quality and the success of plantings on disturbed land, and he went on to become one of the founders of the nascent field of restoration ecology. He collaborated with Dancer et al. (1977a, 1977b, 1979) in conducting a series of experiments on the accumulation of nitrogen in a colliery in Cornwall, Southwest England, through natural processes, the contribution of forage legumes, or the addition of soil amendments (chemical and organic fertilizers). The results of this series of experiments, as well as a subsequent publication (Jefferies, Willson & Bradshaw, 1981), indicated that the establishment of a community of leguminous plants might be the most efficient (in terms of time, cost and effort) methodology to accumulate plant-available nitrogen in nitrogen-deficient sites. This work demonstrates the important link between soil conditions and the success of revegetation, as well as the possibility of human intervention in the process.

Bradshaw (1997) underlined the necessity of restoration scientists and practitioners having an understanding of soil processes. If the soil at a damaged site is relatively undisturbed, a site can generally recover relatively easily, especially if the seed bank is intact. However if the soil is heavily altered, then without intervention a site could take years to establish primary colonists and several decades for any soil processes to recover. Hanselwandter (1997) took this research further by describing the relevance of soil microorganisms (and mycorrhizal fungi in particular) to restoration ecology.

In the intervening years, much of the research on the role of soil processes in restoration has focused on physical and chemical, rather than biological, aspects of the soil, the major focus being on nutrient level recovery or reduction (Blumenthal, Jordan & Russelle, 2003; Cramer, Hobbs & Standish, 2008), pre-restoration site assessment and the evaluation of specific soil amendments (Callaham, Rhoades & Heneghan, 2008). Heneghan et al. (2008) emphasize that it is important to take a holistic view of soil processes (what they term “soil ecological knowledge”), in order to adequately inform restoration practice. It has been noted, for example, that exotic plants alter the structure and function of microbial communities in the soil (Kourtev, Ehrenfeld & Häggblom, 2002). In recent years there has been a push to incorporate the study of microbial communities in restoration ecology, both as indicators of restoration success (Izquierdo et al, 2005; Kardol, Bezemer & Van Der Putten, 2009) and as active agents in ecosystem recovery (Harris, 2009).

Biochar is a soil amendment that may have a role in promoting microbial activity and assisting in ecosystem recovery. It is a fine powder of pyrolyzed (heated in the absence of oxygen) organic material, which has been used successfully as an agricultural soil additive for decades or centuries in some regions, and is garnering increasing international attention for its potential as a agent of carbon sequestration (Lehmann & Joseph, 2009). The input materials and the process of producing biochar can vary widely, as can its effects on soil characteristics. In terms of physical and chemical properties, biochar has frequently been observed to increase soil pH, increase cation exchange capacity, increase water-holding capacity, and decrease soil bulk density (Warnock et al., 2007). A recent study successfully employed activated carbon, which is comparable to biochar (Lehmann & Joseph, 2009), as a restoration tool in old-fields and suggested that its ability to sequester allelochemicals released by invasive plants might be an important mechanism in promoting native plant growth (Kulmatiski, 2011). There has been consistent evidence demonstrating that biochar affects soil biotic communities. It has been shown to increase bacterial activity and populations, as well as root colonization by symbiotic mycorrhizae, by providing micro-refugia from predation (Lehmann et al., 2011).

2.3 Ecological restoration in cities and the problem of urban soils

Conservation efforts have long focused solely on large swaths of land in rural areas (Harris, 2010; Ingram, 2008; Miller & Hobbs, 2002). There are a number of compelling ecological, economic, educational and recreational arguments to greatly increase our efforts in urban centres, and to combine the preservation of natural land with ecosystem restoration. There are also great challenges to working in cities, notably highly disturbed soil properties and processes (de Kimpe and Morel, 2000), which, if not explicitly addressed, can lead to difficulties for attempted restoration projects (Pavao-Zuckerman, 2008).

Small-scale “green infrastructure” (GI) projects can profit urban areas, for example by helping to attenuate the heat-island effect, or by providing opportunities for citizen engagement in nature (Forman, 2008; Standish et al., 2012). Larger-scale GI projects are becoming increasingly common in and around cities worldwide, as the ability for natural areas to provide necessary ecosystem services, such as aquifer recharge or wastewater purification, is recognized by planning agencies. In many cases, the initial costs of GI projects are smaller than those of traditional infrastructure projects, maintenance costs are low, and tangential benefits (eg. recreation, wildlife habitat) are important (Foster, Lowe & Winkelman, 2011).

The vulnerability of some urban areas to natural calamities has been all-too-clearly exposed in recent years, as has the role of human-mediated habitat destruction in exacerbating the process (Day et al., 2007). In response, some truly pharaonic ecosystem restoration programs are being implemented. Louisiana’s $50-billion, 1500-project, master plan hopes to reverse the state’s loss of coastal wetlands and barrier islands (CPRAL, 2012), while up to $10-billion will be invested into restoring hundreds of coastal features in New York’s Hudson-Raritan estuary (USACE, 2009). Other massive restoration projects have been initiated in Florida’s Everglade ecosystem and San Francisco Bay. Ecological restoration is increasingly seen as a solution towards boosting resilience in the face of increasingly rapid global environmental change (Suding, 2011).

Conducting wetland restorations in urban areas can be marred by layers of complexity, due to the properties of urban ecosystems, which have been altered by anthropogenic influence (Ingram, 2008). Many of these same facets of the urban environment cause parallel issues in terrestrial ecosystems, such as, for example: pollution, altered drainage patterns, the heat island effect and invasive exotic species. The soil is an aspect of urban environments that, while also affecting aquatic systems, disproportionally affects terrestrial ones.

There are few publications describing the distinct properties of urban soils as compared to rural soils. This is partly due to the great variability of soils in general and urban soils in particular, which makes generalization quite difficult; the remainder of the equation comes down to a gap in the academic research. Pouyat et al. (1995) described variation in soil properties along a rural-urban gradient along a 130km transect originating in New York City. They noted trends towards increasing heavy metal and salt concentrations, higher temperature, higher levels of organic matter and total nitrogen as well as elevated earthworm populations (especially invasive exotic species) and decreasing pH, fungal populations (*idem* mycophagous invertebrates) in urban areas. It stands to reason that these trends will vary depending on the land use patterns and the geophysical characteristics under scrutiny (for example see Biasioli, Barberis & Ajmone-Marsan, 2006). There are, however, some general patterns that emerge in soils across urban areas (Craul, 1992):

1. Great vertical and spatial variability
2. Modified soil structure leading to compaction
3. Presence of a (usually hydrophobic) surface crust on bare soil
4. Modified pH, usually elevated
5. Restricted aeration and water drainage
6. Interrupted nutrient cycling and modified soil organism population and activity
7. Presence of anthropeic materials and other contaminants
8. Highly modified soil temperature regimes.

A site's history, soil properties, seed bank, mycorrhizae and other remnants (collectively termed "soil ecological memory") all have lasting effects on an eventual replacement community or ecosystem which may take hold following disturbance (Schaefer, 2009). This is especially poignant in cities, where repeated disturbance and resilient invasive species can mask the history of a site. Without active restoration, it is unlikely that many urban sites will return to their former natural states: novel ecosystems are likely to be formed (Schaefer, 2009).

For a variety of reasons (such as their location near rivers and estuaries), human settlement is disproportionately located in regions of globally important biodiversity (Miller & Hobbs, 2002; Rudd et al., 2002). Some cities (eg. Perth, Cape Town), built in regions identified as global biodiversity hotspots, may be carrying a particularly high extinction debt. Intervention may play an important role in mitigating the future loss of species in those areas (Standish et al., 2012). Yet the academic literature is now rife with heady debate as to whether or not restoration is appropriate in many areas, especially in the light of possible sustained climate change (Harris et al. 2006; Jackson & Hobbs, 2009). Given limited resources, is it worthwhile to invest in re-creating historic species assemblages if there is no guarantee of success? Is it best to adopt a new paradigm centred on promoting ecosystem function and resilience (Shackelford et al., 2013)? In any case, it is likely that restoration work, of some from or other, will continue to increase in cities, for reasons driven both by sentimentality and science (Harris, 2010). The meadow, for instance, is one threatened ecosystem that we are not ready to dispense with just yet (Kardol, Bezemer & Van Der Putten, 2009).

2.4 Old fields

The history of abandoned cultivated land is as long as that of agriculture itself. While the effects of cultivation might essentially disappear in a matter of decades in some areas, in others they can alter the ecology of the land indefinitely (Dale & Carter, 1955). For a number of ecological, economic, cultural and demographic reasons, the trend of farmland abandonment has been accelerating rapidly in recent years (see Figure 1). Whether this trend is troubling or promising depends on one's

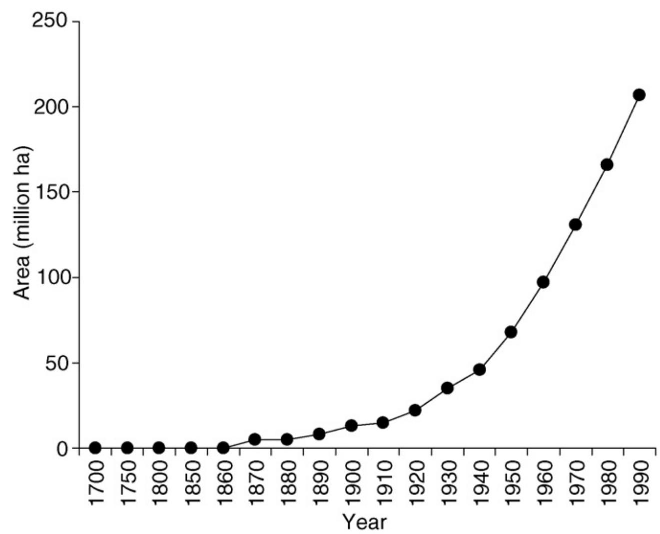


Figure 1: Estimated area of abandoned cropland, worldwide, between 1700 and 1990. With permission from Hobbs and Cramer (2007). They note that there are very few sources compiling this data on a global scale.

perspective, but there is no doubt that it has greatly contributed to our understanding of ecological processes. Succession ecologists have greatly informed their theory through long-term observations of old-fields (Young, Chase & Huddleston, 2001). Abandoned farmland also offers an otherwise-unavailable canvas for the testing of basic ecological theory (Huberty, Gross & Miller, 1998). Rejmánek and Van Katwyk (2005) called old-fields the '*Drosophila* of terrestrial ecology' (see http://botanika.bf.jcu.cz/suspa/pdf/BiblioOF.pdf for a comprehensive bibliography of old-field literature). Insights from those disciplines as well as more recent experimental investigation have greatly contributed to the theoretical foundations of restoration ecology. It is likely that restoration scientists and practitioners will now play a major role in the management of abandoned farmland worldwide (cf. Navarro & Pereira, 2012; Proença, Honrado & Pereira, 2012).

There is debate in the literature as to whether predictable (succession theory) or stochastic (community assembly theory) forces dominate the process of ecosystem formation or recovery. The former situation would lead to a stable 'climax' state, while the latter, to one of a number of alternative climax states. Young, Chase and Huddleston (2001) note that both theories are relevant to restoration ecology in that each one describes patterns observed in natural ecosystems, and suggest that ecosystem type and geophysical context might be the predominant factors in determining which model is most relevant to a particular situation. Indeed, in a synthesis of the old-field literature, Cramer, Hobbs and Standish (2008) were able to describe some broad trends that illustrate both theories. They propose that if no biotic or abiotic thresholds are crossed, a field is likely to return to its former vegetation cover with minimal intervention. However, if biotic (eg. depleted seed bank) and/or abiotic (eg. nutrient excess or deficiency) thresholds are crossed, it is likely that a field will attain an alternate stable state or develop a novel ecosystem form, if no active restoration management is implemented. They emphasize the relevance of context, such as the proximity of sources of seed dispersal, topography and climate, in the process of ecosystem recovery (Cramer, Hobbs & Standish, 2008).

Abandoned farmland in Eastern North America, and other temperate zones with deep soils worldwide, tends to revert to forest cover within a few decades. Indeed the vast majority of forest within this region is secondary growth on land that was once cleared for cultivation (Hobbs & Cramer, 2007). The general pattern described by ecologists observing old-fields in Eastern North America is the rapid establishment of annual forbs, including legumes, and grasses, followed in quick succession, generally two to three years, by biennials. Perennial shrubs displace these species within five to ten years, which in turn are outcompeted by trees in a matter of decades (Meiners, Pickett & Cadenasso, 2002). It has been suggested that the predictability of this sequence might be a reflection of the adaptation of the flora to repeated disturbance, both in the form of recent glaciation and Native American burning practices (Cramer, Hobbs & Standish, 2008). While this sequence is broadly applicable, some authors have pointed out that there is considerable nuance both within and between sites. Maycock and Guzikowa (1984) described an old-field that was still dominated by early succession species more than fifty years after abandonment. They and others (Emery & Gross, 2007; Huberty, Gross & Miller, 1998) emphasize that site history, among other local factors, will affect the rate and identity of establishing species.

Studies on old-field vegetation in Southern Ontario have been surprisingly scant. In fact, there is only one published survey of this plant community, on a plot of abandoned farmland in Erindale, ON, (Maycock & Guzikowa, 1984) and there have been no long-term experiments or observations in the province to date. Maycock and Guzikowa found a total of 118 species, of which 61 had measurable cover; *Elymus repens*, *Vicia cracca*, and *Poa pratetnsis* were major dominants, followed by *Solidago altissima*. Although native species (52%) outnumbered exotic ones (48%), the made up only between 20-33% of the ground cover. Murphy (2010, unpublished data) conducted a vegetation and faunal survey at Huron Natural Area in Kitchener, ON (one of the field sites used in this study; see section 3.1.1) in May 2010. In Murphy's survey, of 59 plant species identified, only 21 were native. The reason for the higher proportion of invasive species in the latter study could be the shorter time frame since abandonment (~20 vs. 50 years). Meiners, Pickett and Cadenasso (2002), as part of the Buell-Small Succession Study in New Jersey, observed that exotic species cover decreased significantly in successional fields over 20 years of age. The tendency for old-fields in many biomes to be dominated by exotic, often invasive plant species is well documented and is often cited as rationale for active ecological restoration (Tognetti et al., 2010).

2.5 Meadow ecology

A meadow is commonly defined as an open ecosystem dominated by non-woody vegetation such as grasses, forbs and sedges. In temperate, fertile zones worldwide, a meadow can be understood as representing the earliest successional stages in forest dynamics – annual, biennial and, eventually, perennial non-woody plants colonizing an opening in the surrounding forest matrix. Natural agents of disturbance (eg. lightning-caused fires, treefall due to windstorms or disease, flooding, ecosystem engineering wildlife species, etc.) can create a forest opening, in which a meadow will develop. These are the classic disturbances discussed in the gap dynamics literature (Pickett and White, 1985). Alternatively, such disturbances can be wrought by anthropogenic means (eg. logging, clearing for cultivation, etc.). Natural factors such as climate, edaphic conditions or intense herbivory might temporarily or indefinitely maintain a meadow at an early successional stage. This, again, can also be accomplished artificially, through livestock grazing, mowing, and burning. In Europe, there is a long history of anthropogenic maintenance of meadows, especially for pasture or hay production. Those systems are valued for their floral and faunal biodiversity, but modern mechanized farming practices have greatly reduced their extent. Restoration efforts on that continent aim to renew or mimic those historic models of stewardship, while rewilding projects reintroduce extirpated or extinct-surrogate large mammal species (Navarro & Pereira, 2012; Proença et al., 2012).

The picture in Eastern North America, and Southern Ontario specifically, is somewhat more complex. Some fields are currently maintained for pasture or hay production (Milne & Bennett, 2007). There is no doubt, however, that landscape openness has dramatically decreased in this region since European settlement. Vast areas of prairie, savanna, and meadow have succeeded into closed-canopy forest (sometimes preceded by a period of use as cropland). This pattern has been attributed to the demise of the traditional use of fire by native peoples (Nowacki & Abrams, 2008). The extent to which fire was used to alter landscape-level ecological process in the province, however, has been debated. Russell (1983) suggested that native use of fire was highly localized or accidental, and did not affect vegetation patterns on a landscape scale. Szeicz and MacDonald (1991) proposed that the development of oak savanna in Southern Ontario, between 8000 and 4000 BP, may have been due to climate change rather than anthropogenic activities. Their argument against anthropogenic involvement, however, was centered upon the fact that the cultures known to inhabit the area at the time were categorized as "hunter-gatherer", and thus not inclined to altering their environment at a large scale. There has been increasing evidence in other regions demonstrating that "hunter-gatherer" tribes in fact conducted regular and long-term manipulation that caused cumulative and lasting effects in plant associations, species composition and genetic structures, through pruning, sowing, weeding, tilling, selective harvesting, and most significantly, burning (Anderson & Moratto, 1996). There is now substantial evidence in the literature that anthropogenic use of fire and other forest-clearing methods had long-term impacts on the Southern Ontarian landscape (Clark & Royall, 1995; Dey & Guyette, 2000; Munoz & Gajewski, 2010) and in comparable environments (Dorney & Dorney, 1989), though the scale of the influence is still unclear. Burning woodlands improved the quality of browse for wild game, increased the ease of hunting (Mann, 2005), cleared fields for agriculture and increased the yields of foraged foods (Dey & Guyette, 2000).

The historic importance of anthropogenic disturbance in providing habitat for early-succession species was certainly highly significant (Munoz & Gajewski, 2010). In some North American ecosystems, fire-adapted species have already been lost (Nowacki & Abrams, 2008). Anderson (1996) proposes that if land managers aim to preserve or restore ecosystems in a state resembling their pre-contact structure and function, then they must recognize historical anthropogenic effects and investigate the possibility of simulating some of these cultural practices.

Clark and Royall (1995) also acknowledge that the reintroduction of anthropogenic means of stewardship might be important for the restoration of certain plant communities in Southern Ontario, and indeed this has been implemented on some properties (Thompson, personal communication, 2012). It should be acknowledged that these practices might interfere with other restoration priorities, such as the elimination of exotic or invasive species (Tognetti et al., 2010). Some exotic species thrive after periodic burning, and given that anthropogenic (native and European) burning continued for some time after the introduction of some invasive species (Dey & Guyette, 2000), that cultural practice may have contributed to the expansion of their range. Integrating traditional ecological knowledge and customs with modern scientific experimental methods will advance our ability to restore healthy and diverse ecosystems to the landscapes of Southern Ontario.

2.6 Restoring urban meadows

The vast majority of the meadow restoration literature originates from Western Europe, and this stems from a long history of human-landscape interactions in that part of the continent (see section 2.2.5) and concerted efforts by the European Community to promote biodiversity in the agricultural landscape (Berendse et al 1992; Smith et al., 2003). There, as in Eastern North America and other temperate zones, forb-dominated meadows are an early successional community within a patchwork of temperate forest and are thus, by their very nature, ephemeral without concerted anthropogenic management. Nevertheless, maintained meadows, sometimes termed cultural meadows, can be highly diverse and act as habitat for a number of threatened and endangered species. In southern Ontario, cultural meadows are notable for their importance to bird and insect populations, but many are transitioning either to development or reverting to forest cover (Milne & Bennett, 2007).

The traditional conservation paradigm tended to view urban spaces, at best, as lost land whose redeeming quality was to keep vast numbers of people out of unblemished natural areas. Increasingly, however, the value of urban natural habitat is being recognized by researchers (Rudd et al., 2002; Standish et al., 2012). A number of taxa, including rare and endangered species, rely on remnants of native ecosystems, backyard habitat, and even novel ecosystems. Increasing the extent, quality, and connectivity of urban green space will be crucial in sustaining these species in the long term (Standish et al., 2012). The work of restoring meadows in cities has, to an extent, already begun. NGOs and governments have programs in place promoting various forms of gardening whose aim it is to promote biodiversity (native species gardening, wildlife-friendly gardening, pollinator gardens, etc.; Goddard, Dougill & Benton, 2010). These private and corporate gardens can be considered as meadow habitat to the extent that they are forb-dominated, early successional systems. There has also been a movement towards landscape naturalization in urban parks, including, in many cases the restoration of meadow habitat (Handel, Saito & Takeuchi, 2013). Restoring meadows in urban areas may hold some appeal to public land managers: they require little maintenance and provide aesthetic, ecological, recreational and educational benefits. Increasing their extent through de-paving, old-field restoration, and the conversion of lawn and other land of low ecological value will help reverse local and global trends of grasslands loss. Research into the ecological value of urban green spaces has been scant, due their historic perception as depauperate systems, and difficulty of access owing to fragmented ownership. Yet as awareness of these areas' conservation potential increases, academics, planners and policymakers are increasingly collaborating to maximize their value (Goddard, Dougill & Benton, 2010; Rudd, Vala & Schaefer, 2002)

There is increasing recognition in the literature of the importance of addressing soil factors in the process of meadow restoration. Topsoil translocation has been assessed for its potential to conserve entire plant communities (Vécrin & Muller, 2003) and turf transplantation has demonstrated similar promise (Pywell, Webb & Putwain, 1995). Smith et al. (2003) tested a variety of management techniques and soil additives and concluded that seed sowing was correlated with increased plant diversity while mineral fertilizer or farmyard manure application was not. Changes in the soil microbial communities linked with the growth of legume species may be important for long-term increases in plant diversity (Smith et al., 2008). Compost ameliorant may assist in the early germination of desired plant species, but in the long term it provides a substrate prone to invasion by competitive species (Carrington & Diaz, 2011). As discussed above, activated carbon or biochar may constitute a superior soil additive in old-field habitats because it does not directly provide available nutrients, but it can enhance bacterial populations and mycorrhizal root colonization, as well as sequestering allelochemicals (Kulmatiski, 2011).

Chapter 3
Methodological Approach

3.1 Study Sites

3.1.1 Huron Natural Area

Huron Natural Area (HNA) in Kitchener, Ontario is a 107-hectare site (see Figure 2), comprising a variety of ecosystems, including wetlands, forests, ponds and a coldwater stream. It is run as a partnership between the City of Kitchener and the Waterloo Catholic and Waterloo Region District School Boards (City of Kitchener, 2010). The meadow at HNA (43°23'N, 80°28'W; elevation 342m; Google Earth, 2011) is an ex-agricultural field, which was tilled until the late 1970s. Some of the northern reach of the meadow was quarried starting in the late 1980s, during the construction of Trillium Rd. and industrial development on adjacent lots. Meanwhile the southern extent saw the establishment of a system of trails, which became grown over following a 1990 municipal decision to protect the site. Huron Natural Area formally opened to the public in 2006, and continues to run an active stewardship and ecological restoration program (City of Kitchener, 2010). The soil of the region is a sandy-loamy luvisol (Chapman and Putnam, 1984). The area receives on average 940 mm/yr of precipitation, 83% as rainfall and 17% as snowfall (Environment Canada, 2011).

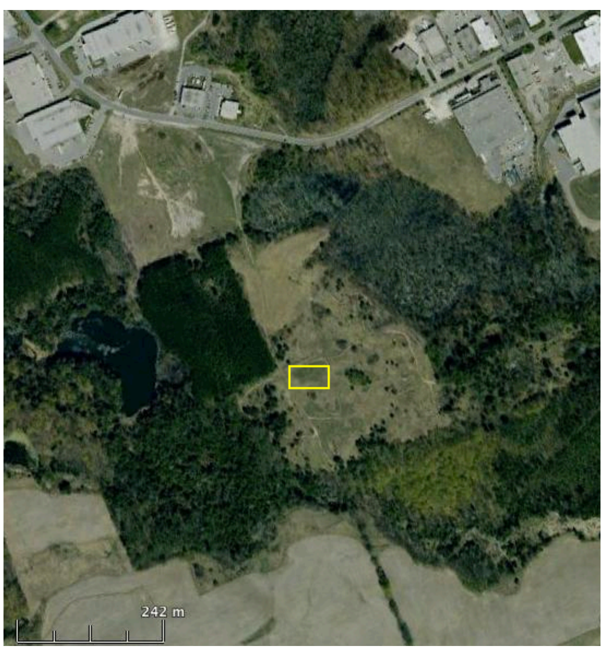


Figure 2: Huron Natural Area. The meadow ecosystem features prominently within the natural area (Google Earth, 2013). The rectangle denotes the approximate location of the worksite.

The specific site for the study plot was chosen in the southern stratum of the meadow, which is the area the City has prioritized for restoration. Within that section, the site choice was constrained by the pre-existing restoration plots and had to exclude steeper sections (logistical consideration for tractor access). A site was agreed upon with City personnel and within it, the specific plot location was chosen by converting a map of the site into an x,y grid and using a random number generator to pick coordinates.

3.1.2 *rare* Charitable Reserve

rare Charitable Research Reserve was founded in 2001 and is a 402-hectare property in Cambridge, Ontario (see Figure 3). It is a privately run charitable foundation with a strong focus on conservation, monitoring, restoration and community education (*rare*, 2011). The study site at *rare* is an old-field in the Springbank Farm section of the reserve (43°22'N, 80°21'W; elevation 306m; Google Earth, 2011), which was last cultivated for canola and other crops in 2006-7 (Kelly, personal communication). The site currently houses a greenhouse and apiary, community garden plots, native plant gardens, educational facilities and art installations. The soil is a clay-loam and the precipitation patterns are comparable to those at HNA.

Figure 3: *rare* Charitable Research Reserve, sitting at the confluence of the Grand and Speed rivers, comprises natural ecosystems as well as farmland (Google Earth, 2013). The rectangle denotes the approximate location of the worksite.

The specific site for the research plot was again chosen taking logistical constraints into account (distance from community garden plots and tractor accessibility). Within the site agreed upon with *rare* staff, the specific plot coordinates were chosen using the same methodology as at HNA.

3.1.3 Defining plots

At each site, the periphery of the experimental area (13x16m) was marked using wooden stakes. The sites were tilled using a tractor (October 2011) and then individual treatment plots were delineated using white nylon string pegged with 8” nails. Each research plot is 5 subplots (2x2m) long by 4 subplots wide, for a total of 20 subplots, with 1m-wide buffers around the edges separating each subplot from its neighbours (see Figure 4).

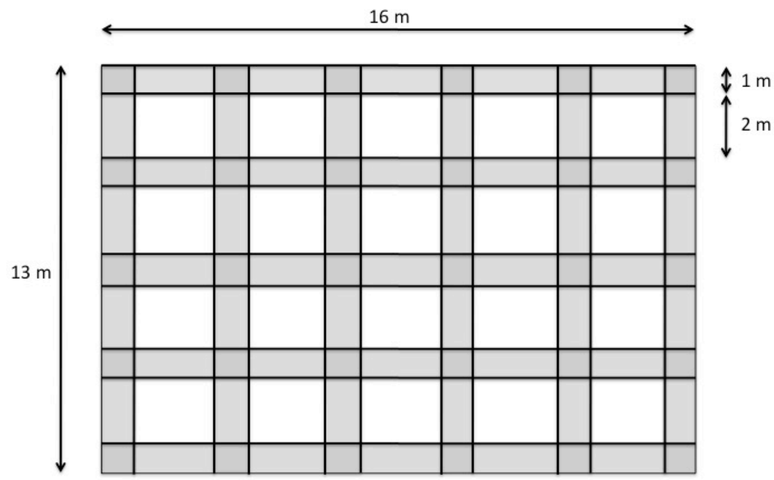


Figure 4: Depiction of the research plot layout, white areas are subplots and grey areas are buffer zones.

3.2 Soil testing

Using a 3cm diameter soil auger, 3 soil samples were collected from each subplot (using random coordinates and excluding 50 cm from the edges) to a depth of 15cm, on October 23, 2011, as well as September 13, 2012. All 3 samples from individual sub-plots were bulked in a single, re-sealable plastic bag, immediately put on ice in the dark and then frozen in the laboratory for no more than 2 months (as in Kulmatiski, 2011). Soil samples were tested, throughout December 2011 and September/October 2012, for:

- pH
- moisture content and organic matter; loss on ignition method
- nitrate; cadmium reduction method
- phosphate; ascorbic acid reduction method
- potassium; tetraphenylboron method (LaMotte, 2012)

3.3 Application of treatments

Treatments were applied following soil testing on October 30, 2011. They consist of (1) chemical fertilizer, (2) legume plants, (3) biochar, (4) a combination of 1, 2 and 3 and (5) a non-treatment control. Each subplot (4 subplots per treatment per plot: total of 40 subplots) was randomly assigned one of the four treatments, or a control status.

1. Fertilizer treatment subplots each received 85g of Scotts® Turf Builder® 32-0-4 PRO Lawn Food fertilizer (equivalent to 67.5 kg N/ha).
2. Legume treatment subplots were seeded with a mixture of *Astragalus canadensis* L. and *Lupinus perennis* L. 5g of each species were hand spread in each plot. *L. perennis* was scarified using sand paper. Seeds were obtained from the Ontario Seed Company, Waterloo, ON, and treated with genus-appropriate rhizobia.
3. Biochar treatment subplots received 4 kg (1 kg/m^2, equivalent to 10 tonnes/ha) of biochar (Abritech, Namur, QC; ash wood parent material; fast pyrolysis at 450°). 2 kg of biochar were applied in combination-treatment plots instead of 4 (due to material constraints). It was raked into the top 10cm of the soil. All other plots were similarly raked but without biochar addition.
4. Treatments 1, 2 and 3 were applied to these combination subplots.

Control subplots did not receive any soil amendments, but they were raked similarly to other subplots and then seeded with meadow plants.

3.4 Seeding of meadow plants

A mixture composed of 25% *Panicum virgatum* L., 25% *Schyzachirium scoparium* (Michx.) Nash, 25% *Carex brevior* (Dewey) Mack. / *C. muhlenbergii* Schkuhr ex. Willd., 12.5% *Rudbeckia hirta* L., 12.5% *Desmodium canadense* (L.) DC., obtained from Native Plant Source, Kitchener, ON, was hand-broadcast in each subplot. An effort was made to broadcast as evenly as possible. A total of 1.04 kg was seeded, which corresponds to a seeding rate of 25 kg/ha.

3.5 Assessment of meadow plant establishment

A 50x50 cm quadrat was dropped on a randomly selected area of each 2x2m subplot. Each quadrat was assessed for species present and number of individuals of each species (Shannon's Diversity Index; Rosenzweig, 1995). Vegetation was assessed in May, June, and August, 2012.
All above-ground (live and dead) plant material was clipped and harvested on September 13, 2012, and then dried and weighed following the USDA-NRCS Above-Ground Biomass Determination protocol (USDA-NRCS, 1997).

3.6 Monitoring of plot temperature and moisture

Soil temperature was measured within each subplot using a Digi-Sense ® Thermistor 400 Series (Cole-Parmer Co., Montreal, QC) electronic thermometer connected to a probe. A point within the plot was chosen randomly, the probe was inserted to a depth of 10 cm and the temperature reading was allowed to stabilize. Moisture was measured using a Kelway Soil Tester (Kelway Instruments Co., Wyckoff , NJ). The soil was loosened using a trowel, the meter was fully inserted into the soil and then the soil was made firm around it. Moisture readings were generally taken from the same location within the plot to minimize disturbance to the vegetation, as plant material needed to be removed in order to take the measurement. See Table 1 for the dates on which temperature and moisture sampling took place. It was conducted, whenever possible, on a weekly basis.

Table 1: Dates of temperature and moisture sampling at Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON.

Date	Temperature	Moisture	Huron NA	Springbank
May 21, 2012	x	x	x	x
May 27, 2012	x	x	x	x
June 5, 2012	x	x	x	x
June 10, 2012	x	x	x	x
June 16, 2012	x	x		x*
June 24, 2012	x	x	x	x
June 28, 2012	x	x	x	x
July 13, 2012	x	x	x	x
July 19, 2012	x	x	x	x
August 18, 2012	x	x	x	x
August 22, 2012	x	x	x	
August 23, 2012	x	x		x
September 13, 2012	**	x	x	

* Some data points were taken on June 19, 2012 for this site.

** Temperature not measured due to technical issue with thermometer.

3.7 Data analysis

Data was tested for normality using the Shapiro-Wilk regression test. Differences in means for soil variables and vegetation diversity and biomass were tested using univariate or repeated measures analysis of variance (ANOVA), as appropriate. Tukey's *post hoc* pairwise comparisons were used to determine mean differences (as in Kulmatiski, 2011). In some instances, Pearson's product-moment correlations were employed for comparative purposes (Blythe & Merhaut, 2007). A significance level of 0.1 was used in this experiment instead of the customary p-value of 0.05 in order to increase the chance of observing relationships that might otherwise be missed. All statistical tests, unless otherwise noted, were performed using R (version 2.15.1, The R Foundation for Statistical Computing). Graphs, unless otherwise noted, were produced using ggplot2, a plotting system for R. All comparisons of data between field sites were performed using unpaired Student's *t*-tests.

3.7.1 Soil nutrients, pH and organic matter

A repeated-measures ANOVA, with treatment as independent variable and soil factor as dependent variable, was employed to discern differences in means for soil factors over time. A univariate ANOVA with Tukey's *post hoc* was run for each sampling event in order to discover any significant differences in the means of the measured soil factors.

3.7.2 Vegetation – Species Richness

A repeated measures ANOVA, with treatment as independent variable and species richness as dependent variable, was run for each site to discern differences in mean species richness between treatments, over time. The direction and strength of relationships between species richness and the various soil factors measured were determined through Pearson's product-moment correlations.

3.7.3 Vegetation – Biomass

A univariate ANOVA with Tukey's *post hoc* was run for the data from each site in order to determine whether any of the treatments had significantly different means. Pearson's product-moment correlations were computed in order to determine the strength and directions of the relationships between above-ground biomass and the various pre-and post-treatment soil factors. A correlation was calculated in order to assess the relationship between biomass and species richness.

3.7.4 Soil temperature and moisture

Repeated measures ANOVAs were used in order to detect differences in means between treatments, over time, and treatment-time interactions. Pearson's product-moment correlations were run in order to discern the strength and direction of the relationships between soil temperature, moisture, species richness and biomass.

Chapter 4
Analysis of the Experimental Results

4.1 Results

4.1.1 Soil nutrients, pH and organic matter

Most soil factors varied significantly between sites in pre-treatment plots (see Table 2). Concentrations of phosphorus (P; $p < 0.005$), potassium (K; $p < 0.001$) and organic matter (OM; $p < 0.001$) from October 2012 soil samples were all, on average, higher at Springbank Farm (Springbank) than at Huron Natural Area (HNA), while the pH ($p < 0.001$) was lower at that site. Only nitrogen (N; p = 0.16) did not vary between sites.

Concentrations of soil factors measured post-treatment, in September 2012, all differed significantly between sites. N ($p < 0.001$), P ($p < 0.01$), K (p < 0.001) and OM ($p < 0.001$) were all higher at Springbank than at HNA, while pH ($p < 0.001$) remained significantly lower.

Table 2: Average values (μ) and standard deviations (SD) for soil factors, at Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON. Significant differences obtained using paired two-tailed t-tests ($p < 0.1$).

Test	October 2011				September 2012			
	HNA		Springbank		HNA		Springbank	
	μ	SD	μ	SD	μ	SD	μ	SD
Nitrate (ppm)	0.14	0.01	0.15	0.06	0.18	0.04	0.26[°]	0.05
Phosphorus (ppm)	1.12	1.09	2.35[*]	1.54	1.07	1.02	1.77[°]	0.66
Potassium (ppm)	0.90	0.20	2.63[*]	1.85	1.07	0.38	2.59[°]	1.64
pH	8.08	0.13	7.81[*]	0.11	8.20	0.14	7.75[°]	0.08
Organic matter (% dry weight)	3.13	0.49	6.08[*]	0.75	3.62	0.75	6.95[°]	0.66

[*] Significant difference between sites, in 2011 (pre-treatment application).

[°] Significant difference between sites, in 2012.

Analysis of the soil factors demonstrates that many changed in concentration between the two field seasons. However, there were no significant treatment effects (effects of individual treatments on the magnitude of the change), or treatment-time interactions (effects of treatments on the direction of change over time). At HNA (see Table 3a), average concentration of N increased between 2011 and 2012 (0.14 to 0.19 ppm; $p < 0.001$), but there was no effect of treatment on the magnitude ($p = 0.87$) or direction ($p = 0.83$). K (0.90 to 1.07 ppm; $p < 0.1$), pH (8.08 to 8.20; $p < 0.05$) and OM (3.13 to 3.62%; $p < 0.01$) also increased over time, and similarly showed no significant treatment impacts, though the treatment-time interactions for pH and OM were more important (p of 0.16 and 0.20, respectively). P did not vary significantly between years.

At Springbank (see Table 3b), N (0.15 to 0.26 ppm; $p < 0.001$) and OM (6.08 to 6.95%; $p < 0.001$) increased significantly over time, while pH (7.81 to 7.75; $p < 0.05$) showed a decrease. Average concentration of P (2.35 to 1.77 ppm; $p = 0.15$) decreased substantially but not significantly, while K differed little between sampling events. Just as at HNA, there were no treatment effects on soil factors, either in terms of the magnitude or the direction of change. Only for pH, with a treatment-time interaction p of 0.12, was there a value nearing statistical significance.

There were no significant differences between treatments for any post-treatment (2012) soil factor at either site, with a single exception (see Table 4). A univariate ANOVA indicated a divergence ($p < 0.1$) between treatments in terms of percent organic matter at HNA, with Tukey's *post hoc* analysis pointing towards significant differences between control and biochar treatments ($p = 0.095$) and fertilizer and biochar ($p = 0.096$). It is of note that the same analysis, using the pre-treatment (2011) organic matter data, indicates no significant variation between treatments ($p = 0.64$), with Tukey's *post hoc* analysis showing no difference between control and biochar ($p = 0.99$) or fertilizer and biochar ($p = 0.90$).

Among the other treatments at HNA, the next nearest to statistical significance is P ($p = 0.23$), followed by, in decreasing order, K ($p = 0.52$), pH ($p = 0.77$) and N ($p = 0.89$). At Springbank, though none achieved statistical significance in terms of any tested soil factor, the treatment with the highest divergence between treatments was also OM ($p = 0.25$). It was followed again by P ($p = 0.33$), and then pH ($p = 0.34$), K ($p = 0.76$) and N ($p = 0.99$).

Table 3: Average values for Oct. 2011 and Sept. 2012, as well as change between field seasons, for soil factors at (a.) Huron Natural Area and (b.) Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON. Significant differences (p<0.1) obtained using repeated measures ANOVA (Tr: treatment; *t*: time; Tr : *t*: treatment-time interaction).

a.

Test	Average		Average change						Significance (p)		
	2011	2012	μ	C	B	F	L	X	Tr	t	Tr : t
Nitrate (ppm)	0.14	0.19	+0.05	+0.06	+0.03	+0.07	+0.04	+0.05	0.87	<0.001*	0.83
Phosphorus (ppm)	1.12	1.07	-0.05	+0.32	+0.33	-0.60	-0.30	-0.01	0.32	0.88	0.89
Potassium (ppm)	0.90	1.07	+0.17	+0.13	+0.32	+0.12	-0.10	+0.38	0.59	<0.10*	0.57
pH	8.08	8.20	+0.12	+0.01	+0.08	0.00	+0.32	+0.17	0.95	<0.05*	0.16
Organic matter (%)	3.13	3.62	+0.49	-0.14	+1.23	+0.30	+0.41	+0.67	0.58	<0.01*	0.20

b.

Test	Average		Average change						Significance (p)		
	2011	2012	μ	C	B	F	L	X	Tr	t	Tr : t
Nitrate (ppm)	0.15	0.26	+0.11	+0.13	+0.13	+0.11	+0.09	+0.11	0.51	<0.001*	0.82
Phosphorus (ppm)	2.35	1.77	-0.58	-0.54	-0.39	-0.12	-0.89	-0.24	0.29	0.15	0.95
Potassium (ppm)	2.63	2.59	-0.04	0.63	-0.20	-0.75	-0.18	+0.16	0.43	0.95	0.95
pH	7.81	7.75	-0.06	0.00	0.04	-0.11	-0.05	-0.13	0.99	<0.05*	0.12
Organic matter (%)	6.08	6.95	+0.87	+0.98	+1.01	+1.48	+0.04	+1.39	0.51	<0.001*	0.39

* Significant difference over time.

Table 4: Average values for soil factors in 2012 for different treatments, at Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON. μ: overall average; C: control; B: biochar; F: fertilizer; L: legume; X: combination. Significant differences (p<0.1) obtained using univariate ANOVA.

Test	Huron Natural Area							Springbank						
	μ	C	B	F	L	X	*p*	μ	C	B	F	L	X	*p*
Nitrate (ppm)	0.19	0.20	0.18	0.20	0.18	0.18	0.89	0.26	0.27	0.25	0.26	0.26	0.24	0.99
Phosphorus (ppm)	1.07	1.61	1.68	0.66	0.27	1.15	0.23	1.77	1.44	1.70	1.75	1.57	1.69	0.33
Potassium (ppm)	1.07	1.08	1.20	0.90	0.88	1.28	0.52	2.59	2.68	1.55	2.95	2.95	2.06	0.76
pH	8.20	8.14	8.20	8.18	8.28	8.21	0.77	7.75	7.79	7.77	7.76	7.76	7.70	0.34
Organic matter (%)	3.62	3.10	4.35	3.10	3.75	3.81	0.07*	6.95	6.71	6.98	7.38	6.45	7.05	0.25

* Significant difference between treatments.

4.1.2 Vegetation – Species Richness

The total recorded in-plot species richness at HNA was 22 species, while at Springbank it was 43 species. On average, species richness per subplot was greater at Springbank than at HNA (averages of 10.57 and 7.58 respectively; *t*-test p <0.0001). The highest recorded mean (per treatment) at HNA was of 8.50, in May 2012, while the 3 treatments were tied for a low of 7.00 (Control, May; Fertilizer and Legume, June). At Springbank, the highest mean was in the Fertilizer subplots (μ = 16.00) in June, while a low of 6.75 was recorded in the Biochar subplots, in May. At HNA, species richness per-subplot was higher in May, 2012 (μ = 7.80; see Table 5) than in June (μ = 7.35). Meanwhile, at Springbank, species richness was highest in June (μ = 14.05), surpassing the values for August (μ = 9.55) and May (μ = 8.10).

A repeated-measures ANOVA indicated no significant effect of treatment, time, or treatment-time interaction (see Table 6). At both field sites, significance values for time indicated a stronger effect than those for treatment (p = 0.35 at HNA and p = 0.22 at Springbank, versus p = 0.92 and p = 0.98, respectively), while treatment-time interaction was intermediate in significance (p = 0.74 at HNA, p = 0.66 at Springbank).

Table 5: Average values (μ) and standard deviations (SD) for species richness, at Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON.

Treatment	Huron Natural Area				Springbank Farm					
	May 22, 2012		June 19, 2012		May 22, 2012		June 16, 2012		August 28, 2012	
	μ	SD	μ	SD	μ	SD	μ	SD	μ	SD
Control	7.00	0.82	7.50	1.00	9.25	3.20	13.75	2.50	9.50	1.29
Biochar	8.00	1.15	7.75	2.22	6.75	1.71	12.25	2.22	10.00	0.82
Fertilizer	7.75	1.26	7.00	2.16	7.00	3.37	16.00	1.41	9.50	1.91
Legume	8.50	2.38	7.00	1.41	7.25	2.06	14.75	3.86	10.00	1.63
Combination	7.75	0.50	7.50	1.29	10.25	2.75	13.50	3.32	8.75	1.71
Overall	7.80	1.32	7.35	1.53	8.10	2.79	14.05	2.80	9.55	1.43

Table 6: Significance values for repeated-measures ANOVAs for species richness, for Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON.

Significance (p)	Huron Natural Area	Springbank Farm
Treatment	0.92	0.98
Time	0.35	0.22
Treatment : Time	0.74	0.66

At HNA, subplot-level mean species richness (SR) was positively correlated (Pearson product-moment correlation) with pre-treatment N, K and OM levels (see Table 7). It was negatively correlated with pre-treatment P and pH. Statistically significant correlations were those between species richness and pre-treatment N ($r = 0.42$; $p = 0.07$; see Figure 5a) and OM ($r = 0.52$; $p = 0.02$; see Figure 5b).

Mean SR at that site was positively correlated with post-treatment N, and negatively correlated with pH and OM (see Table 7). The relationships with P and K were both approaching zero ($r = 0.02$ and $r = -0.01$, respectively). There were no statistically significant relationships between SR and any of the post-treatment soil factors studied.

At Springbank, subplot-level mean SR was positively correlated with pre-treatment (October 2011) K and pH, while it was negatively correlated with N, P and OM. None of these trends were statistically significant (see Table 8). The directions of the trends are all the same as those between SR and post-treatment soil factor levels. The negative correlation between N and SR was highly significant ($r = -0.67$; $p < 0.001$; see Figure 6), while the positive relationship with pH was near significance ($r = 0.36$; $p = 0.11$).

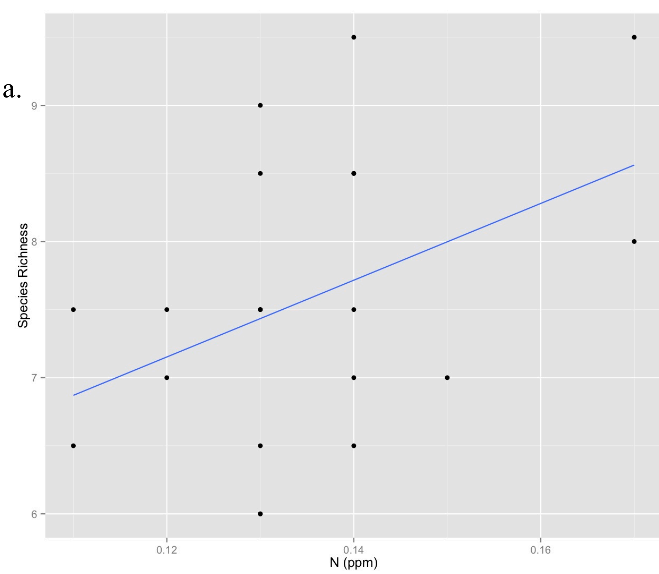
It is noteworthy that at HNA, for every soil factor, the correlation with SR was tightest for its pre-treatment (2011) value (with a single exception, pH). Meanwhile the reverse was true at Springbank. For every category the strongest relationship was between SR and the post-treatment (2012) soil nutrient or pH level. Also noteworthy is the opposite trends between SR and N at the two sites. While 2011 and 2012 N levels were positively correlated with SR at HNA, the relationship was negative for both sampling events at Springbank. Trends were also opposite at both sites for pH, and 2011 OM.

Table 7: Pearson product-moment correlations between average species richness in 2012 and pre- (2011) and post-treatment (2012) soil factors for experimental plots at Huron Natural Area, Kitchener, ON. *r*: correlation coefficient; *p*: probability of α-error (p<0.1).

Correlation	2011		2012	
	r	*p*	*r*	*p*
Nitrate (ppm)	0.42	0.07*	0.15	0.54
Phosphorus (ppm)	-0.13	0.57	0.02	0.92
Potassium (ppm)	0.27	0.24	-0.01	0.96
pH	-0.05	0.82	-0.11	0.62
Organic Matter (%)	0.52	0.02*	-0.06	0.81

* Significant correlation.

a.



b.

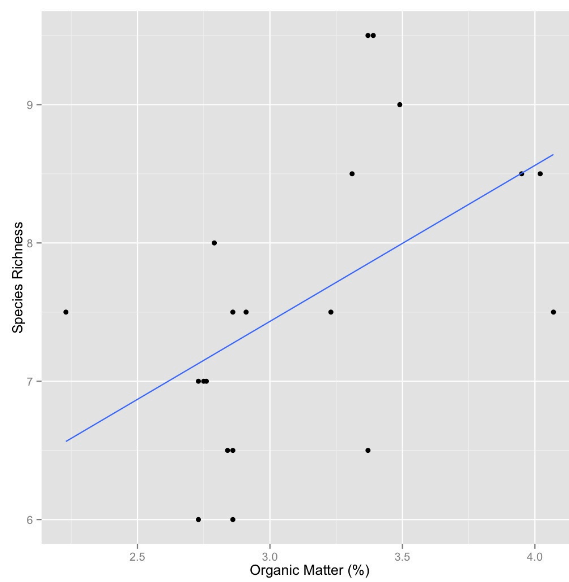


Figure 5: Graphs illustrating statistically significant correlations between species richness and (a) pre-treatment (2011) nitrate concentration (r = 0.42) and (b) pre-treatment (2011) organic matter concentration (r = 0.52), for Huron Natural Area, Kitchener, ON.

Table 8: Pearson product-moment correlations between average species richness in 2012 and pre- (2011) and post-treatment (2012) soil factors for experimental plots at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON. *r*: correlation coefficient; *p*: probability of α-error (*p*<0.1).

Correlation	2011		2012	
	r	*p*	*r*	*p*
Nitrate (ppm)	-0.19	0.41	-0.67	0.001*
Phosphorus (ppm)	-0.26	0.26	-0.27	0.25
Potassium (ppm)	0.19	0.42	0.23	0.33
pH	0.30	0.19	0.36	0.11
Organic Matter (%)	-0.15	0.53	-0.18	0.44

* Significant correlation.

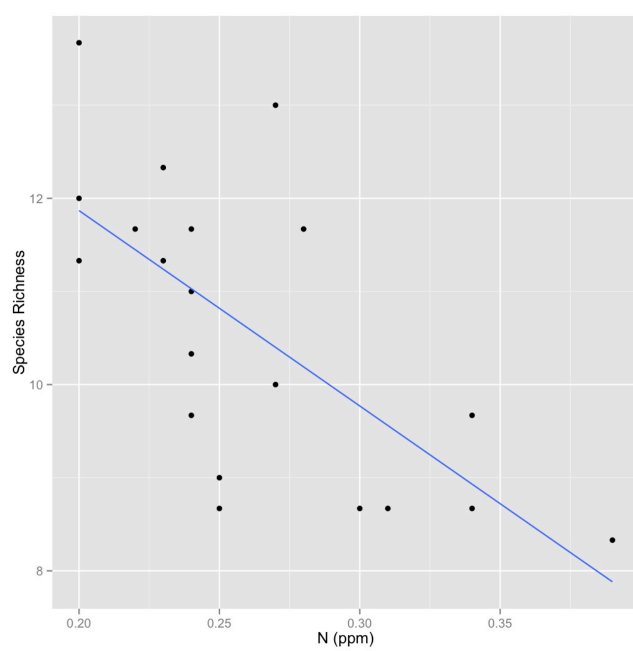


Figure 6: Graph illustrating the statistically significant correlation between species richness and post-treatment (2012) nitrate concentration (r = -0.67), at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON.

4.1.3 Vegetation – Biomass

The average above-ground biomass (BM) at HNA was 33,926 kg/ha, while at Springbank it was 67,138 kg/ha. An unpaired Student's *t*-test indicates that these values are significantly different ($p < 0.0001$). BM means by treatment, at HNA, ranged from 29,340 (Legume) to 42,090 kg/ha (Fertilizer), while at Springbank they ranged from 43,830 (Biochar) to 90,990 kg/ha (Legume; see Figure 7). At HNA, a univariate ANOVA with Tukey's *post hoc* indicated no significant differences between treatments ($p = 0.681$), while at Springbank, the same test pointed to a significant difference between the means of the biochar-treated and the legume-seeded plots ($p = 0.066$; see Figure 7).

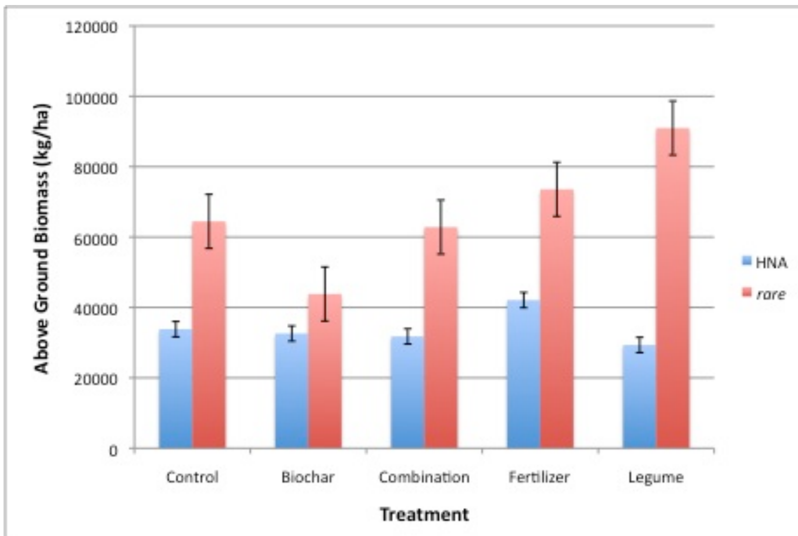


Figure 7: Histogram illustrating average above-ground biomass (kg/ha) by treatment, with 95% confidence intervals, for Huron Natural Area, and Springbank Farm (*rare* Charitable Research Reserve). Chart produced in Excel (Microsoft Excel for Mac, version 12.3.6).

At HNA, BM levels, harvested, dried and weighed in October 2012, were positively correlated with pre-treatment P, K and pH levels, while they were negatively correlated with levels of N and OM (see Table 9). The positive relationship between BM and P was significant ($r = 0.52$; $p = 0.02$; see Figure 7a), while the negative relationship with N was nearly significant ($r = -0.37$; p = 0.11).

BM was positively correlated with post-treatment N, P, K and OM levels. It trended negatively only with 2012 pH levels (see Table 9). It is noteworthy that the correlations between BM and 2012 levels of N, pH and OM were all opposed in direction to those between BM and the 2011 reading for the same soil factor. The only significant relationship was between BM and P, with the same correlation coefficient as with the 2011 levels ($r = 0.52$; $p = 0.02$; see Figure 7b). The negative correlation between BM and pH approached significance ($r = -0.36$; $p = 0.12$).

At HNA, the relationships between BM and individual soil factors were not notably stronger for a particular sampling season. Correlations for N and OM were stronger with 2011 levels, as opposed to those for K and pH, who trended more closely with 2012 levels. The correlation coefficient for P was equal for both years. P was the only soil factor at the site whose level did not change significantly between seasons (see Table 3a).

At Springbank, BM trended positively with levels of all five 2011 soil factors measured (see Table 10). Four of the correlations were statistically significant: BM and N (r = 0.62; p = 0.004; see Figure 8a), P (r = 0.54; p = 0.01; see Figure 9a), K (r = 0.61; p = 0.005; see Figure 10a) and OM (r = 0.62; p = 0.004; see Figure 11). The positive relationship between BM and pH was reasonably strong (r = 0.33; p = 0.15). This is the first and only case, for species richness or biomass alike, where nitrate and pH have trended in the same direction.

BM was positively correlated with all 2012 soil factors besides pH (see Table 10). Correlations between BM and N (r = 0.49; p = 0.03; see Figure 8b), P (r = 0.44; p = 0.05; see Figure 9b) and K (r = 0.45; p = 0.05; see Figure 10b) were all significant. As opposed to the trend with 2011 pH levels, BM was negatively correlated with 2012 pH. Neither relationship was significant, however. The soil factors whose levels changed significantly between 2011 and 2012 at Springbank were N, pH and OM (see Table 3b).

Unlike at HNA, there was a clear pattern with respect to the relationships between biomass and soil factors at Springbank. Although the direction was the same between years for each factor (except for pH), BM correlated best with the pre-treatment levels than those measured at the end of the field season. This contrasts with the pattern seen for species richness, whereby SR correlated most strongly with all 2012 soil factors at the site (see section 4.1.2, Table 8).

At both field sites, a negative correlation emerges when exploring the relationship between mean species richness and above-ground biomass (see Table 11). At HNA, the relationship fairly weak (r = -0.15; p =0.58) while it was slightly stronger at Springbank (r = -0.22; p = 0.35); it was not statistically significant at either site.

Table 9: Pearson product-moment correlations between above-ground biomass in 2012 and pre- (2011) and post-treatment (2012) soil factors for experimental plots at Huron Natural Area, Kitchener, ON. *r*: correlation coefficient; *p*: probability of α-error (p<0.1).

Correlation	2011		2012	
	r	*p*	*r*	*p*
Nitrate (ppm)	-0.37	0.11	0.21	0.37
Phosphorus (ppm)	0.52	0.02*	0.52	0.02*
Potassium (ppm)	0.18	0.44	0.24	0.31
pH	0.15	0.54	-0.36	0.12
Organic Matter (%)	-0.14	0.54	0.11	0.64

* Significant correlation.

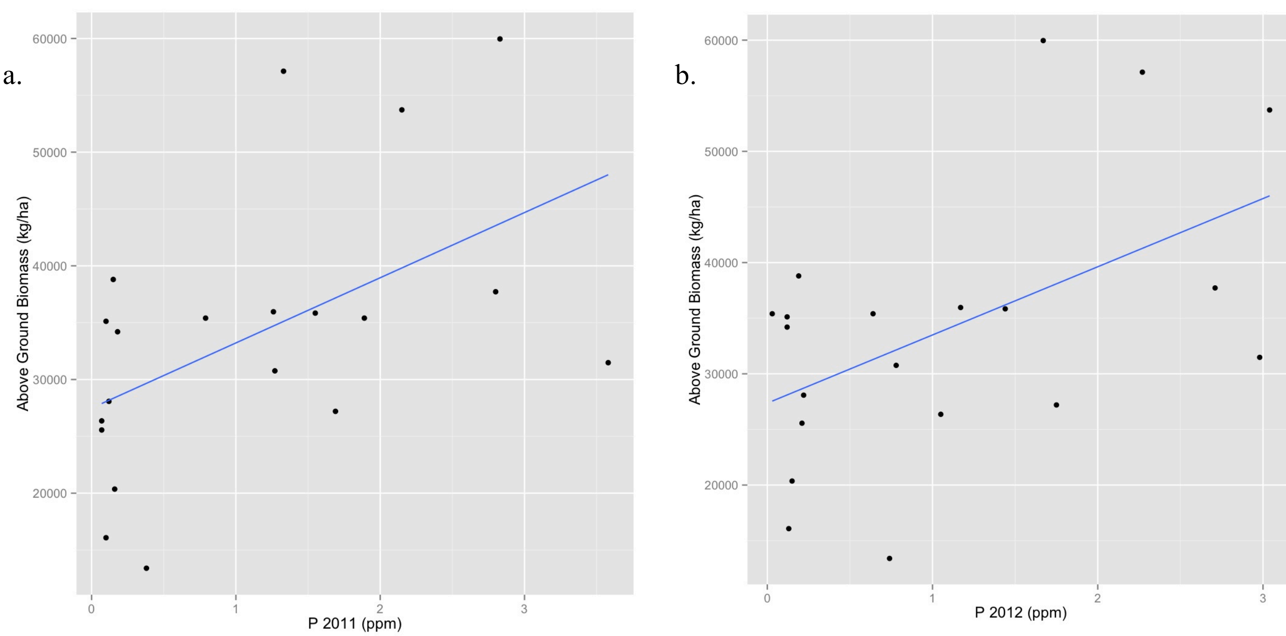
a.
b.


Figure 8: Graphs illustrating statistically significant correlations between above-ground biomass and (a) pre- (2011; r = 0.52) and (b) post-treatment (2012; r = 0.52) Phosphorus concentrations, for Huron Natural Area, Kitchener, ON.

Table 10: Pearson product-moment correlations between above-ground biomass in 2012 and pre- (2011) and post-treatment (2012) soil factors for experimental plots at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON. *r*: correlation coefficient; *p*: probability of α-error (p<0.1).

Correlation	2011		2012	
	r	*p*	*r*	*p*
Nitrate (ppm)	0.62	0.004*	0.49	0.03*
Phosphorus (ppm)	0.54	0.01*	0.44	0.05*
Potassium (ppm)	0.61	0.005*	0.45	0.05*
pH	0.33	0.15	-0.27	0.25
Organic Matter (%)	0.62	0.004*	0.24	0.30

* Significant correlation.

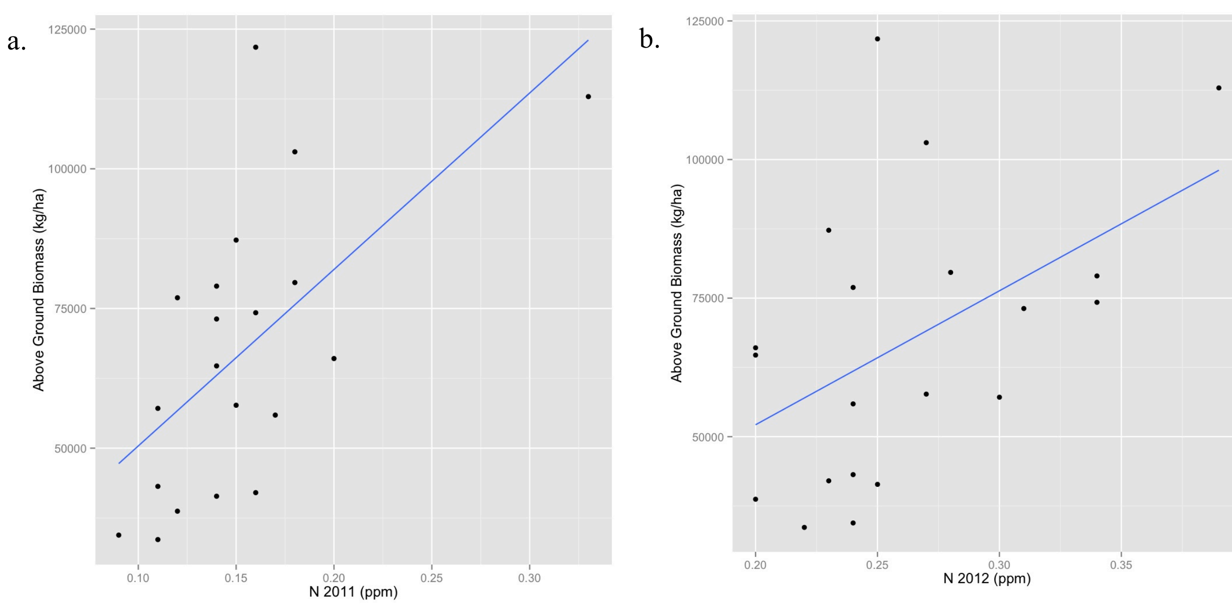


Figure 9: Graphs illustrating statistically significant correlations between above-ground biomass and (a) pre- (2011; r = 0.62) and (b) post-treatment (2012; r = 0.49) nitrate concentrations at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON.

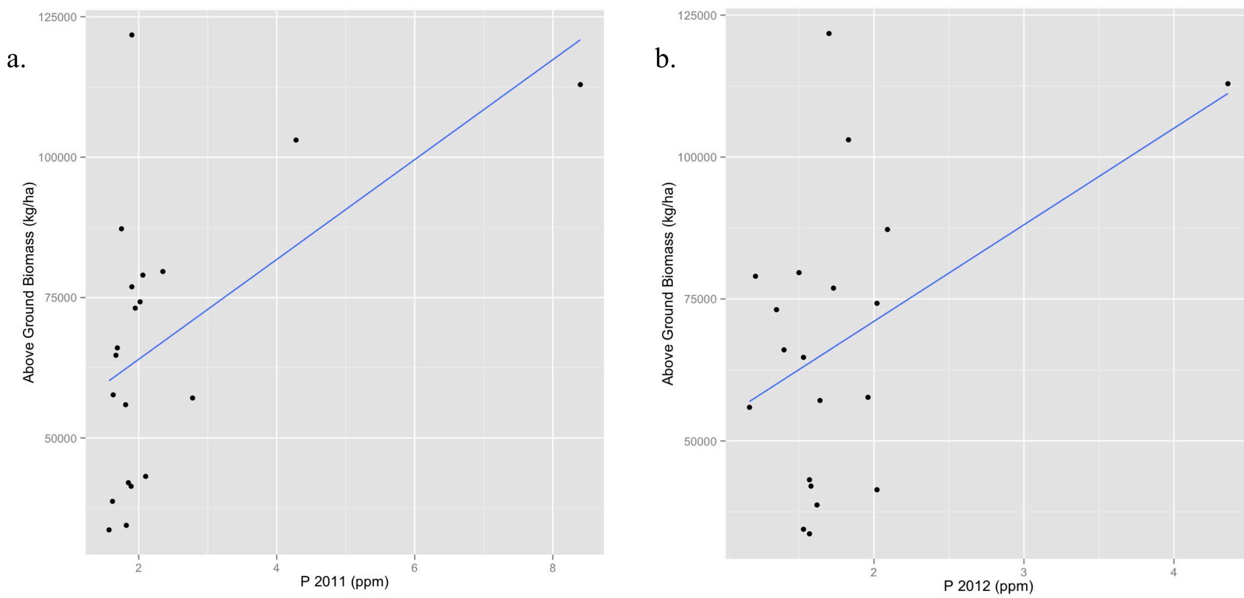


Figure 11: Graphs illustrating statistically significant correlations between above-ground biomass and (a) pre- (2011; r = 0.54) and post-treatment (2012; r = 0.44) Phosphorus concentrations at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON.

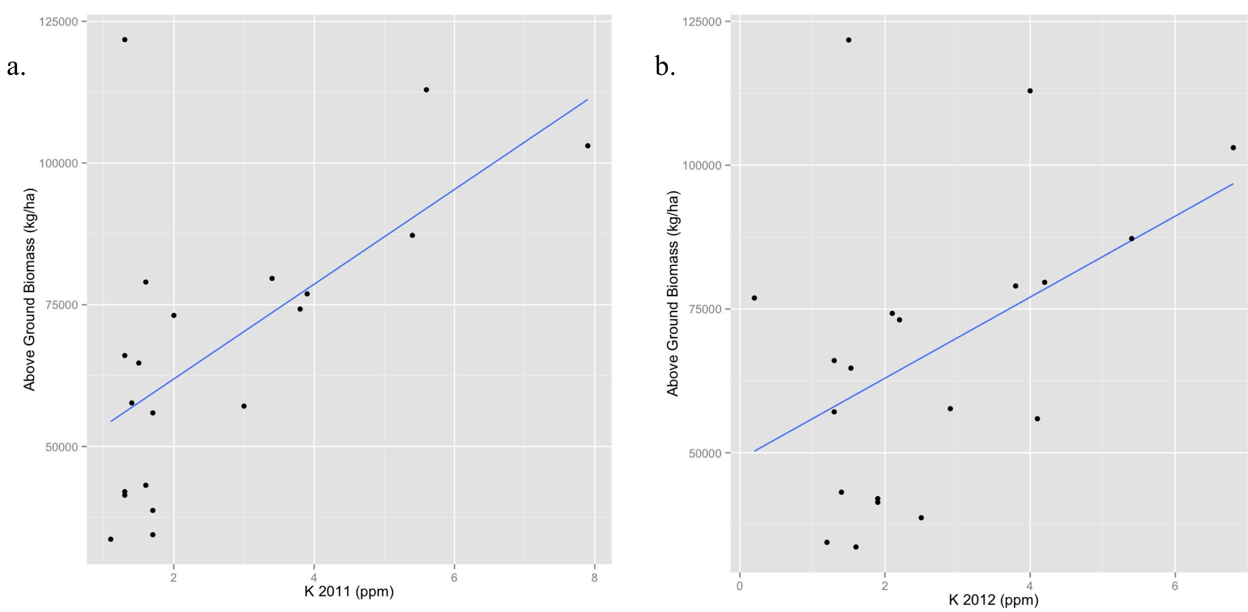


Figure 10: Graphs illustrating statistically significant correlations between above-ground biomass and (a) pre- (2011; r = 0.61) and post-treatment (2012; r = 0.45) potassium concentrations at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON.

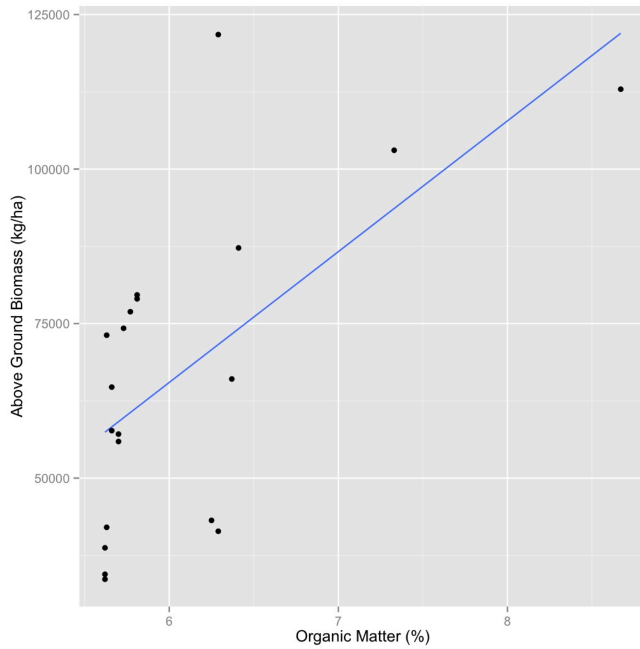


Figure 12: Graph illustrating the statistically significant correlation between above-ground biomass and pre-treatment (2011) organic matter concentration (*r* = 0.62), at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON.

Table 11: Pearson product-moment correlations between above-ground biomass and average species richness in 2012 for experimental plots at Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON.

Correlation	Huron Natural Area		Springbank Farm	
	r	*p*	*r*	*p*
Species Richness	-0.13	0.58	-0.22	0.35

4.1.4 Soil temperature and moisture

The average per-subplot soil temperature at HNA, measured over 9 sampling events throughout the spring and summer of 2012, was of 23.85°C (see Table 12). The range of seasonal averages, by treatment, was from a high of 24.37°C, in the legume-seeded subplots, to a low of 23.51°C, in the biochar-treated subplots. On individual sampling dates, average per-treatment values ranged from 17.33°C to 32.33°C. The average per-subplot soil moisture at the same site was of 32.81%, ranging from a high of 34.93%, in the control subplots, to a low of 30.83%, in the combination subplots. On individual sampling dates, average per-treatment values ranged from 0.00% to 79.50%.

Meanwhile, the average per-subplot soil temperature at Springbank, measured over 10 sampling events throughout the spring and summer of 2012, was of 22.08°C (see Table 13). The treatment per-subplot seasonal averages ranged from a high of 22.33°C, in control subplots, to a low of 22.01°C in biochar-treated subplots. On individual sampling dates, average per-treatment values ranged from 19.33°C to 32.75°C. The average per-subplot soil moisture at the site was of 52.35%, ranging from a high of 55.83% in combination subplots, to a low of 47.97%, in legume-seeded subplots. On individual sampling dates, average per-treatment values ranged from 4.25% to 83.50%.

Temperatures were not significantly different at either sampling site (paired *t*-test, adjusting for unequal sample sizes). Soil moisture was, however, significantly lower at HNA than at Springbank ($p = 0.07$). It is noteworthy that at both sites, the coolest subplots were those treated with biochar.

A repeated-measures ANOVA indicated no significant differences between treatments, or treatment-time interactions, for soil temperature at either site (see Table 14). However soil temperatures did vary significantly over time at both sites ($p = 0.001$ at HNA; $p < 0.0001$ at Springbank). There were no significant differences between treatments, nor were there significant treatment-time interactions, at either site in terms of soil moisture (see Table 15). Just as with soil temperature, however, soil moisture fluctuated in a statistically significant manner over the course of the field season ($p < 0.0001$ at HNA; $p = 0.07$ at Springbank).

At HNA, subplot species richness was positively, but not significantly, correlated with both soil temperature ($r = 0.16$) and moisture ($r = 0.04$; see Table 16). At the same site, above-ground biomass was significantly negatively correlated with average subplot soil temperature ($r = -0.64$, $p = 0.002$; see Figure 12a), and significantly positively correlated with soil moisture ($r = 0.38$, $p = 0.1$; see Figure 12b).

Trends for the relationships described above between vegetation and soil physical factors were similar at Springbank. Species richness was positively correlated with soil temperature ($r = 0.07$) and moisture ($r = 0.23$). Above-ground biomass was negatively correlated with soil temperature ($r = -0.25$), while it was positively correlated with moisture ($r = 0.23$). None of the preceeding relationships were statistically significant.

Table 12: Average values for soil temperature and moisture on each sampling date in 2012 for different treatments, at Huron Natural Area, Kitchener, ON. μ: overall average; C: control; B: biochar; F: fertilizer; L: legume; X: combination.

Sampling Date	Soil Temperature						Soil Moisture					
	μ	C	B	F	L	X	μ	C	B	F	L	X
May 21, 2012	24.24	24.10	24.03	23.88	24.83	24.38	31.35	43.75	26.25	33.00	33.75	20.00
May 27, 2012	24.43	24.15	24.63	23.73	25.23	24.43	33.25	42.50	27.50	40.00	35.00	21.25
June 5, 2012	18.13	17.50	18.48	18.05	18.65	17.98	71.15	72.00	70.00	64.25	79.50	70.00
June 10, 2012	17.93	17.70	18.28	18.25	18.10	17.33	74.25	73.75	72.50	68.25	79.50	77.25
June 24, 2012	22.77	22.98	22.58	22.58	23.50	22.20	19.85	23.00	20.00	21.00	16.50	18.75
June 28, 2012	30.64	30.13	29.85	30.95	29.95	32.33	38.40	35.00	41.25	32.50	41.25	42.00
July 13, 2012	30.22	30.40	29.90	30.18	30.93	29.68	0.10	0.00	0.50	0.00	0.00	0.00
July 19, 2012	23.69	24.65	22.53	23.53	24.60	23.15	20.75	22.75	19.50	23.25	18.75	19.50
August 17, 2012	22.72	24.28	21.65	22.78	23.00	21.90	27.10	25.25	30.00	26.25	27.00	27.00
August 22, 2012	23.69	23.38	23.18	23.73	24.93	23.23	11.90	11.25	18.25	6.25	11.25	12.50
Average	23.85	23.93	23.51	23.77	24.37	23.66	32.81	34.93	32.58	31.48	34.25	30.83

Table 13: Average values for soil temperature and moisture on each sampling date in 2012 for different treatments, at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON. μ: overall average; C: control; B: biochar; F: fertilizer; L: legume; X: combination.

Sampling Date	Soil Temperature						Soil Moisture					
	μ	C	B	F	L	X	μ	C	B	F	L	X
May 21, 2012	23.59	23.80	23.78	23.35	24.08	23.93	52.00	51.25	62.50	51.25	43.75	61.25
May 27, 2012	23.79	24.00	24.25	23.23	24.40	23.75	54.00	51.25	63.75	53.75	45.00	63.75
June 5, 2012	21.19	21.45	21.10	20.78	21.60	21.15	43.70	49.75	46.50	33.00	41.75	50.00
June 10, 2012	22.01	22.03	22.70	21.00	21.65	22.20	50.50	55.00	52.50	47.50	47.50	55.00
June 16/19, 2012	22.99	23.68	21.55	24.25	22.63	23.13	68.89	68.75	62.33	83.50	59.00	66.25
June 24, 2012	20.15	20.25	20.15	19.88	20.28	20.23	58.15	59.75	52.75	64.75	55.75	62.75
June 28, 2012	31.19	32.75	31.38	30.40	30.68	32.00	41.15	40.00	38.75	46.25	38.75	41.25
July 13, 2012	20.32	20.43	20.28	19.98	20.75	20.30	10.30	10.75	10.75	16.50	4.25	12.00
July 19, 2012	20.19	20.20	19.65	20.35	20.68	20.15	62.38	65.75	55.88	65.75	57.50	65.38
August 17, 2012	19.69	19.45	19.33	20.20	19.88	19.65	64.33	65.00	61.25	65.50	63.13	65.50
Average	22.08	22.33	22.01	21.94	22.21	22.22	52.35	53.95	51.88	54.05	47.97	55.83

Table 14: Significance values for repeated-measures ANOVAs for soil temperature, for Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON.

Significance (p)	Huron Natural Area	Springbank Farm
Treatment	0.93	0.86
Time	0.001*	<0.0001*
Treatment : Time	0.95	0.98

* Significant difference over time.

Table 15: Significance values for repeated-measures ANOVAs for soil moisture, for Huron Natural Area and Springbank Farm (*rare* Charitable Research Reserve), Region of Waterloo, ON.

Significance (p)	Huron Natural Area	Springbank Farm
Treatment	0.90	0.54
Time	<0.0001*	0.07*
Treatment : Time	0.56	0.50

* Significant difference over time.

Table 16: Matrix of Pearson product-moment correlations between 2012 average, per-subplot, species richness, above-ground biomass, soil temperature (°C) and soil moisture (%) at Huron Natural Area, Kitchener, ON.

Factor	Species Richness	Above-ground Biomass	Soil T°	Soil Moisture
Species Richness		-0.13	0.16	0.04
Above-ground Biomass	-0.13		-0.64**	0.38*
Soil T°	0.16	-0.64**		-0.23
Soil Moisture	0.04	0.38*	-0.23	

* Significant correlation (*p*<0.1).

** Significant correlation (*p*<0.01).

a.

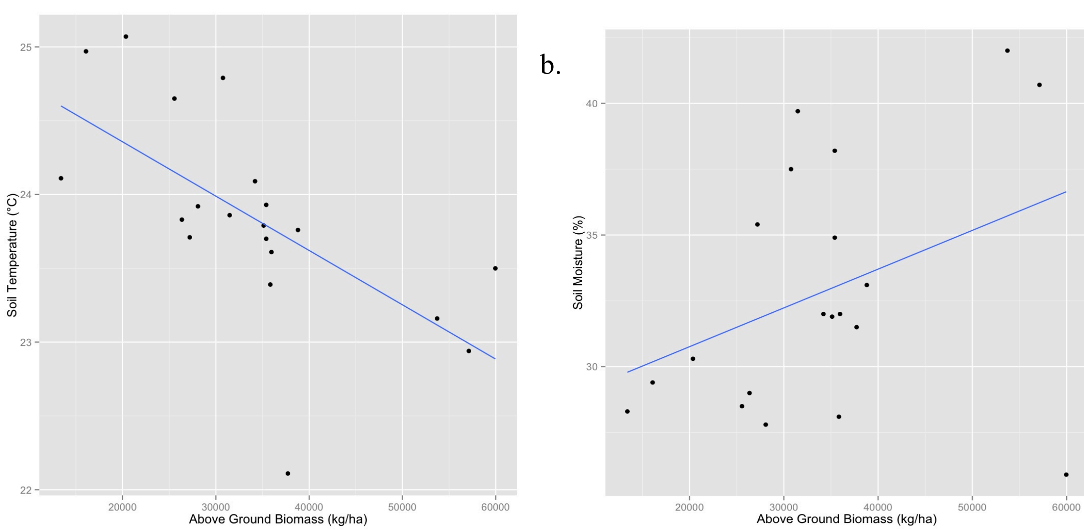


Figure 13: Graphs illustrating statistically significant correlations between above-ground biomass and (a) average subplot soil temperature (°C; *r* = 0.38, *p* = 0.1) and (b) average subplot soil moisture (%) in 2012 (*r* = -0.64; *p* = 0.002) at Huron Natural Area, Kitchener, ON.

Table 17: Matrix of Pearson product-moment correlations between 2012 average, per-subplot, species richness, above-ground biomass, soil temperature (°C) and soil moisture (%) at Springbank Farm (*rare* Charitable Research Reserve), Cambridge, ON.

Factor	Species Richness	Above-ground Biomass	Soil T°	Soil Moisture
Species Richness		-0.22	0.07	0.23
Above-ground Biomass	-0.22		-0.25	0.23
Soil T°	0.07	-0.25		-0.31
Soil Moisture	0.23	0.23	-0.31	

4.2 Discussion

Increased paces of environmental degradation and species extinction, combined with a growing acknowledgement of the multifaceted benefits of natural areas, have lead to an increase in the societal demand for ecosystem restoration. For this reason, there is an ever-greater need for multivariate research in ecological restoration methods (Suding, 2011). While protocols are relatively well established for certain ecosystems (for example see Packard and Mutel, 1997), scientists have only recently begun to explore urban meadow restoration from a methodological perspective (Klaus, 2013), and there is still no published work on the topic from North America. Working on the premise that the first aspect to address in restoring a terrestrial ecosystem is the soil (cf. Bradshaw, 1997), this study tested a variety of techniques for meadow restoration, each one aiming to improve soil conditions. The techniques ranged from routine (fertilizer application) to promising but poorly documented (native legume planting) to relatively novel (biochar treatment).

One caveat is that the effects of such treatments often only appear some time after their application. Forage legume species, such as clovers, can take several seasons to accrue significant nitrogen capital in the soil, and native species may take even longer (Dancer et al., 1977b). The full impact of biochar on soil biota, nutrient cycling and plant growth can take years to become fully apparent (Lehmann & Joseph, 2009; Lehmann et al., 2011), as the soil community adapts to the new conditions provided by the material. Native grassland species often do not appear in a sward until several years after they are seeded (Fischer et al., 2013a). Therefore, the results observed in this study should be regarded as a test of the early impacts of restoration. Such early results are interesting in their own right, and they also offer context to future monitoring efforts (Murphy, 2005). Indeed many of the results discussed in this chapter are significant and offer interesting insight into meadow ecosystem dynamics.

4.2.1 Soil conditions

Analysis of edaphic factors at both study sites demonstrated a number of important differences in quality (see Table 2). The soil at HNA was sandy, while that at Springbank was a clay-loam. For both sampling events, all nutrient (besides nitrogen on 2011) and organic matter levels were lower at HNA than at Springbank. pH at both sites was in the basic range, but levels at HNA were significantly higher than those at Springbank. In all likelihood, more differences exist, in terms of other nutrients and micronutrients, as well as mycorrhizal and soil microfaunal diversity.

Such disparities are reflective of natural fertility gradients, but they likely also relate to historical land use type and intensity. Importantly, they have implications regarding successional pathways and restoration potential (Hobbs & Cramer, 2007). Soil texture has major impacts on such factors as nutrient availability, water holding capacity, and rates of leaching: sandy soils drain more quickly than soils with a higher proportion of clay, and therefore have higher rates of leaching and lower "inherent fertility" (Dancer et al., 1977b). While a major focus of this study was on increasing the concentration of nitrogen in the soil, water supply can also be a limiting factor in plant growth. As water becomes scarce, mineralization rates decrease and thus plant-available nitrogen dwindles. If nitrogen is already limiting, then the deficient plants' shorter roots may compound the problem. On the other hand, if water levels increase to the point of saturation, oxygen can become scarce. Productivity is highest when drying follows periods of wetness, pumping oxygen through the soil (Thomson & Troeh, 1973). This suggests that restoration methods that increase the availability of moisture to a reasonable extent might be as effective in increasing the success of plantings as those that simply increase nutrient concentrations. Biochar, as recalcitrant organic matter with a fine particle size, is known to increase soil moisture, especially in sandy soils. While this result was not evident in the results of this study, other published works have established the correlation (Sohi et al., 2009). If this property is demonstrated consistently, then it may serve as an important tool for restoration practitioners.

In this study, fertilizer-treatment plots measured in the fall of 2012 did not have significantly higher concentrations of nitrogen than controls. Given that they did not have higher levels of productivity than other treatments, it seems likely that the nutrients were leached before the growing seedlings could use them. According to Thompson and Troeh (1973), soils are generally leached if they are subjected to cool and moist spring conditions. It is possible that applying a slow-release fertilizer, or adding it gradually over the growing period, instead of prior to seeding, might be a more effective and efficient use of the resource.

As mineral fertilizer provides an immediate supply of available nitrogen, it undoubtedly has its place among restoration methods for extreme cases of deficiency, or where no nutrient capital exists. On the other hand, it needs to be applied repeatedly, and its use is relatively costly (Dancer et al., 1979). Its value on old-fields dominated by ruderal species, where a sward is already established, can be put into question. An alternative is the use of symbiotic nitrogen-fixing microorganisms, which can fix between 50 and 200 kg/ha/year (1-4% of which becomes available to plants). At that rate, a significant store can build up relatively quickly. Given that planting nitrogen-fixing species is cheaper and requires less effort than repeated fertilizer applications, Bradshaw (2002) has concluded that the former is the superior restoration tool in temperate ecosystems where a large nitrogen capital is required.

Exotic legume species (e.g. *Trifolium repens, T. pratense, Medicago lupulina*) are already prevalent at both field sites (Kastner, pers. obv.). Allowing these non-native species to remain in the community, while promoting the establishment of native legume species, will allow the eventual build-up of a significant nitrogen store. There was a consistent increase in nitrogen levels across all treatments at both sites between sampling seasons (statistically significant at Springbank). The source of this accretion is not obvious: it may be due to any combination of leached nitrogen fertilizer, atmospheric deposition, symbiotic fixation and the mineralization of decomposing plant material incorporated into the soil during tillage. Mineralization rates are known to be especially high during hot and dry weather (Thompson and Troeh, 1973), as was experienced over the summer of 2012.

Available phosphorus levels can be a limiting factor for plant growth in some environments. This is due to the fact that the nutrient is not fixed from atmospheric gases, as is nitrogen, but instead is derived from mineral sources. Phosphorus has low levels of solubility (0.1% per acre), and therefore low plant-availability, but this is balanced by its low leaching rates. However, in mildly alkaline conditions with an abundance of calcium, typical of the Region of Waterloo (Martin & Frind, 1998), soluble P reverts to an insoluble form in hydroxyapatite ($Ca_5(PO_4)_3OH$; Thompson and Troeh, 1973). Interestingly, at HNA, the site with the more basic soil pH, above-ground plant biomass was significantly, and exclusively, correlated with phosphorus levels for both sampling events. This suggests that plant growth at that site may in fact be phosphorus-limited.

As is typical for urban (Fischer et al., 2013b) and old field (Hobbs & Cramer, 2007) sites, there was significant spatial heterogeneity in soil nutrient distribution. For example, in 2012, phosphorus levels per subplot ranged from 0.07 to 3.58 and 1.54 to 8.30 ppm, while potassium levels ranged from 0.2 to 1.3 and 1.1 to 7.9 ppm, at HNA and Springbank, respectively. At HNA, there were concentrations of rocky substrates in certain subplots that likely bear evidence to past soil disturbance at the site. At Springbank, vigorous and immediate plant growth occurred in one subsection of the plot following tillage in the fall, indicating a localized area of particularly high fertility (Kastner, pers. obv.).

Either the treatments were ineffective, or in-plot heterogeneity overrode any treatment-related effects thus far on plant establishment (see sections 4.1.2 and 4.1.3). Biomass was correlated specifically with phosphorus levels at HNA, while it was associated with various soil factors (N, P, K, OM) at Springbank. At both sites vegetation measures were generally more strongly correlated with pre-treatment levels of soil factors than with post-treatment levels. It seems, therefore, that the meadow community reacted more strongly to local, in-subplot conditions (those that were present before any treatment was applied), than to any treatment effects.

Results from other recent urban grassland restoration projects indicate that spatially heterogeneous sites allows the establishment of native meadow species among the dominant ruderal vegetation, because of the wide variety of microhabitats available (Fischer et al., 2013b). This reflects the positive relationship between heterogeneity and species richness predicted by classical niche theory (Kadmon & Allouche, 2007).

4.2.2 Vegetation establishment

The explicit goal of this project was to promote plant species diversity by improving soil conditions on old fields set aside for conservation. As in Foster and Gross (1998), a limited number of native species not present (or uncommon) at the experimental sites were seeded in order to determine the effectiveness of the treatments in aiding in plant establishment. This promotion of diversity is common practice in habitat restoration projects, as a logical extension of the aim of conservation projects to conserve native extant species (Stevens et al., 2004). Given that only one of the five species planted (*Rudbeckia hirta*) successfully germinated and established in the treatment plots, these results were omitted from the analysis. As such, only total species richness was used as an indicator of treatment success. Total above-ground biomass was also assessed, as a measure of productivity (as in Mittelbach et al., 2001).

The results indicated that soil fertility had a major influence on species richness in our plots. At Huron Natural Area, species richness was significantly and positively correlated with 2011 soil nitrogen and organic matter levels. Conversely, at Springbank Farm, species richness was exclusively, significantly, and negatively correlated with 2012 soil nitrogen levels. Soil treatments did not have a major effect on species richness at either study site. The opposing trends of species richness with respect to soil fertility levels seems to indicate that either a threshold exists whereby beyond a certain level of fertility competitive exclusion prevents the establishment of certain species, or that the two different sites have differing limiting nutrients.

In studies wherein limiting nutrients are added to herbaceous plant communities, a frequently described result is increased productivity concomitant to a decrease in species richness (Foster & Gross, 1998; Vermeer & Berendse, 1983). Indeed, in a nitrogen-addition experiment in a successional old field in Michigan, Foster and Gross (1998) reported that nitrogen addition promoted plant productivity, which in turn decreased species richness by suppressing subordinate forb species. This is congruent with qualitative observations by the author at Springbank Farm, whereby tall grasses dominated highly productive subplots, while less productive subplots hosted a variety of grass and forb species. This appears to occur through competition for light by living plants, as well as shading and mechanical impediment of germination and growth by litter (Foster & Gross, 1998). Given that a natural or human-mediated fire regime may have been historically important in Southern Ontario (see Section 2.2.5), it is conceivable that litter accumulation on the present-day, fire-suppressed landscape is having a particularly negative impact on native fire-adapted species.

The productivity-species richness relationship for terrestrial vascular plant communities has generally been found to be unimodal, or "hump-shaped" (Mittelbach et al., 2001; Waide et al., 1999). Therefore, at most geographical scales, the number of species within a habitat type will increase up to a certain level of productivity (generally measured as standing biomass), beyond which competitively dominant plants will dominate and exclude less adapted species. The mechanisms by which this functions vary, but can include shading and litter accumulation, as discussed above, as well as competition for resources and allelopathy. Guo and Berry (1998) describe a scenario whereby plants must trade-off between competition and colonization at different soil nutrient levels, allowing a maximum number of species to coexist at intermediate levels.

The question of scale is an important consideration when interpreting these trends. For example, the more productive site in our study also hosted the greater number of species. However, within each site, the most productive subplots were also the most species-poor. It would be interesting to survey a greater variety of urban meadows to determine whether these trends are widespread. It seems that habitat heterogeneity is important in these cases in mediating species richness, by providing niches where species may "escape" competitively dominant plants. A review of publications describing the biomass-species richness relationship also found that trends within microhabitat sites differed from those across habitats (Guo & Berry, 1998).

An interesting result is the discrepancy between the apparent limiting nutrient and the one affecting species richness at Huron Natural Area. Biomass at the site was significantly positively correlated with phosphorus levels. Species richness, however, was positively correlated with levels of nitrogen and organic matter. It seems, therefore, that a number of different factors affected plant establishment at the site, defying the productivity-species richness relationships described above. At a broad level, it seems that the mechanism whereby higher levels of the limiting nutrient leads to higher productivity and, consequently, the competitive exclusion of less dominant species applies to the study site. Indeed, species richness was negatively correlated with biomass at the site, though the relationship was moderate and not significant. However, it may be that on a relatively species-poor and highly heterogeneous urban site such as this one, a diversity of species can coexist on small pockets of nitrogen and organic matter-rich soil. If the apparent phosphorus limitation of the site is indeed due to its alkaline nature – and thus soluble P is being sequestered by calcium (as proposed in Section 4.2.1) – then at a highly local level, areas of lower pH may have greater concentrations of plant-available P. Both organic matter and certain nitrogenous compounds are known to lower pH due to their propensity for proton release (Thompson & Troeh, 1973). It follows that micro-sites of "release" from phosphorus limitation may exist in areas containing more fertile, organic soil. This could explain the elevated species richness of subplots with higher nitrogen and organic matter levels.

Foster and Gross (1998) found that despite some hindrance by litter, *A. gerardii*, their target native species, was able to germinate from seed and establish mature plants on control plots. On their experimental nitrogen- and/or litter-addition plots, however, the recruitment of that species was fully inhibited. This led them to conclude that propagule availability, as opposed to inhibition from established vegetation, was the major factor preventing the establishment of that species. Their study site, a moderately productive successional field in Michigan, is likely comparable to Springbank Farm, and perhaps in some respects to Huron Natural Area. From a restoration perspective, their result suggests that seeding alone may be sufficient to reestablish absent native species into a grassland ecosystem. While the seeding aspect of this experiment was not successful (except for *R. hirta*), this may be due, at least in part, to its short length. Certainly the results from Springbank Farm, where nitrogen levels and species richness were inversely related, do not suggest that nitrogen-addition is a necessary restoration tool in that context. The situation at Huron Natural Area is more complex. Given the positive relationship observed between nitrogen and species richness, it is conceivable that addition of the nutrient may assist in the establishment of desired species. However, exploiting the natural heterogeneity of the site, or adding a one-off supplement such as biochar, which may promote nutrient cycling, might be preferred. The implications of the apparent phosphorus limitation on the site should be explored further.

4.2.3 Soil temperature and moisture

At both study sites, above-ground plant biomass was negatively correlated with soil temperature, and positively correlated with soil moisture. While the relationships were statistically significant at Huron Natural Area, they were not so at Springbank Farm. It is likely that this inconsistency is due to the lesser water retention capacity of HNA's sandy soils compared to Springbank's clay-loam. This difference in drainage would accentuate any cooling and moisture-holding effects of the standing plant matter and litter. At both study sites, soil moisture and temperature were moderately but not significantly negatively correlated. There were no significant effects of treatment on the soil temperature or moisture of the subplots.

It has previously been reported that biochar has water retention capacities (Sohi et al., 2009) but this was not evident in the results of this experiment. It is possible that the quantity of biochar applied was insufficient to observe a significant result. Alternatively, it may be that the sampling frequency was insufficient to detect major differences. Even if biochar-treated plots only retain moisture for a few hours or days longer than control plots, this may be ecologically significant but difficult to identify without very frequent sampling. There is also evidence in the literature that mean soil temperature and diurnal temperature fluctuations are impacted by biochar, through its effect on soil colour (Sohi et al., 2009). It might be expected that on tilled soil, biochar's dark colour would lead to a significant increase in temperature by absorbing a broader spectrum of solar radiation than the surrounding substrate. This would be especially evident in the springtime when much of the soil surface is exposed to direct sunlight. While no effect was noted in the results of this experiment, this could be due to the quantity of the additive applied, or to the depth at which soil temperature was measured. If biochar did have a significant effect on soil moisture and temperature, this could impact the speed and success of seed germination and plant establishment.

It is a well-established principle in soil science that organic matter is crucial in water retention (Thompson & Troeh, 1973; Bradshaw & Chadwick, 1980), due to its effects on pore size, bulk density and chemical properties. It also acts as a store of nitrogen and other nutrients. A positive feedback mechanism may exist whereby areas with higher organic matter retain more moisture and have higher nutrient levels, allowing a greater plant biomass to develop, which then adds to the organic matter stock through decomposition. At HNA, pre-treatment organic matter – along with nitrogen – was significantly positively correlated with species richness.

Conversely, at Springbank, pre-treatment organic matter – along with along soil nutrients measured – was significantly positively correlated with biomass. While this implies that the two sites are at different points along the productivity-species richness gradient (as discussed in Section 4.2.2), it also emphasizes the importance of organic matter in soil function. Biochar increased organic matter levels at both study sites compared to controls (significantly so at HNA). These results support the notion that biochar may be a useful soil amendment for restoration, particularly at relatively infertile, well-drained sites.

It is difficult to quantify the effects of extreme weather on the experimental results. The months of June and July 2012 were particularly hot and dry in Southern Ontario. Average soil temperatures on June 28, 2012, at a depth of 10cm, were above 30°C at both study sites. Between June 24 and July 19, mean soil temperature was nearly 4°C higher at HNA than at Springbank, while moisture was over 23% lower at the former site.

This is likely related to a combination of soil type and surrounding environment (notably, there is a hedgerow at Springbank which shaded the study plot at certain times of day). The extreme weather had a greater impact at HNA than at Springbank, where plants appeared withered and significant mortality may have occurred throughout July (Kastner, pers. obv.). This occurrence likely had a major impact on biomass measurements at the end of the season. While no specific treatment appeared to fare better than the others in this experiment, adapting restoration strategies to the climate, and the effects of climate change in particular, is now an important consideration (Harris et al., 2006). The negative impacts of the drought, at both sites, were more evident within the tilled plots than on the surrounding vegetation. The merits of tillage as opposed to alternative restoration techniques will be discussed in the following section.

4.2.4 Future directions

The interconnected issues of environmental degradation and species loss are entering the public consciousness as well as the mainstream political agenda. Ecological restoration is at the top of the list of policy solutions, and a wide range of actors is engaging in the practice. A key role for academic researchers will be providing timely and practical restoration guidance. This requires identifying processes that can be modulated to great effect, and this is best achieved through a multivariate approach (Suding, 2011). A pragmatic, and increasingly prevalent, perspective on the role of restoration science is that the focus should be on efficiency and optimal return on investment (De Groot et al., 2013). Hobbs et al. (2011) describe the process as the search for leverage points: opportunities for intervention whereby a small change can lead to a major shift in behaviour. Such leverage points can lie both within internal system properties and in rules and paradigms constructed around the system.

Within the realm of grasslands, one relatively entrenched principle – indeed it was adopted by this project – is that a necessary step in restoring a field is to till it. This appears to stem from the discipline's agricultural, or "gardening", legacy (Jordan, 2003). An alternative method, albeit rarely employed, is to simply spread the seed of target species within existing old-field sod. Called interseeding, this method was first investigated, with success, by Henry Greene in tallgrass prairie restoration experiments at the University of Wisconsin-Madison Arboretum.

In a prairie ecosystem, not only do desired species eventually establish themselves within, and then replace existing vegetation, but also the method seems to favour rare species that are absent from sites restored by traditional till-and-sow methods (Jordan, 2003). This method has important theoretical implications, raising questions about succession, establishment niches and grassland dynamics. The idea that tillage is not necessary, and may in fact sometimes be counterproductive, moves this technique away from traditional methods and closer to ecologically sophisticated approaches to forestry or range management (Packard and Mutel, 1997). This idea does resonate with old-field theory, given that ruderal, often non-native, species are particularly well adapted to disturbed ground (Hobbs & Cramer, 2007). It should be noted that the effectiveness of this technique has been disputed (Rowe, 2010). Klaus (2013) suggests that a no-plow technique might be appropriate for the urban setting, where authorities are often interested in low-cost alternatives. It would be interesting to compare interseeding with till-and-sow methodology in a long-term experiment on an urban meadow ecosystem.

A technique for grassland restoration that is being increasingly employed in Western Europe is the transfer of seed containing plant-material, usually in combination with some form of mechanical disturbance to the existing sward (Fischer et al., 2013b; Schmiede, Otte, & Donath, 2012). This technique, in a recently published experiment, has proven to be as effective in establishing target grassland plant species as a seeding trial (Fischer et al., 2013b).

An advantage to this technique is that the plant material provides some physical protection to the propagules. On the other hand, it provides less direct control over the exact species mix to be disseminated. It is economical if "reference" sites of appropriate size and quality are available relatively near the site to be restored, and if harvesting equipment is available. A hurdle in applying this technique for meadow restoration in the Southern Ontarian context may be finding suitable reference sites. Regardless of restoration methodology, it seems that the primary impediment to grassland restoration on the modern-day landscape is dispersal limitation (Fischer et al., 2013a; Foster & Gross, 1998; Hedberg & Kotowski, 2010; Klaus, 2013). Even within a site, the speed of colonization of meadow species can be quite slow, rarely exceeding ten metres per year. For this reason, fast spread cannot be expected when introducing species into small, isolated patches on a restoration site (Hedberg & Kotowski, 2010).

While restoration ecology has generally focused on returning an ecosystem to a pristine state based on a historic ideal, some researchers propose that it may sometimes be misleading to suggest that this is possible (Hobbs et al., 2011). Increasingly, the notion that the discipline should shift its aim towards the provision of ecosystem services and resilience against future environmental change has been gaining traction (Choi, 2007; Hobbs & Harris, 2001; Hobbs et al., 2006; Jackson & Hobbs, 2009; Shackelford et al., 2013; Suding, 2011). Nowhere are these concepts more apt than in the urban context, where the potential for intervention (*sensu* Hobbs et al., 2011) is great, but the lack of historical analogs can be difficult to negotiate (Handel, Saito & Takeuchi, 2013; Standish et al., 2012). There have been very few studies on the enhancement of urban grasslands that take into account the peculiarities of their setting.

The few that have, however, suggest that there is ample scope for the restoration of native biodiversity, while recognizing that the competition from ruderal and non-native species will be more pronounced than at rural sites (Fischer et al., 2013a; Klaus, 2013). Fischer et al. (2013b) propose that a target community for urban meadows need not be predefined, but that instead the native species can be integrated into a community including non-natives as part of a novel assemblage. They emphasize that many native species can be established in heterogeneous urban soils, to the benefit of their pollinators and other wildlife as well as contributing to the conservation of regional genetic diversity. The also note that planting on extant soils, as opposed to amending, importing or stripping topsoil, may be beneficial both from a conservation perspective (in providing a diversity of niches) and a recreational perspective (in producing structural diversity).

Meadows are an early successional ecosystem: in temperate zones, they are created through the disturbance of their forested matrix. If not maintained, they are colonized by woody vegetation within a matter of decades (see Section 2.2.5). The combined forces of development, rural depopulation (particularly in Europe), the loss of megaherbivores and the disruption of fire regimes have led to a massive loss of grasslands worldwide (Navarro & Pereira, 2012; Nowacki & Abrams, 2008). In Southern Ontario, widespread clearing for agriculture and urban development has lead to the loss of much of the vast majority of the province's grasslands (Bakowsky & Riley, 1994), and the regional extinction of its largest herbivore (*Cervus elaphus*; Bellhouse & Broadfoot, 1998).

The majority of old fields in the province end up either slated for development, or return to a forested state (Milne & Bennett, 2007). Given the importance of grasslands for conservation, and the potential for their restoration in urban areas, a major consideration will be determining their optimal maintenance regime. While prescribed burning may be difficult to implement in some urban contexts (Handel et al., 2013), it may be particularly valuable to some species. Alternatively, the ideal timing and frequency of mowing (Berendse et al., 1992) or grazing for the promotion of biodiversity will need to be determined.

4.3 Conclusions

In a recent valuation study by De Groot et al. (2013), the benefits of investing in the restoration of any terrestrial ecosystem surpassed the costs. Interestingly, the highest benefit-to-cost ratio proposed in the study was for grassland ecosystems. On the low end, they estimate the value at 5:1, while in their best-case scenario, the benefit-to-cost ratio for grasslands is of 35:1. The authors add that in most cases, certain benefits are not captured by the analysis, suggesting that the figures are underestimates of the actual welfare effects of restoration. In the case of urban grasslands, many of the benefits are relatively intangible, such as engaging community members in the process, restoring people's connection to nature and reinforcing the place of people in natural systems (Shackelford et al., 2013). Furthermore, restoration offers the opportunity to educate the public on the intricacies of ecosystem function and the value of stewardship (Standish et al., 2012).

There is a major research gap when it comes to the restoration of urban grasslands. Very few studies on the topic exist, and those that do originate exclusively from Western Europe (Klaus, 2013). The studies that have been published demonstrate that urban grasslands have a high biodiversity potential (Fischer et al., 2013a; Fischer et al., 2013b), and suggest that the restored system should be considered as a novel ecosystem due to the inevitable ecological differences with historical analogs. To my knowledge, this experiment is the first in North America on the restoration of meadows in an urban context. More studies, covering the complete socioecological gradient, will contribute to developing an effective restoration protocol for the ecosystem. Longitudal studies will be especially effective in shedding light on the dynamics of urban meadows, their responses to different treatments and management regimes and their value to wildlife and human visitors.

No major treatment effects were evident in the findings of this study. This may be due to their nature, their interactions with site conditions, or simply the short duration of the experiment. The results of the literature review suggest that while chemical fertilizers are effective in providing plant-useable nitrogen in the immediate, legume species are a more economical and effective long-term solution for increasing site fertility. Biochar has potential as a soil amendment, in particular for well-drained sites with disrupted soil biota, but more studies are needed to demonstrate is utility for restoration projects. Recent publications propose that besides the most extreme cases, however, extant soils may be adequate for the restoration of diverse grassland communities in an urban context. The difference in soil conditions between study sites, and the heterogeneity within sites, was consequential on the development of their respective plant communities. Specifically, pre-treatment nutrient concentrations were influential on both species richness and productivity. These results illustrate the critical importance of initial site conditions in affecting the outcome of a restoration project.

References

Anderson, M.K. & M.J. Moratto. (1996). *Native American land use practices and ecological impacts*. Sierra Nevada Ecosystem Research Project: Final report to Congress, Vol. 2. University of California, Davis, CA.

Andrés, P. (1999). Ecological risks of using sewage sludge as a fertilizer in soil restoration: effects on the soil microarthropod populations. *Land degradation & development* 77: 67-77.

Bakowsky, W. & J.L. Riley. (1994). A survey of the prairies and savannas of Southern Ontario. Pages 7-16 *in* Proceedings of the Thirteenth North American Prairie Conference. Windsor, ON.

Blumenthal, D.M., Jordan, N.R. & M.P. Russelle. (2003). Soil Carbon addition controls weeds and facilitates prairie restoration. *Ecological Applications* 13(3): 605-615.

Beier, P. & R.F. Noss. (1998). Do habitat corridors provide connectivity? *Conservation Biology* 12 (6) 1241-1252.

Bell, A.C. and J.V. DeMarco. (1999). “Protecting, conserving and restoring biodiversity in Ontario.” Toronto: CIELAP. <http://www.cielap.org/pdf/EnvAgenda_Biodiversity.pdf> Accessed March 10, 2013.

Bellhouse, T. & J. Broadfoot. (1998). Plan for the restoration of elk in Ontario. Ontario Ministry of Natural Resources Internal Report, Ontario, Canada.

Berendse, F., Oomes, M.J.M., Altena, H.J. & W.T. Elberse. (1992). Experiments on the restoration of species-rich meadows in The Netherlands. *Biological Conservation* 62: 59-65.

Biasioli, M., Barberis, R. & F. Ajmone-Marsan. (2006). The influence of a large city on some soil properties and metals content. *Science of the Total Environment* 356 (1): 154-164.

Blythe, E.K. & D.J. Merhaut. (2007). Testing the assumption of normality for pH and electrical conductivity of substrate extract using the pour-through method. *HortScience* 42 (3): 661-669.

Bradshaw, A.D. & M.J. Chadwick. (1980). *The restoration of land: the ecology and reclamation of derelict and degraded land*. University of California Press, Berkeley, CA.

Bradshaw, A.D. (1987). Restoration: an acid test for ecology. In *Restoration ecology: a synthetic approach to ecological research.* W.R. Jordan, M.E. Gilpin, J.D. Aber., Eds.; Cambridge University Press, Cambridge, U.K. pp. 23–30.

Bradshaw, A.D. (1992). The biology of land restoration. In *Applied population biology*. Jain, S.K., Botsford, L.W., Eds. Kluwer Academic Publishers, Dordrecht, Netherlands. Chapter 2.

Bradshaw, A.D. (1996). Underlying principles of restoration. *Canadian Journal of Fisheries and Aquatic Sciences* 53 (Suppl. 1): 3-9.

Bradshaw, A.D. (1997). The importance of soil ecology in restoration science. In *Restoration Ecology and Sustainable Development.* Urbanska, K.M., Webb, N.R., Edwards, P.J., Eds.; Cambridge University Press, Cambridge, U.K. Chapter 4.

Bradshaw, A.D. (2002). The importance of soil. In *The Restoration and management of derelict land: Modern approaches*. Wong, M.H., Bradshaw, A.D., Eds. World Scientific, Hackensack, NJ.

Cabin, R.J. (2007). Science-driven restoration: A square grid on a round earth? *Restoration Ecology* 15 (1): 1-7.

Callaham, Jr., M.A., Rhoades, C.C. & L. Heneghan. (2008). A Striking Profile: Soil Ecological Knowledge in Restoration Management and Science. *Restoration Ecology* 16 (4): 604-607.

Carrington, L.P. & A. Diaz. (2011). An Investigation into the Effect of Soil and Vegetation on the Successful Creation of a Hay-Meadow on a Clay-Capped Landfill. *Restoration Ecology* 19 (1): 93-100.

Chapman, L.J. & D.F. Putnam. (1984). *The physiography of southern Ontario.* 3rd Ed. (Ontario Geological Survey. Special Volume 2). Ontario Ministry of Natural Resources, Toronto, ON.

Choi, Y.D. (2007). Restoration Ecology to the Future: A Call for New Paradigm. *Restoration Ecology* 15 (2): 351-353.

City of Kitchener. (2010). Huron Natural Area. <http://www.kitchener.ca/en/livinginkitchener/HuronNaturalArea.asp> Accessed July 15, 2011.

Clark, J.S. & P.D. Royall. (1995). Transformation of a northern hardwood forest by aboriginal (Iroquois) fire: charcoal evidence from Crawford Lake, Ontario, Canada. *The Holocene* 5 (1): 1-9.

Coastal Restoration and Protection Authority of Louisiana. (2003). Integrated Ecosystem Restoration and Hurricane Protection: Louisiana's Comprehensive Master Plan for a Sustainable Coast. Baton Rouge, LA.

Cramer, V.A. (2007). Old Fields as Complex Systems: New Concepts for Describing the Dynamics of Abandoned Farmland. In *Old Fields: Dynamics and Restoration of Abandoned Farmland*. Cramer, V.A., Hobbs, R.J., Eds.; Island Press, Washington, DC. Chapter 3.

Cramer, V.A., Hobbs, R.J. & R.J. Standish. (2008). What's new about old fields? Land abandonment and ecosystem assembly. *Trends in Ecology and Evolution* 23 (2): 104-112.

Craul, P.J. (1992). *Urban Soils in Landscape Design*. John Wiley & Sons, Inc, New York, NY.

Dale, T. & Carter, V.G. (1955). *Topsoil and Civilization*. University of Oklahoma Press, Norman, OK.

Dancer, W.S., Handley, J.F. & A.D. Bradshaw. (1977*a*). Nitrogen Accumulation in Kaolin Mining Wastes in Cornwall I – Natural Communities. *Plant and Soil* 48 (1): 153-167.

Dancer, W.S., Handley, J.F. & A.D. Bradshaw. (1977*b*). Nitrogen Accumulation in Kaolin Mining Wastes in Cornwall II – Forage Legumes. *Plant and Soil* 48 (2): 303-314.

Dancer, W.S., Handley, J.F. & A.D. Bradshaw. (1979). Nitrogen Accumulation in Kaolin Mining Wastes in Cornwall III – Nitrogenous Fertilizers. *Plant and Soil* 51 (4): 471-484.

Day, J.W., Boesch, D.F., Clairain, E.J., Kemp, C.P., Laska, S.B., Mitsch, W.J., Orth, K., Mashriqui, H., Reed, D.J., Shabman, L., Simenstad, C.A., Streever, B.J., Twilley, R.R., Watson, C.C., Wells, J.T. & D.F. Whigham. (2007). Restoration of the Mississippi Delta: Lessons from Hurricanes Katrina and Rita. *Science* 315: 1679-1684.

de Groot, R.S., Blignaut, J., van der Ploeg, S., Aronson, J., Elmqvist, T. & J. Farley. (2013). The benefits of investing in ecological restoration. *Conservation Biology* 27 (6): 1286-1293.

de Kimpe, C.R. & J.L. Morel. (2000). Urban Soil Management: A Growing Concern. *Soil Science* 165 (1): 31-40.

Dey, D.C. & R. P. Guyette. (2000). Anthropogenic fire history and red oak forests in south-central Ontario. *The Forestry Chronicle* 76 (2): 339-347.

Diemont, S.A.W. & J.F. Martin. (2009). Lacandon Maya ecosystem management: sustainable design for subsistence and environmental restoration. *Ecological Applications* 19 (1): 254-266.

Dorney, C.H. & J.R. Dorney. (1989). An unusual oak savanna in Northeastern Wisconsin: The effect of Indian-caused fire. *American Midland Naturalist* 122 (1): 103-113.

Emery, S.M. & K.L. Gross. (2007). Dominant species identity, not community evenness, regulates invasion in experimental grassland plant communities. *Ecology* 88 (4): 954-964.

Environment Canada. (2011). Canadian Climate Normals 1971-2000. <http://www.climate.weatheroffice.gc.ca/climate_normals/results_e.html?stnID=4831&lang=e&dCode=0&StationName=WATERLOO&SearchType=Contains&province=ALL&provBut=&month1=0&month2=12> Accessed July 15, 2011.

Faber-Langendoen, D., & P. F. Maycock. (1994). A vegetation analysis of tallgrass prairie in southern Ontario. Pages 17-32 *in* Proceedings of the thirteenth North American Prairie Conference. Windsor, ON.

Fischer, L.K., von der Lippe, M., Rillig, M.C. & I. Kowarik. (2013*a*). Creating novel urban grasslands by reintroducing native species in wasteland vegetation. *Biological Conservation* 159: 119-126.

Fischer, L.K., von der Lippe, M. & I. Kowarik. (2013*b*). Urban grassland restoration: Which plant traits make desired species successful colonizers? *Applied Vegetation Science* 16 (2): 272-285.

Forman, R.T.T. (2008). *Urban Regions: Ecology and Planning Beyond the City*. Cambridge University Press, New York, NY.

Foster, B.L. & K.L. Gross. (1998). Species Richness in a Successional Grassland: Effects of Nitrogen Enrichment and Plant Litter. *Ecology* 79 (8): 2593-2602.

Foster, J., Lowe, A. & S. Winkelman. (2011). The value of green infrastructure for urban climate adaptation. Washington, D.C: Center for Clean Air Policy. Retrieved December 11, 2011, from http://mi.mi.gov/documents/dnr/TheValue_347538_7.pdf

Francis, G. (2005). Overview of Concepts and Insights from Complex Systems. Working Paper, Biosphere Sustainability Project. Department of Environment and Resource Studies, University of Waterloo, Waterloo, ON.

Giardina, C.P., Litton, C.M., Thaxton, J.M., Cordell, S., Hadway, L.J. & D.R. Sandquist. (2007). Science-driven restoration: A candle in a demon-haunted world – Response to Cabin (2007). *Restoration Ecology* 15 (2): 171-176.

Gilbert-Norton, L., Wilson, R., Stevens, J.R. & K.H. Beard. (2010). A Meta-Analytic Review of Corridor Effectiveness. *Conservation Biology* 24 (3) 660-668.

Goddard, M.A., Dougill, A.J. & T.G. Benton. (2010). Scaling up from gardens: biodiversity conservation in urban environments. *Trends in Ecology and Evolution* 25 (2): 90-98.

Griffiths, C.J., Hansen, D.M., Jones, C.G., Zuël, N. & S. Harris. (2011). Resurrecting Extinct Interactions with Extant Substitutes. *Current Biology* 21 (9): 762-765.

Guo, Q. & W.L. Berry. (1998). Species richness and biomass: Dissection of the hump-shaped relationships. *Ecology* 79 (7): 2555-2559.

Handel, S.N., Saito, O. & K. Takeushi. (2013). Restoration ecology in an urbanizing world. In *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities*. Elmqvist, T., Fragkias, M., Goodness, J., Güneralp, B., Marcotullio, P.J., McDonald, R.I., Parnell, S., Schewenius, M., Sendstad, M., Seto, K.C., Wilkinson, C., Eds. Chapter 31.

Hanselwandter, K. (1997). Soil micro-organisms, mycorrhiza, and restoration ecology. In *Restoration Ecology and Sustainable Development.* Urbanska, K.M., Webb, N.R., Edwards, P.J., Eds.; Cambridge University Press, Cambridge, U.K. Chapter 5.

Harris, J.A., Hobbs, R.J., Higgs, E. & J. Aronson. (2006). Ecological restoration and global climate change. *Restoration Ecology* 14 (2): 170-176.

Harris, J.A. (2009). Soil Microbial Communities and Restoration Ecology: Facilitators or Followers? *Science* 325 (5940): 573-574.

Harris, J.A. (2010). Restoration in the City. *Ecological Restoration* 28 (1): 3.

Hedberg, P. & W. Kotowski. (2010). New nature by sowing? The current state of species introduction in grassland restoration and the road ahead. *Journal for Nature Conservation* 18 (4): 304-308.

Heneghan, L., Miller, S.P., Baer, S., Callaham, Jr., M.A., Montgomery, J., Pavao-Zuckerman, M., Rhoades, C.C. & S. Richardson. (2008). Integration Soil Ecological Knowledge into Restoration Management. *Restoration Ecology* 16 (4): 608-617.

Higgs, E. S. (1994). Expanding the scope of restoration ecology. *Restoration Ecology* 2 (3): 137–146.

Higgs, E.S. (2005). The Two-Culture Problem: Ecological Restoration and the Integration of Knowledge. *Restoration Ecology* 13 (1): 159-164.

Hilderbrand, R.H., Watts, A.C. & A.M. Randle. (2005). The Myths of Restoration Ecology. *Ecology and Society* 10 (1): 19 [online] URL: http://www.ecologyandsociety.org/vol10/iss1/art19/.

Hobbs, R.J. & D.A. Norton. (1996). Towards a Conceptual Framework for Restoration Ecology. *Restoration Ecology* 4 (2): 93-110.

Hobbs, R.J. & J.A. Harris. (2001). Restoration Ecology: Repairing the Earth's Ecosystems in the New Millenium. *Restoration Ecology*: 9 (2): 239-246.

Hobbs, R.J., Cramer, V.A., Eds. (2007). *Old fields: Dynamics and restoration of abandoned farmland.* Island Press, Washington, DC.

Hobbs, R.J., Hallett, L.M., Ehrlich, P.R. & H.A. Mooney. (2011). Intervention ecology: Applying ecological science in the twenty-first century. *BioScience* 61 (6): 442-450.

Hobbs, R.J., Higgs, E.S., Hall, C.M., Eds. (2013). *Novel ecosystems: intervening in the new ecological world order.* Wiley-Blackwell, Hoboken, N.J.

Huberty, L.E., Gross, K.L. & C.J. Miller. (1998). Effects of nitrogen addition on successional dynamics and species diversity of Michigan old fields. *Journal of Ecology* 86 (5): 704-803.

Ingram, M. (2008). Urban Ecological Restoration. *Ecological Restoration* 26 (3): 175-177.

Izquierdo, I., Caravaca, F., Alguacil, M.M., Hernández, G. & A. Roldán. (2005). Use of microbiological indicators for evaluating success in soil restoration after revegetation of a mining area under subtropical conditions. *Applied Soil Ecology* 30 (1): 3-10.

Jackson, S.T. & R.J. Hobbs. (2009). Ecological restoration in the light of ecological history. *Science* 325 (5940): 567-569.

Jeffries, R.A., Willson, K. & A.D. Bradshaw. (1981). The potential of legumes as a nitrogen source for the reclamation of derelict land. *Plant and Soil* 59 (1): 173-177.

Jordan, W. R. (2003). *The sunflower forest: ecological restoration and the new communion with nature.* University of California Press, Berkeley, CA.

Kadmon, R. & O. Allouche. (2007). Integrating the effects of area, isolation and habitat heterogeneity on species diversity: a unification of island biogeography and niche theory. *The American Naturalist* 170 (3): 443-454.

Kardol, P., Bezemer, T.M. & W.H. Van Der Putten. (2009). Soil Organism and Plant Introductions in Restoration of Species-Rich Grassland Communities. *Restoration Ecology* 17 (2): 258-269.

Klaus, V.H. (2013). Urban grassland restoration: A neglected opportunity for biodiversity conservation. *Restoration Ecology* 21 (6): 665-669.

Kourtev, P.S., Ehrenfeld, J.A. & M. Häggblom. (2002). Exotic Plant Species Alter the Microbial Community Structure and Function in the Soil. *Ecology* 83 (11): 3152-3166.

Kulmatiski, A. (2011). Changing Soils to Manage Plant Communities: Activated Carbon as a Restoration Tool in Ex-arable Fields. *Restoration Ecology* 19 (101): 102-110.

LaCroix, C.J. (2010). Urban agriculture and other green uses: Remaking the shrinking city. *The Urban Lawyer* 42 (2): 225-285.

Lehmann, J., Joseph, S., Eds. (2009). *Biochar for Environmental Management: Science and Technology*. Earthscan, Sterling, VA.

Lehmann, J., Rillig, M.C., Thies, J., Masiello, M.A., Hockaday, W.C. & D. Crowley. (2011). Biochar effects on soil biota – A review. *Soil Biology & Biochemistry* 43 (9): 1812-1836.

Li, Y. & M. Norland. (2001). The Role of Soil Fertility in Invasion of Brazilian Pepper (*Schinus Terebinthifolius*) in Everglades National Park, Florida. *Soil Science* 166 (6): 400-405.

Light, A. & E.S. Higgs. (1996). The Politics of Ecological Restoration. *Environmental Ethics* 18: 227- 245.

Mann, C.C. (2005). *1491: New revelations of the Americas before Columbus*. Vintage Books, New York, NY.

Martin, P.J. & E.O. Frind. (1998). Modeling a complex multi-aquifer system: The Waterloo moraine. *Ground Water* 36 (4): 679-690.

Maycock, P.F. & M. Guzikowa. (1984). Flora and vegetation of an old field community at Erindale, Southern Ontario. *Canadian Journal of Botany* 62: 2193-2207.

Meiners, S.J., Pickett, S.T.A. & M.L. Cadenasso. (2002). Exotic plant invasions over 40 years of old field successions: community patterns and associations. *Ecography* 25 (2): 215-223.

Miller, D.E. (2013). Hudson River Estuary Habitat Restoration Plan. New York State Department of Environmental Conservation, Hudson River Estuary Program.

Miller, J.R. & R.J. Hobbs. (2002). Conservation Where People Live and Work. *Conservation Biology* 16 (2): 330-337.

Milne, R.J. & L.P. Bennett. (2007). Biodiversity and ecological value of conservation lands in agricultural landscapes of southern Ontario, Canada. *Landscape Ecology* 22: 657-670.

Mittelbach, G.G., Steiner, C.F., Scheiner, S.M., Gross, K.L., Reynolds, H.L., Waide, R.B., Willig, M.R., Dodson, S.I. & L. Gough. (2001). What is the observed relationship between species richness and productivity? *Ecology* 82 (9): 2381-2396.

Munoz, S.E. & K. Gajewski. (2010). Distinguishing prehistoric human influence on late-Holocene forests in southern Ontario, Canada. *The Holocene* 20 (6): 967-981.

Murphy, S.D. (2005). Concurrent management of an exotic species and initial restoration efforts in forests. *Restoration Ecology* 13 (4) 584-593.

Navarro, L.M. & H.M. Pereira. (2012). Rewilding abandoned landscapes in Europe. *Ecosystems* 15 (6): 900-912.

Newman, A. (2008). Inclusive Planning of Urban Nature. *Ecological Restoration* 26 (3): 229-234.

Newmark, W.D. (1987). A land-bridge island perspective on mammalian extinctions in western North American parks. *Nature* 325 (6103): 430-432.

Noss, R.F. (1988). Corridors in real landscapes: A reply to Simberloff and Cox. *Conservation Biology* 1 (2): 159-164.

Nowacki, G.J. & M.D. Abrams. (2008). The demise of fire and “mesophication” of forests in the Eastern United States. *BioScience* 58 (2): 123-138.

Packard, S. & C.F. Mutel. (1997). *The tallgrass restoration handbook: For prairies, savannas and woodlands*. Island Press, Washington, DC.

Pavao-Zuckerman, M.A. (2008). The Nature of Urban Soils and Their Role in Ecological Restoration in Cities. *Restoration Ecology* 16 (4): 642-649.

Pickett, S.T.A & White, P.S. (1985). *The Ecology of Natural Disturbance and Patch Dynamics.* Academic Press, San Diego, CA.

Pouyat, R.V., McDonnell, M.J. & S.T.A. Pickett. (1995). Soil characteristics of oak stands along an urban-rural land-use gradient. *Journal of Environmental Quality* 24 (4): 516-526.

Proença, V., Honrado, J. & H.M. Pereira. (2012). From abandoned farmland to self-sustaining forests: challenges and solutions. *Ecosystems* 15 (6): 881-882.

Pywell, R.F., Webb, N.R. & P.D. Putwain. (1995). A comparison of techniques for restoring heathland on abandoned farmland. *Journal of Applied Ecology* 32: 400-411.

rare. (2011). A *rare* idea. <http://www.raresites.org/cms/en/AboutUs/Overveiwofwhoweare.aspx?menuid=154> Accessed June 12, 2011.

Rey Benayas, J. M. & J. M. Bullock. (2012). Restoration of Biodiversity and Ecosystem Services on Agricultural Land. *Ecosystems* 15 (6): 883-899.

Ripple, W.J., Rooney, T.P. & R.L. Beschta. (2010). Large predators, deer, and trophic cascades in boreal and temperate ecosystems. In *Trophic cascades: Predators, prey and the changing dynamics of nature*. Terborgh, J., Estes, J.A., Eds. Island Press, Washington, D.C. Chapter 9.

Rosenzweig, M. L. (1995). *Species Diversity in Space and Time.* Cambridge University Press, New York, NY.

Rowe, H.I. (2010). Tricks of the trade: Techniques and opinions from 38 experts in tallgrass prairie restoration. *Restoration Ecology* 18 (S2): 253-262.

Rudd, H., Vala, H. & V. Schaefer. (2002). Importance of Backyard Habitat in a Comprehensive Biodiversity Conservation Strategy: A Connectivity Analysis of Urban Green Spaces. *Restoration Ecology* 10 (2): 368-375.

Russell, E.W.B. (1983). Indian-set fires in the forests of the Northeastern United States. *Ecology* 64 (1): 78-88.

Schaefer, V. (2009). Alien Invasions, Ecological Restoration in Cities and the Loss of Ecological Memory. *Restoration Ecology* 17 (2): 171-176.

Schmiede, R. Otte, A. & T.W. Donath. (2010). Enhancing plant biodiversity in species-poor grassland through plant-material transfer – the impact of sward disturbance. *Applied Vegetation Science* 15 (2): 290-298.

Shackelford, N., Hobbs, R.J., Burgar, J.M., Erickson, T.E., Fontaine, J.B., Laliberté, E., Ramalho, C.E., Perring, M.P. & R.J. Standish. (2013). Primed for change: Developing ecological restoration for the 21st century. *Restoration Ecology* 21(3): 297-304.

Smith, R.S., Shiel, R.S., Bardgett, R.D., Millward, D., Corkhill, P., Rolph, G., Hobbs, P.J. & S. Peacock. (2003). Soil microbial community, fertility, vegetation and diversity as targets in the restoration of a meadow grassland. *Journal of Applied Ecology* 40: 51-64.

Smith, R.S., Shiel, R.S., Bardgett, R.D., Millward, D., Corkhill, P., Evans, P., Quirk, H., Hobbs, P.J. & S.T. Kometa. (2008). Long-term change in vegetation and soil microbial communities during the phased restoration of traditional meadow grassland. *Journal of Applied Ecology* 45: 270-279.

Sohi, S., Lopez-Capel, E., Krull, E. & R. Bol. (2009). Biochar's roles in soil and climate change: A review of research needs. CSIRO Land and Water Science Report 05/09.

Soulé, M.E. & J. Terborgh. (1999). *Continental Conservation: Scientific Foundations of Regional Reserve Networks*. Island Press, Washington, DC.

Standish, R.J., Hobbs, R.J. & J.R. Miller. (2012). Improving city life: options for ecological restoration in urban landscapes and how these might influence interactions between people and nature. *Landscape Ecology* 28 (6): 1213-1221.

Stevens, C.J., Dise, N.B., Mountford, J.O. & D.J. Gowing. (2004). Impact of nitrogen deposition on the species richness of grasslands. *Science* 303 (5665): 1876-1879.

Suding, K.N., Gross, K.L. & G.R. Houseman. (2004). Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology and Evolution* 19 (1): 46-53.

Suding, K.N. & R.J. Hobbs. (2009). Threshold models in restoration and conservation: a developing framework. *Trends in Ecology and Evolution* 24 (5): 271-279.

Suding, K.N. (2011). Towards an era of restoration in ecology: Successes, failures and opportunities ahead. *Annual Review of Ecology, Evolution and Systematics* 42 (1): 465-487.

Szeicz, J.M. & G.M. MacDonald. (1991). Postglacial vegetation history of oak savanna in southern Ontario. *Canadian Journal of Botany* 69 (9): 1507-1519.

Tognetti, P.M., Chaneton, E.J., Omacini, M., Trebino, H.J. & R.J.C. León. (2010). Exotic vs. native plant dominance over 20 years of old-field succession on set-aside farmland in Argentina. *Biological Conservation* 143 (11): 2494-2503.

Thompson, L.M. & F.R. Troeh. (1973). *Soils and Soil Fertility*. 3rd Ed. McGraw-Hill Book Company, New York, NY.

Truett, J.C., Phillips, M., Kunkel, K. & R. Miller. (2001). Managing bison to restore biodiversity. *Great Plains Research* 11 (1): 123-144.

United States Army Corps of Engineers. (2009). Hudson-Raritan Estuary: Comprehensive Restoration Plan. New York, NY.

USDA-NRCS (1997). *National Grazing Lands Handbook*. USDA-NRCS, Washington, DC.

Vermeer, J.G. & F. Berendse. (1983). The relationship between nutrient availability, shoot biomass and species richness in grassland and wetland communities. *Vegetatio* 53 (2): 121-126.

Vécrin, M.P. & S. Muller. (2003). Top-soil translocation as a technique in the re-creation of species-rich meadows. *Applied Vegetation Science* 6: 271-278.

Waide, R.B., Willig, M.R., Steiner, C.F., Mittelbach, G., Gough, L., Dodson, S.I., Juday, G.P. & R. Parmenter. (1999). The relationship between productivity and species richness. *Annual Review of Ecology and Systematics* 30: 257-300.

Warnock, D.D., Lehmann, J., Kuyper, T.W. & M.C. Rillig. (2007). Mycorrhizal responses to biochar in soil – concepts and mechanisms. *Plant and Soil* 300 (1): 9-20.

White, R.E. (2006). *Principles and Practice of Soil Science: the Soil as a Natural Resource*. 4th Ed. Blackwell Publishing, Malden, MA.

Young, T.P. (2000). Restoration ecology and conservation biology. *Biological Conservation* (92): 73-83.

Young, T.P., Chase, J.M. & R.T. Huddleston. (2001). Community succession and assembly: comparing, contrasting and combining paradigms in the context of ecological restoration. *Ecological Restoration* 19 (1): 5-18.

Young, T.P., Petersen, D.A. & J.J. Clary. (2005). The ecology of restoration: historical links, emerging issues and unexplored realms. *Ecology Letters* 8: 662-6