

New Urban Ecological Restoration Techniques: Testing the Short-Term Effects of  
Adding Deciduous Leaf Litter and Plant Residue Compost on Topsoil Quality and  
Native Herbaceous Plant Establishment

by

Jie Si Ma

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## **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.

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## Abstract

Urban expansion has led to native habitat destruction. Meanwhile, cities produce large quantities of plant residue wastes. To explore the potential to use plant residues to restore native habitats in cities, this thesis compared the short-term effects of deciduous leaf litter and plant residue compost on topsoil quality and seeded native herbaceous plant establishment. To determine if natural plant colonisation alone can restore native plants, the thesis compared the establishment and ecological characteristic and diversity of naturally colonised plant species on the barren surface of modified and unmodified soil.

Following three months of repeated measurements after amendment, compost significantly increased soil moisture, organic matter, extractable NPK, and significantly decreased soil pH and bulk density. The decrease in soil pH due to incorporating compost with higher pH than the receiving soil showed that compost may not restore soil pH in a predictable manner. The average number (47.8) and median shoot lengths (40.0 cm) of seeded native plants on compost-amended soil were significantly higher than those on control ( $\#_{\text{avg}} = 28.3$ , shoot length $_{\text{median}} = 23.3$  cm) three months after germination. Compost may therefore be used for purposes such as quick establishment of dense tall native plant cover.

Conversely, leaf litter did not significantly change the abovementioned soil properties in the short term. However, the average number of seeded native plants on leaf litter-amended soil three months after germination (41.5) was significantly higher than control and insignificantly different from that on compost-amended soil. At that time, seeded native plants on leaf litter-soil exhibited stunt growth (shoot length $_{\text{median}} = 14.3$  cm) relative to other treatments. Leaf litter might then be used to establishing slow-growth native lawns for urban native landscaping practices. But this technique requires further refinement.

Above barren soils across experimental treatments, exotic weeds consisted mostly of the naturally colonised plants. This means natural plant colonisation may not effectively restore native plants. There was no significant difference in the colonising plant average species richness ( $R_{\text{leaf}} = 6.0$ ,  $R_{\text{comp}} = 6.0$ ,  $R_{\text{ctrl}} = 5.3$ ) and diversity ( $H'_{\text{leaf}} = 1.445$ ,  $H'_{\text{comp}} = 1.635$ ,  $H'_{\text{ctrl}} = 1.355$ ) across treatments. The steepness of the colonising species' rank-abundance curves were similar between treatments. Thus, natural plant colonisation on amended or un-amended soil could not lead to the establishment of particular plant species. Due to soil nutrient-enrichment, Canada thistles (*Cirsium arvense*) occupied a greater proportion of colonised species on compost-treated soil than the most dominant colonised species on other types of soil. Thus, compost amendment of soil may not restore plant communities with high species diversity.

Moreover, the average number of the colonising weeds above compost-amended soil (19.5) was significantly higher than control (14.5), while that of weeds above leaf litter-amended soil (11.5) was

significantly lower than control three months after site preparation. At that time, weeds on leaf litter-amended soil (shoot length<sub>median</sub> = 8.4 cm) were significantly shorter than those on compost-amended soil (shoot length<sub>median</sub> = 24.8 cm) and control (shoot length<sub>median</sub> = 9.7 cm). This means leaf litter could impede exotic weeds while compost had the reverse effect. Compost should only be used to establish native plants if exotic weeds are removed or when the site soil seed bank and adjacent land contain few exotics.

While this thesis documented the different short-term effects of plant residue compost and deciduous leaf litter on soil and plants, long-term investigations may find potentially different applications for the different types of plant residues in native plant restoration projects with different purposes. Reusing plant residues differentially could mean that composting may not always be necessary in plant waste management and urban ecological restoration may help to reduce waste output from cities.

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## **Dedications**

To all!

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## 1.0 Introduction

### 1.1 Ecological Restoration of Native Plants in the Urban Environment

Humans have altered 75% of the Earth's landscape not covered by ice (Allison 2012: p.49). Urban areas such as the city clusters around Toronto in southern Ontario (Canada) can occupy large tracts of once natural habitat and are regional biodiversity hotspots (Cristine and Kerr 2011). To rehabilitate damaged ecosystems as a result of rapid urban expansion (Irwin and Bockstael 2007), ecological restoration should take place in cities (Kocs 2013: pp.34-37, Newman 2011, Ingram 2008). While large tracts of farmland outside cities may continue to offer sites for restoration work (Wade *et al.* 2008), the restored habitats may be distant from urban dwellers (Warber *et al.* 2015). With more than half of the world's population now living in cities (World Bank 2015), little or no sight and knowledge about ecological restoration among the large body of urban residents may lead to low ecological consciousness and little support for habitat conservation and restoration (Standish *et al.* 2013).

As the basis of the terrestrial ecosystems, the world's plant communities are experiencing rapid declines (e.g. Matthews *et al.* 2000, White *et al.* 2000) and their restoration should deserve attention. Although recently there are opinions on using exotic plants for ecological restoration (Hobbs *et al.* 2014, Allison 2012: pp.12, 127, Antonio and Meyerson 2002), landscapes revegetated with exotic plants of immediate human interest may not guarantee the long-term provision of critical ecosystem functions such as the provision of habitat for other native species (Baker and Quinn-Davidson 2011). From a precautionary perspective, the restoration of native plants may help to improve the ecological integrity of the urban ecosystem (Allison 2012: p.13). The restoration of native plants in urban environment may also increase city dwellers' knowledge and appreciation for local species. This increased public knowledge and appreciation for local species can gear greater support for the protection of native habitats and local natural heritage (Standish *et al.* 2013).

The selection of the type of native plants may affect the success of an ecological restoration initiative in the urban environment. In southern Ontario, the restoration of dense native forests, or the restoration of an ecosystem state before European settlement preferred by many North American restoration ecologists (Allison 2012: p.12), may not always be compatible with human-intensive urban land use there. Returning to the ecological state of a randomly assigned past time point begs the question of the purpose of restoration. Technically speaking, returning precisely to a particular state can be difficult to achieve all at once (Hilderbrand *et al.* 2005), meaning a long-term step-wise approach in restoration may be needed (e.g. Lammerant *et al.* 2013). As part of many standard restoration practices, the selection of a historical "pristine" reference site might be a challenge in areas that already experienced

high degree of human perturbation (Gobster 2010, Ruiz-Jaen and Aide 2005, Bright *et al.* 2002). With calls for connecting ecological restoration with human development (Instone 2015, Newman and Dale 2013, Standish *et al.* 2013, Yu *et al.* 2012, Newman 2011, Ingram 2008), restoration to native herbaceous groundcover that reflects the potential landscaping preference for lawns among urban dwellers (Bliss and Fischer 2011, Kettenring and Adams 2011) might be a more suitable choice in cities.

Large tracts of rarely used conventional monocultural turfs in urban parks and green spaces require constant human maintenance (Robbins *et al.* 2001) and they can provide very limited ecosystem and social services (Ignatieva and Ahrné 2013, Robinson and Lundholm 2012). However, these lawns can directly provide the space that is needed for restoring native plant cover. Depending on the shape and distance between patches of urban green spaces, restoration work may create intact and interconnected native habitats (Yu *et al.* 2012). Combined with sound landscape architectural designs, the deployment of native herbaceous plants in green space can be aesthetically appealing in the urban environment (e.g. Loder 2014). Helping to increase the ecological status of cities, to improve the aesthetical appeals of urban landscape, and to connect urban residents to native habitats, urban parks and green spaces should be viewed as opportunities to restore native herbs and wildflowers (Klaus 2013, Meurk and Hall 2006).

Due to chronic landscape alteration and exotic species presence, deliberate human efforts may be needed to accelerate the establishment of native plant cover in the urban environment (Zahawi *et al.* 2015, Zahawi *et al.* 2014, Hilderbrand *et al.* 2005). To obtain visible restoration outcome, sowing native plant seeds can initiate the restoration of native ground cover (e.g. Gong *et al.* 2013). The demonstration of initial success through human interference can win public support for ecological restoration (Klaus 2013, Bright *et al.* 2002). The establishment of desired species sets the basis for site rehabilitation as well as adaptive management (Zedler *et al.* 2012). But human involvement should not mean the permanent need to deliberately maintain restored ecosystems that are not self-sustainable (Prach and del Moral 2015, Hilderbrand *et al.* 2005). More research on low effort but effective methods to restore native plants in the urban environment is needed (e.g. Fischer *et al.* 2013). It might be interesting to test the assumption whether adjustment of physical and chemical parameters of the biophysical environment (Hilderbrand *et al.* 2005), such as soil conditions, alone can lead to successful establishment of native plants. However, to avoid disappointing results and waste of large amount of restoration efforts, small-scale field experiments and review of past restoration experiences are necessary to minimise the uncertainties associated with proposed restoration techniques before their large-scale applications (Herringshaw *et al.* 2010).

## 1.2 Native Plant Restoration through Amendment of Soil Conditions

The restoration of native herbaceous plants in cities needs to consider urban soil quality. Intense human activities and altered atmospheric composition and local climate (Pathirana *et al.* 2014, Aikawa *et al.* 2006, Ahmad-Shah *et al.* 1993, Böhm 1979) can impact the physical and chemical properties of topsoil (Huang *et al.* 2012, Zhang *et al.* 2011, Pavao-Zuckerman 2008), the medium upon which herbaceous plants are established. Construction activities disrupt the original soil horizon and can lead to soil compaction (Jim 1998). Urban soil may have high bulk density, low water conductivity, and variable nutrient contents (Zhao *et al.* 2007, Gregory *et al.* 2006). Exposure of topsoil to air and storm water can accelerate the loss of soil organic matter (SOM) (Six *et al.* 1998, Ruppenthal *et al.* 1997). Soil moisture content may be reduced due to exposure to warmth and dryness (de Carvalho *et al.* 2016) as result of urban heat island effects. Overland flow of storm water and snowmelt can lead to soil erosion and depletion of soil nutrients (An *et al.* 2013, Beek *et al.* 2009). Soil pH may also be altered by factors such as exposure to acidified precipitations (Singh and Agrawal 2008). The altered soil pH and organic matter content can change the content and plant-bioavailability of soil nutrients such as nitrate, phosphate, and potassium (Pavao-Zuckerman 2008, Singh and Arawal 2008).

Allison (2012: p.148) noted that soil restoration for native plant restoration should continue to deserve attention. While there are several avenues for research available, a promising one is the use of abundant processed organic waste in the form of compost. Composts can be made from landscaping plant residues, kitchen food residues, manure, or municipal sewage treatment plant biosolids. In composting, the source organic waste is separately collected or isolated from other wastes, followed by an aerobic two-step conversion into a dark-brown loam-like material known as mature compost. With resemblance to decomposed soil organic matter, mature compost consists largely of humic substances – it should be moist, plant nutrient-rich, pathogen free, and odourless (Environment Canada 2013: p.3-1 – 3-15). The presence of organic matter in soil has been demonstrated to increase soil aggregation, reduce soil compaction, prevent soil erosion, and retain soil nutrients and moisture (Kononova 1961: pp.165-171). The addition of mature compost to soil as a soil amendment agent has been shown to achieve this positive effect on soil quality to aid plant growth (Larney and Angers 2012). Curtis and Claassen (2009) found higher nitrogen and carbon contents as well as improved water holding capacity and reduced bulk density in soil amended with plant residues compost from backyards in California. There was also higher aboveground perennial grass biomass following compost addition. Working with compost made from rapeseed (*Brassica napus* L.) and red clover (*Trifolium pratense* L.) residues, Tejada *et al.* (2009) found an 87% increase in plant biomass attributed to higher soil nutrient levels four years following compost amendment.

The use of compost for soil improvement can help to reduce the negative effects of large amount of organic wastes. Each North American and European produced approximately 0.3 to 0.7 tonnes of waste per year (Themelis 2012); globally, approximately 1 billion tonnes of garbage are sent to landfills each year. Of the total amount of waste produced alone from the residential sector, more than 40% can consist of organic wastes from kitchens and landscaping-derived plant residue (Environment Canada 2013: p.1-3, Themelis 2012). Among the organic wastes, plant residues can occupy as much as half of the total amount annually (German Federal Environment Agency 2012: p.11), demonstrating its importance in the overall composition of organic wastes and total household waste. The large amount of plant residues can exacerbate demand for landfill capacity – each year, 23 million m<sup>2</sup> of land is converted to landfill in the United States. When mixed and buried with mainstream garbage in landfills, organic matter produces the potent greenhouse gas methane under anoxic conditions; in 2006, US landfills alone generated 11.8 million tonnes of methane. Moreover, decomposing organic wastes contribute to the formation of toxic landfill leachate and offensive odour (Themelis 2012).

Producing plant residue compost through decomposition, the process of composting also has its costs. Depending on the technique, open-field windrow composting can take as long as 10 months (e.g. recorded by Gajalakshmi *et al.* 2005 and Suzuki *et al.* 2004). Also, energy is required to maintain the appropriate temperature ranges for microbial activation, disinfection, and soil humic organic matter formation during composting (Canadian Council of Ministers of Environment 2005: p.15) and land is required for compost production and storage. There can also be a considerable cost associated with other aspects of compost production such as quality control. To reflect the cost of production, the estimated purchasing price for compost is at least \$20 to \$30 per tonne (Compost Council of Canada 2010). Depending on public familiarity and attitude toward composting (McKenzie-Mohr 2000), the establishment of composting facility and the application of compost can be hindered. For instances, low public participation in source separation can dramatically reduce raw material supply for composting; odour control has been an important issue with site selection for compost production (Bidlingmaier 1996); and lack of stringent reinforcement of quality control measures can affect the sales and use of compost (Golueke and Diaz 1996). As result of high production cost and potential public misunderstanding, compost application can be limited on actual properties.

Are there other ways to join the nutrient cycling between plant residues and the soil besides composting? What is the prospect of directly using plant materials taken from safe sources free of contamination and plant disease? It seems there is an opportunity to explore if addition of unprocessed plant residues to soil can also deliver acceptable soil and herbaceous plant restoration results. The addition of plant material such as forest leaves to soil surface is a natural process following leaf fall and

branch snaps. Afterward, microorganisms decompose the fresh organic matter. This conversion process is in fact an important input of humus and nutrients to the soil (Rinkes *et al.* 2014, Berg and McClaugherty 2008: p.6). Because of the similarity to natural soil decomposition process, public acceptability for soil improvement with direct application of plant material may be higher than for compost usage.

Research on effects of plant material on soil properties and plant dynamics has been plentiful (Perry *et al.* 2010). The addition of high C:N ratio sawdust and wood residues can restore native herbaceous species (Averett *et al.* 2004). Working with wood waste from the forestry sector in British Columbia (Canada), Venner *et al.* (2011) hypothesised that incorporation of wood chips to soil may improve plant seed germination. Regarding the importance of tree leaf litter to soil nutrient cycling, Dhanya *et al.* (2013) demonstrated the returning of N, P, and K to soil by tree litter in tropical agroforestry systems. Altering the soil quality for preferential establishment of desired plants, Kowalewski *et al.* (2009) found that laying oak and maple leaves on top of soil can deter weed establishment.

Unprocessed and processed plant residues may have different effects on soil properties and native plant number and growth. Often, deciduous tree leaf litter from previous autumn season may be the most common type of plant residue wastes from temperate urban environment, raising the question if they can be reused directly to enhance urban soil quality to aid native plant establishment. The effects of this unprocessed plant residue can be compared to those of plant residue compost produced from landscaping wastes, and research is needed to systematically evaluate this.

### **1.3 Thesis Research Objectives**

Initiating urban native habitat restoration with the reuse and recycling of plant residues, this thesis examined the prospect of using urban landscaping plant residues for native habitat restoration in a Canadian city. A short-term field experiment was set up to investigate if amendment of urban park soil with deciduous leaf litter and plant residue compost each can give rise to different soil physical and chemical properties, which in turn, help to establish native herbaceous plants differently to suit different social and ecological needs from restoration. More specifically, the thesis investigated

- if topsoil incorporated with deciduous leaf litter and plant residue compost would have significantly different levels of soil moisture, bulk density, soil organic matter (SOM), pH, and plant-available nitrogen, phosphorus, and potassium from the other treatment and from the un-amended soil (control), and



- if the number and shoot lengths of deliberately seeded native plants would be significantly different on topsoil amended with the leaf litter, the compost, or none in the first season of growth.

As a potential restoration method with perceivably less human involvement and cost, the thesis also examined the possibility of establishing native herbaceous plants through the colonisation of amended and un-amended barren soil surfaces by wild plants. To answer this question, the thesis investigated

- if the number and shoot lengths of naturally colonised plants would be significantly different on topsoil amended each by deciduous leaf litter, plant residue compost, or none in the first growth season,
- if the naturally colonised plants would consist largely of native plants, and
- if each treatment or no treatment of the soil would lead to significantly different number of colonised plant species, distribution of the colonised species' relative abundances, and overall Shannon-Wiener diversity index value of the colonised species from other types of treatment to favour the establishment of particular plant species over others.

The restored native herbaceous plant cover may offer habitat for other native species such as monarch butterflies (Farhat *et al.* 2014). Also, the restored areas may provide aesthetical appeal in the parkland while continuing to provide unobstructed view of other landscaping features to match with possible preference of urban park users (Figure 1).

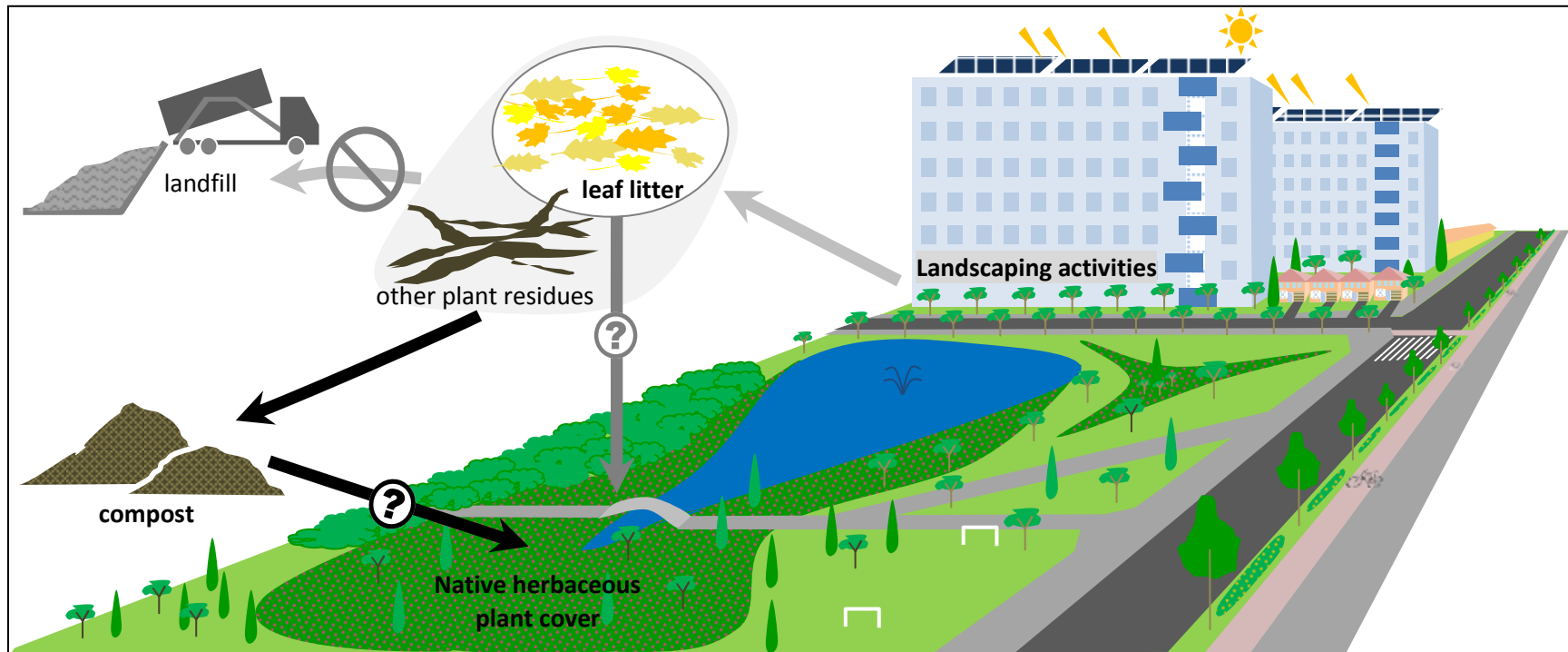


Figure 1. Pictorial depiction of the goal of the thesis to investigate the short-term effects (indicated as ⊗ on diagram) of two types of plant residues, namely deciduous tree leaf litter and municipal plant residue compost, in changing soil properties to initiate the restoration of native herbaceous ground cover in the urban park setting to make use of plant residues to deliver desirable native plant restoration outcomes.

In an attempt to establish native plant cover with plant residues, this thesis aimed to foster urban native habitat restoration through the reduction of net plant waste output. Before deciding for large-scale implementation of a restoration technique, this thesis aimed to be a realistic small-scale field experiment (Herringshaw *et al.* 2010) to find out the type of plant communities that would immediately follow the application of particular soil amendment agents. By testing out the short-term effects of different types of plant residues on urban top soils and native plant establishment, better use of particular types of plant residues, such as unprocessed deciduous tree leaves and compost from plant residues, for particular restoration goals could be possible. This knowledge could have implications for future diversion and reuse of different plant residues. Moreover, contrasting the efficacies between unprocessed and processed plant residues in bringing out desired restoration outcomes, results from this thesis experiment could help to answer the question whether composting of plant residue is always necessary for their reuse and recycling in organic waste management (German Federal Environment Agency 2012: p.7).

## **2.0 Methods**

The experimentation with urban park native herbaceous plant establishment involved finding an open field in an urban park, the collection of urban deciduous tree leaf litters and horticultural composts as the soil amendment agents, the tillage of the experimental site soil for seeding native herbaceous plants and wild plant colonisation, and the laboratory analyses and plant surveys to find out potential differences in soil and plant responses.

### **2.1 Experimental Site and Materials**

#### **2.1.1 Experimental Site**

The experiment was conducted on a slightly elevated open field in the Columbia Lake Park in City of Waterloo in southwestern Ontario (Canada) (43°28'24.01" N, 80°33'31.05"W; 336 m abs.; Figure 2). The geographical region of the experimental site has a temperate continental humid climate. The region has a winter mean daily temperature and total precipitation of -4.07 C and 252.3 mm from the end of November to the beginning of April, respectively; a spring mean daily temperature of 9.35 C and total precipitation of 156.8 mm in April and May; a summer mean daily temperature of 20.0 C and total precipitation of 352.7 mm from June to the end of September; and an autumn mean daily temperature of 5.35 C and total precipitation of 154.5 mm in October and November (Environment Canada 2015).

The topsoil of the experiment site consisted of fine sandy loam soil with good drainage (Presant and Wicklund 1971: p.39) and appeared brown in colour. Historically, the experimental site was part of the eastern North American mixed-wood plain ecozone. After European settlement of southern Ontario, the location became part of a farmland. With post-WWII suburban expansion, the location gradually became part of a park covered with wild plants and lawn. In preparation for the current field experiment, the land was tractor-tilled to a depth of 10 cm to remove the surface vegetation by park maintenance staff (University of Waterloo) in May 2016.



Figure 2. The location of the experimental site in Columbia Lake Park in City of Waterloo (Ontario), Canada. Satellite images © Google Earth 2016.

**2.1.2 Soil Amendment Agents**

*Deciduous Leaf Litter*

Overwinter deciduous tree leaves were collected from the floor of an urban forest near an industrial park in Cambridge (Ontario), Canada in late April 2015. The leaves on the top layer of the forest floor litter pile were dry and light brown in colour. These leaves had clear shapes and venation pattern allowing the identification of original tree species. The leaves beneath the top layer showed signs of degradation and were moist and had dark colour. The majority of the leaves consisted of oak tree leaves – bur oak (*Quercus macrocarpa*) and black oak (*Quercus velutina*) – and the rest were from sugar maple (*Acer saccharum*) and American beech (*Fagus americana*).

### *Plant Residue Compost*

Horticultural plant residue compost produced from an outdoor open-windrow process was used. The compost production process took 10 months. The raw materials for composting had a C:N ratio between 25:1 and 35:1 and were grass residues, weeds, brush, plant trimmings, wood chips, Christmas trees, florist wastes, leaves, some contents of fruit and vegetable wastes, and biodegradable containers of the plant residues. Pasteurization with minimum 5 turns at minimum temperature of 55 C for at least 15 consecutive days was carried out to eliminate pathogens and weed seeds. Then, the raw material underwent composting near 45 C before proceeding to curing at a lower temperature with monthly turning of the windrow. During composting, the moisture and the oxygen content of the windrow was 45-55% and more than 10%, respectively. The mature stable compost had a C:N ratio of less than 22:1 (Region of Waterloo 2014). The compost appeared dark-brown, was moist, odourless, loose in texture, and contained some small pieces of un-decomposed woody matter.

### *Amendment Agent Characterisation*

Compared to the topsoil (depth 0-15 cm) at the experimental site, both deciduous leaf litter and plant residue compost had significantly higher moisture and organic matter contents as well as lower bulk densities. The moisture and organic matter contents of the leaf litter were even higher than those of the compost, and the bulk density of the leaf litter was even lower than that of compost. While the leaf litter had a weak acidic pH, the compost had a neutral pH that was significantly higher than the experimental site soil. Both amendment agents had significantly higher extractable P and K than the experimental site soil; those of the compost were much higher than those of leaf litter. While compost contained significantly higher extractable N than the experimental site soil, that of the leaf litter was below method detection limit (Table 1). The comparison of differences between the soil amendment agents and the experimental site soil was done by subjecting 5 randomly picked sample of equal weight of each type of material to physical and chemical analyses (Appendix I Part I). See Sections 2.2.3 and 2.3 for analytical and statistic analysis procedures, respectively.

Table 1. The physical and chemical characteristics of the soil amendment agents (deciduous leaf litter and plant residue compost) comparing to those of the experimental site soil ( $n = 5$ ).

<b>Material Characterisation Parameter</b>	<b>Experimental Material</b>		
	<b>Leaf litter</b>	<b>Compost</b>	<b>Experimental Site Soil</b>
Moisture Concent (% ww)	51 ± 5 a*	45 ± 4 a	13 ± 1 b
Bulk Density (g/cm <sup>3</sup> dw)	0.02 ± 0.02 a	0.31 ± 0.02 b	0.92 ± 0.06 c
Organic Matter Content (% dw)	92 ± 1 a	28 ± 11 b	3 ± 0.1 c
pH	5.03 ± 0.13 a	7.77 ± 0.04 b	7.20 ± 0.11 c
Extractable N (kg NO <sub>3</sub> <sup>-</sup> /ha air dw)	<0.10 (below)	234 ± 59 b	6.5 ± 4.0 c

	detection range) a		
Extractable P (kg PO <sub>4</sub> <sup>3-</sup> /ha air dw)	134 ± 28 a	612 ± 26 b	84 ± 31 c
Extractable K (kg K <sup>+</sup> /ha air dw)	343 ± 131 a	3490 ± 458 b	63 ± 13 c

\* Following one-way ANOVA and Bonferroni post-hoc analyses, statistically significant differences in specific physical or chemical property between the experimental materials are indicated with different letters after the mean ± s.d.

### 2.1.3 Seeds of Native Herbs and Forbs for Ecological Restoration

A commercial mixture of early successional dry prairie meadow grass and wildflower seeds bought from Ontario Seed Company (2015) was used in the restoration experiment. The herbaceous plant species in the mixture are native to southern Ontario, mostly perennial, and are designed to regenerate surface vegetation on disturbed soil, open fields, and forest edges. Eighty percent of the seed mass consisted of grass species: Canada wild rye (*Elymus canadensis*; 25%), switchgrass (*Panicum virgatum*; 25%), followed by big bluestem (*Andropogon gerardii*; 15%) and Virginia wild rye (*Elymus virginicus*; 15%). The wildflower species contained were: black eyed Susan (*Rudbeckia hirta*; 6%), foxglove/beardtongue (*Penstemon digitalis*; 5%), New England aster (*Aster novae-angliae*; 5%), arrow leaved aster (*Aster saggitifolius*; 2%), and wild bergamot (*Monarda fistulosum*; 2%).

## 2.2 Field and Laboratory Procedures

### 2.2.1 Experimental Site Preparation

A random factorial design was used to test the effects of different soil amendment agents. In early May 2015, a total of 12 experimental plots of the dimensions 1 m × 1 m, each surrounded by 1.4 m-wide bare-soil strips, were randomly assigned to the compost and leaf litter treatments and the control (i.e.  $n = 4$ ). Within each experimental plot receiving leaf litter or compost amendment, soil from the ground surface to a depth of 30 cm were dug out, followed by manual soil loosening and mixing with soil amendment agent in a ratio of 1:1 (v/v) before backfilling. The soil amendment incorporation rate equated to 5.7 kg/m<sup>2</sup> (or 17.3 kg/m<sup>3</sup>) for leaf litter and 98.6 kg/m<sup>2</sup> (or 299 kg/m<sup>3</sup>) for compost. The soil from each control plot was dug out to a depth of 30 cm from the ground surface and was loosened by hand and subsequently backfilled. Then, each experimental plot was divided into 9 subplots of equal dimensions (0.33 m × 0.33 m). Approximately 1730 seeds from the commercial restoration seed mixture were sowed in mid-May 2015 by handpicking into the soil of the centre subplot within each experimental plot at a depth of 5 cm. The other subplots within each experimental plot were left for natural plant colonisation and soil sampling (Figure 3). Before seed sowing, the germination potential of the seeds was enhanced by two weeks of

cold stratification in a freezer. After seed sowing, the experimental plots were watered three times a week for the first two weeks.

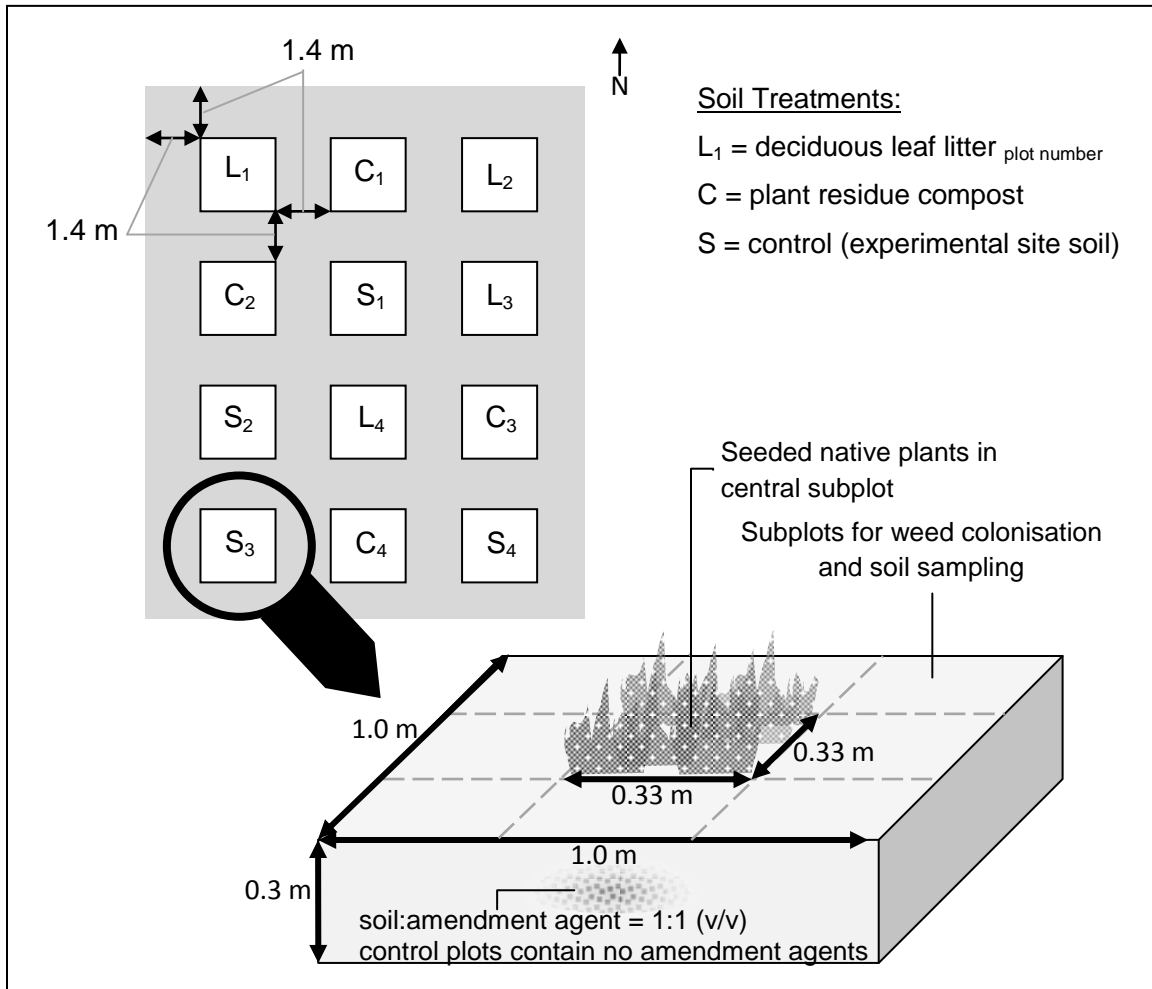


Figure 3. Randomised factorial treatment of experimental site soil with 50% (v/v) of leaf litter, plant residue compost, or none; and the depiction of experimental plot dimensions and structure.

### 2.2.2 Soil Sampling and Preservation

In early June, July, and August 2015, one topsoil (0-15 cm depth) sample from each of the 12 experimental plots was collected by extracting one subsample from one randomly determined subplot and another from another random subplot, exclusive of the centre subplot, within each experimental plot with soil bulk density rings. To prevent unwanted biogeochemical transformations and soil physical integrity alteration, all field samples were stored in tightly sealed Ziplock™ bags and kept frozen before processing. Some soil from each subsample was air-dried at room temperature for soil chemical analyses. All dry samples were frozen or stored in desiccators before processing.



### 2.2.3 Determining Physical and Chemical Properties of Soil Samples and Amendment Agents

#### *Moisture Content, Bulk Density, and Organic Matter Content*

The moisture content, bulk density, and organic matter content of experimental soil before and after amendment application and those of the plant residue compost and the deciduous leaf litter were determined using a serially coupled method. Undisturbed analytical samples within bulk density rings of given volumes ( $v$ ) were weighted to give the total weight of sample ( $w_t$ ), followed by 24-hr oven drying at 110 °C to yield the sample dry weight ( $w_d$ ). Then, the moisture contents (*Moist%*) were determined with the formula  $Moist\% = \frac{w_t - w_d}{w_t} (100)$  and the bulk densities (*BD*) were determined using  $w_d$  with the formula  $BD = w_d/v$ . To determine the organic matter contents (*OM*), the oven-dried soil was subjected to 2-hr muffle-furnace combustion at 500 C to oxidize all organic matters to yield a final weight ( $w_f$ ). The formula  $OM = \frac{w_d - w_f}{w_d} (100)$  was used to calculate the organic matter content of an analytical sample.

#### *pH*

The pH of soil, compost, and deciduous leaf litter samples was tested using the Hanna Portable pH/EC Tracer Meter (Hanna Instruments, Woonsocket, RI). Sample slurries were made by mixing well air-dried analytical samples with distilled water within the range of solid:water ratios commonly used for pH-testing of soils and raw and composted plant materials (e.g. Ramdania *et al.* 2015, Offord *et al.* 2014, Jimenez and Garcia 1991, Whitehead *et al.* 1981). Then, stable pH-readings of the sample slurries were obtained with the pH meter.

#### *Plant Nutrients*

The levels of readily bioavailable plant macro-nutrients nitrate-nitrogen, phosphate, and potassium in experimental site soils before and after soil amendment applications as well as in compost and leaf litter samples were determined colourimetrically using the aqueous analytic methods by LaMotte Company (2012; Chestertown, MD). The nitrate-nitrogen, phosphate, and potassium ion were referred to as the extractable N, P, and K (Ex-N, Ex-K, and Ex-P) in this study, respectively.

Sample nutrient extracts were made with air-dried analytical samples mixed well with distilled water and LaMotte Acid Extraction Solution (Cat.# 6361-H; 3% HCl and 2% H<sub>2</sub>SO<sub>4</sub>), followed by funnel-filtering the extraction slurry through filter paper (pore size 11 µm). Next, the aqueous nutrient extracts were chemically processed for colourimeter readings to give the analytical sample nutrient levels. The Ex-N levels of the analytical samples were determined using the cadmium reduction method; the Ex-P levels were determined using the ascorbic acid reduction method; and the Ex-K levels of analytical

samples were determined using the tetraphenylborate method. To obtain reliable colourimeter reading above the method detection limits and within the detection ranges, necessary dilutions or concentrations of the analytical solutions were made by adjusting the ratio of nutrient extract and distilled water. The detection range for the cadmium reduction method, the ascorbic acid reduction method, and the tetraphenylboron method by the SMART 3 Colorimeter™ were 0.10-3.00 ppm  $\text{NO}_3^- \text{N}_{(\text{aq})}$  (Ex-N), 0.05-10.0 ppm  $\text{PO}_4^{3-}{}_{(\text{aq})}$  (Ex-P), and 0.8-3.00 ppm  $\text{K}^+{}_{(\text{aq})}$  (Ex-K), respectively.

#### **2.2.4 Measuring the Establishment of Seeded Native Plants and Naturally Colonised Plants**

##### *Seed Germination*

The number of juvenile plants and their shoot lengths inside each experimental plot were measured in mid-June 2015 to investigate the effect of soil alterations by amendment agents on seed germination. Specifically, within the centre subplot of each experimental plot, the number and shoot lengths of all native seedlings along two diagonal transects were taken. The number and shoot lengths of seedlings emerged from wild plant seeds that colonised the barren soil surface of the surrounding subplots within each experimental plot were also taken.

##### *Summer Time Plant Dynamics*

Monthly plant measurements at the middle of July and August, 2015 were taken to monitor the effect of soil alterations on the number and shoot lengths of seeded native plants and those of the naturally colonised plants plus colonised plant species characteristics and diversity. The number and shoot lengths of all seeded native plants along two horizontal transects located randomly perpendicular to the reference edge of the central subplot inside each experimental plot were recorded. Similarly, the number, shoot lengths and species name of all naturally colonised plants along three random horizontal transects within the subplots surrounding the centre subplot in each experimental plot were recorded (Figure 4). The herbaceous flowering species were identified using *Newcomb's Wildflower Guide* (Newcomb 1977).

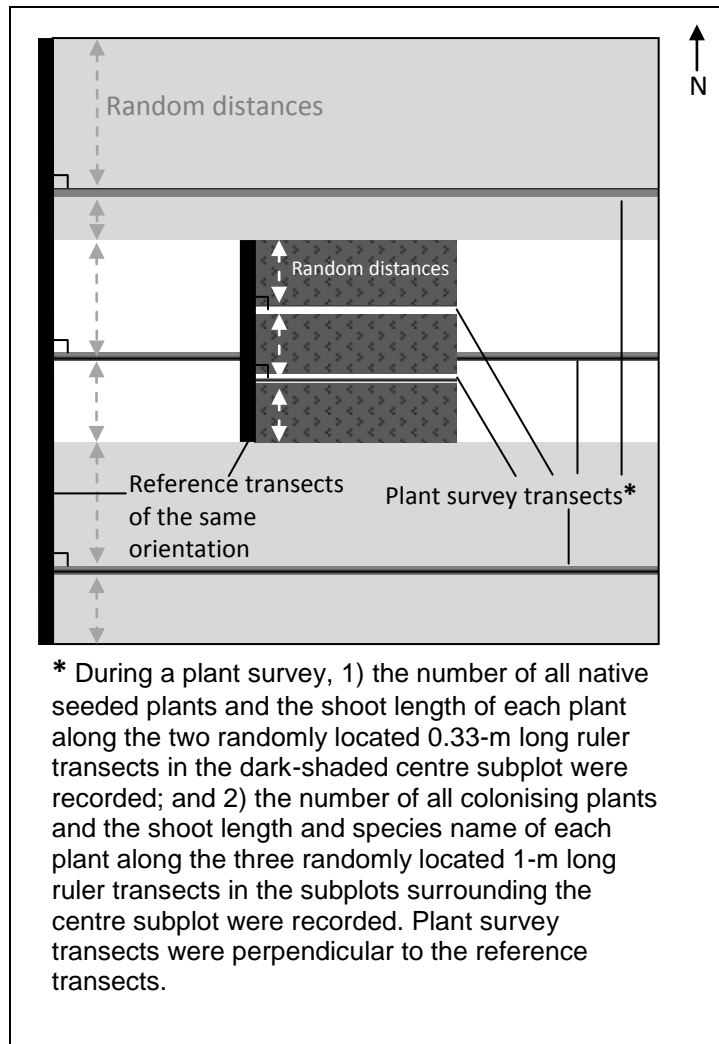


Figure 4. Random-transects scheme for surveying seeded native plants in the dark-shaded centre subplot and surveying naturally colonised plants in the surrounding subplots inside a square experimental plot.

## 2.3 Data Analyses

For characterisation of the physical and chemical properties of experimental site soil and those of the soil amendment agents, the values of each of the five analytical samples of each type of material were the true replicate values for statistical analyses (Appendix I Part I). For analyses of the physical and chemical properties of experimental soils with and without soil amendment agents, the arithmetic average of the measurements of the two subsamples from each experimental plot was used as the true-replicate values for that experimental plot of particular treatment (Appendix I Part II).

For seeded native plants, the total number of plants ( $P_{\#}$ ) found along all sampling transects in an experimental plot was the true-replicate value of the number of native plants in that experimental plot. The same rule was used to obtain the true-replicate value of the number of naturally colonised plants in an experimental plot. The average shoot length ( $P_L$ ) of seeded native plants and that of naturally colonised plants in each experimental plot was the respective true-replicate value of shoot length of that plant type for that experimental plot; this was calculated using the formula  $P_L = \frac{\sum \text{Shoot length of plants}}{P_{\#}}$ . The Shannon-Wiener Index ( $H'_w$ ) was used to measure the species diversity of the naturally colonised plants in each experimental plot, which was calculated using the formula  $H'_w = -\sum_{i=1}^R P_i \ln P_i$ ; where  $R$  was the species richness, or the total number of colonised plant species found on the barren soil within an experimental plot, and  $P_i$  stood for the percentage abundance of a species relative to  $P_{\#}$  of colonised plants on the same barren soil. The  $H'_w$  of an experimental plot is the true-replicate value of the diversity index of naturally colonised plants for that experimental plot (Appendix I Part III and IV).

Afterward, the inferential statistical analyses were based on true-replicate values from the laboratory analysis and field survey datasets (Appendix I). Table 2 summarised the inferential statistical analyses performed to investigate the potential differences in the physical and chemical characteristics of the soil amendment agents and the experimental site soil, and the research objectives: 1) potential differences in the properties of the experimental site soil after treatment with each soil amendment agent in comparison with control, 2) potential differences in the number and shoot lengths of seeded native plants along their respective sampling transects and those of naturally colonised plants on differentially treated barren soils and on control, and 3) potential differences in the species richness, evenness, and  $H'_w$  diversity index of the colonised plants along the sampling transects on the differentially treated soils and on un-amended soil control. For each specific research objective, the parametric tests were used when the original or natural logarithmic-transformed ( $\ln(x)$ ) sample datasets failed to reject the null hypothesis of normal distribution tested by the Shapiro-Wilk test; otherwise, the non-parametric tests were used when the original and  $\ln(x)$ -transformed sample datasets rejected the null hypothesis of normal distribution by Shapiro-Wilk test. Coupled with the presentation of the results of the inferential statistical analyses were the displays of boxplots showing the structure of the analysed sample datasets for visual examination of the degree of differences between experimental treatments on different sampling months. All inferential statistical analyses were conducted with the software IBM © SPSS © Statistics (version 23) at significance level  $\alpha = 0.05$ .

Table 2. Inferential statistical analyses used to answer the research questions on effects of deciduous tree leaf litter and plant residue compost, which can have different physicochemical properties, on soil properties, the number and shoot lengths of seeded native plants and those of naturally colonised plants, and the species diversity of the colonised plants.

<b>Analysis</b>	<b>Null hypothesis</b>	<b>Parametric or non-parametric statistical tests* performed and sample size</b>
Characterisation of soil amendment agents and the experimental site soil	There was no significant difference in the composition of the soil amendment agents (leaf litter and compost) and that of the experimental soil in terms of bulk density, moisture content, organic matter content, pH, N, P, and K.	One-way ANOVA with Bonferroni post-hoc tests (Kruskal-Wallis test followed by Mann-Whitney's U tests) <i>n</i> = 5
Comparing soil response to treatments	There was no significant difference in the physical and chemical properties of the soil receiving no treatment, leaf litter, or compost in terms of soil bulk density, moisture content, SOM, pH, Ex-N, Ex-P, and Ex-K; and the effect was constant in time across June, July, and August, 2015.	Two-way ANOVA repeated-measure with Bonferroni post-hoc tests (Kruskal-Wallis test of monthly data across three treatments and observe for consistency in between-treatment differences across sampling months) <i>n</i> = 4
Comparing seeded native plant and naturally colonised plant seedling count and shoot lengths in June 2015	There was no significant difference in the number and shoot lengths of seedlings along the survey transects on experimental plots with soils receiving no treatment, leaf litter, or compost.	One-way ANOVA with Bonferroni post-hoc tests (Kruskal-Wallis test followed by Mann-Whitney's U tests) <i>n</i> = 4
Comparing seeded native plant and naturally colonised plant number and shoot lengths in the summer (July and August) 2015	There was no significant difference in the number and shoot lengths of plants along the sampling transects due to different soil treatments over the summer (in August) 2015; and within each treatment, there was no significant change in plant number and shoot length along the survey transects between July and August, 2015.	One-way ANOVA of end-of-summer (August) 2015 plant number and shoot length data followed by Bonferroni post-hoc tests to observe for between treatment effects; within each treatment, one-way ANOVA repeated-measure of monthly data to test for significance in plant number and shoot length change across months. (Kruskal-Wallis test of August data across treatments followed by Mann-Whitney U test; within each treatment, Friedman test across months) <i>n</i> = 4
Comparing naturally colonised plant species richness and diversity	There was no significant difference in colonised plant species richness along the survey transect on experimental plots receiving different soil treatments; and there was no significant difference in colonised plant species' Shannon-Wiener Diversity Index on experimental plots receiving different soil treatments in July	One-way ANOVA of plant lifecycle-corrected <sup>†</sup> species richness and diversity index data across three treatments with Bonferroni post-hoc tests (Kruskal-Wallis test followed by Mann-Whitney's U tests) <i>n</i> = 4

Comparing distribution pattern of the relative abundances of naturally colonised plants	and August 2015. There was no observable difference in the distribution of the relative abundances of naturally colonised plant species along the survey transects on experimental plots with different soil treatments in July and August 2015.	Rank-abundance curve plotted with the average of the plant lifecycle-corrected <sup>†</sup> relative abundance value of each naturally colonised plant species within experimental plots of the same soil treatment across the three soil treatments $n = 4$
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\* the parametric tests were used when the original or natural logarithmic-transformed ( $\ln(x)$ ) sample datasets failed to reject the null hypothesis of normal distribution as tested by the Shapiro-Wilk test; otherwise, the non-parametric tests were used for non-normally distributed sample datasets.

<sup>†</sup> For whole summer analysis of naturally colonised plant diversity across multiple months, see Appendix I Part IV C. 2. for detailed explanation of the method used to correct life-cycle effects of plants.

### 3.0 Results

#### 3.1 Soil Response to Amendment Agent Applications

##### 3.1.1 Soil Physical Properties

Following monthly repeated measurements in June, July, and August 2015, soil receiving compost treatment had significantly higher, by 1.6 times, moisture content ( $20.5 \pm 1.8\%$  ww [mean  $\pm$  S.D.;  $n = 4$ ]) than the un-amended experimental site soil (i.e. the control), while the leaf litter treatment resulted only in a slightly higher soil moisture content ( $14.5 \pm 0.9\%$  ww) than the control (two-way ANOVA,  $F_{\text{treatment}}(2,6) = 55.76$ ,  $p = 1.3 \times 10^{-4}$ ). Across months, there appeared to be statistically significant variation in soil moisture content within each treatment factor (two-way ANOVA,  $F_{\text{time}}(2,6) = 6.69$ ,  $p = 0.03$ ). However, the interaction between treatment factor and sampling time was not significant (two-way ANOVA,  $F_{\text{treatment} \times \text{time}}(4,12) = 3.31$ ,  $p = 0.06$ ) (Figure 5).

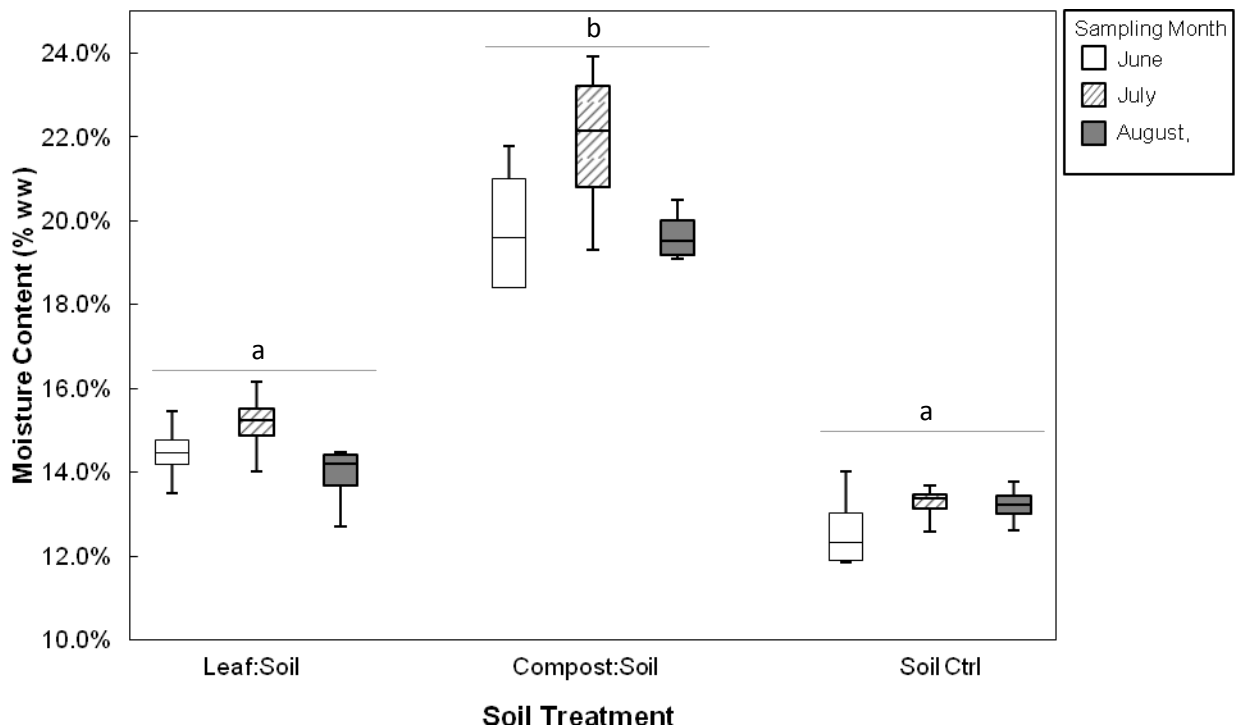


Figure 5. Comparison of the moisture content of experimental site topsoil amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) at the beginning of June, July, and August 2015 ( $n = 4$ ). Note: statistically significant differences are indicated with different letters above boxplots.

Across June, July, and August 2015, the bulk density of soil receiving compost treatment ( $1.0 \pm 0.07$  g/cm<sup>3</sup> dw [mean  $\pm$  S.D.;  $n = 4$ ]) was significantly lower, by 1.3 times, than that of control and leaf

litter treatment, while the bulk density of leaf litter-treated soil was insignificantly different from that of the control (two-way ANOVA,  $F_{\text{treatment}}(2,6) = 152.01$ ,  $p = 7.0 \times 10^{-6}$ ). Across months, there seemed to be a statistically significant variation in soil bulk density (two-way ANOVA,  $F_{\text{time}}(2,6) = 7.56$ ,  $p = 0.02$ ). However, there was no significant interaction between experimental treatment and sampling time (two-way ANOVA,  $F_{\text{treatment} \times \text{time}}(4,12) = 2.34$ ,  $p = 0.11$ ) (Figure 6).

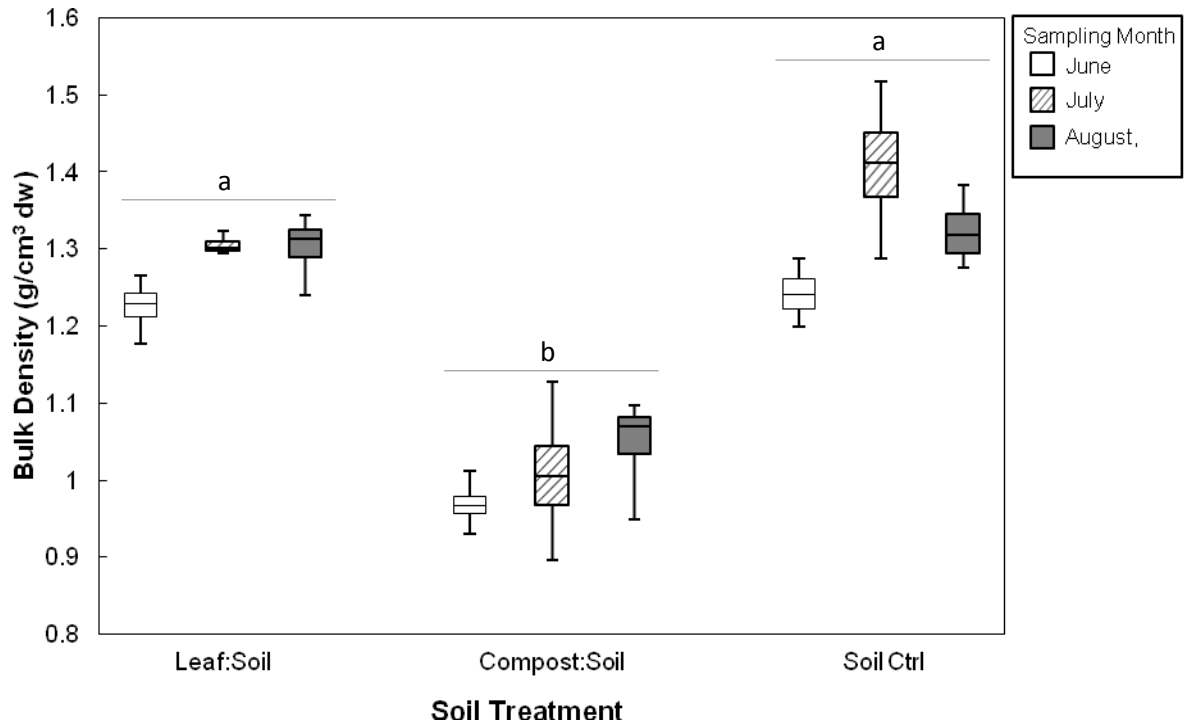


Figure 6. Comparison of the bulk density of experimental site topsoil amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) at the beginning of June, July, and August 2015 ( $n = 4$ ). Note: statistically significant differences are indicated with different letters above boxplots.

### 3.1.2 Soil Chemical Properties

During June, July, and August 2015, the SOM level of compost treatment ( $9.1 \pm 1.8\%$  dw [mean  $\pm$  S.D.;  $n = 4$ ]) was significantly higher, by 3.5 times, than that of the control and by 2.4 times than that of the leaf litter treatment, while the leaf litter treatment yielded 1.4 times higher, but statistically insignificantly different, level of SOM level ( $3.7 \pm 0.8\%$  dw) than the control (two-way ANOVA,  $F_{\text{treatment}}(2,6) = 58.76$ ,  $p = 1.2 \times 10^{-4}$ ). Between months, there was no significant variations in SOM levels within each treatments (two-way ANOVA,  $F_{\text{time}}(2,6) = 0.81$ ,  $p = 0.49$ ). The effect of amendment agent treatment on SOM level was consistent across June, July, and August 2015 – there was insignificant interaction between treatment factors and time (two-way ANOVA,  $F_{\text{treatment} \times \text{time}}(4,12) = 0.70$ ,  $p = 0.61$ ) (Figure 7).



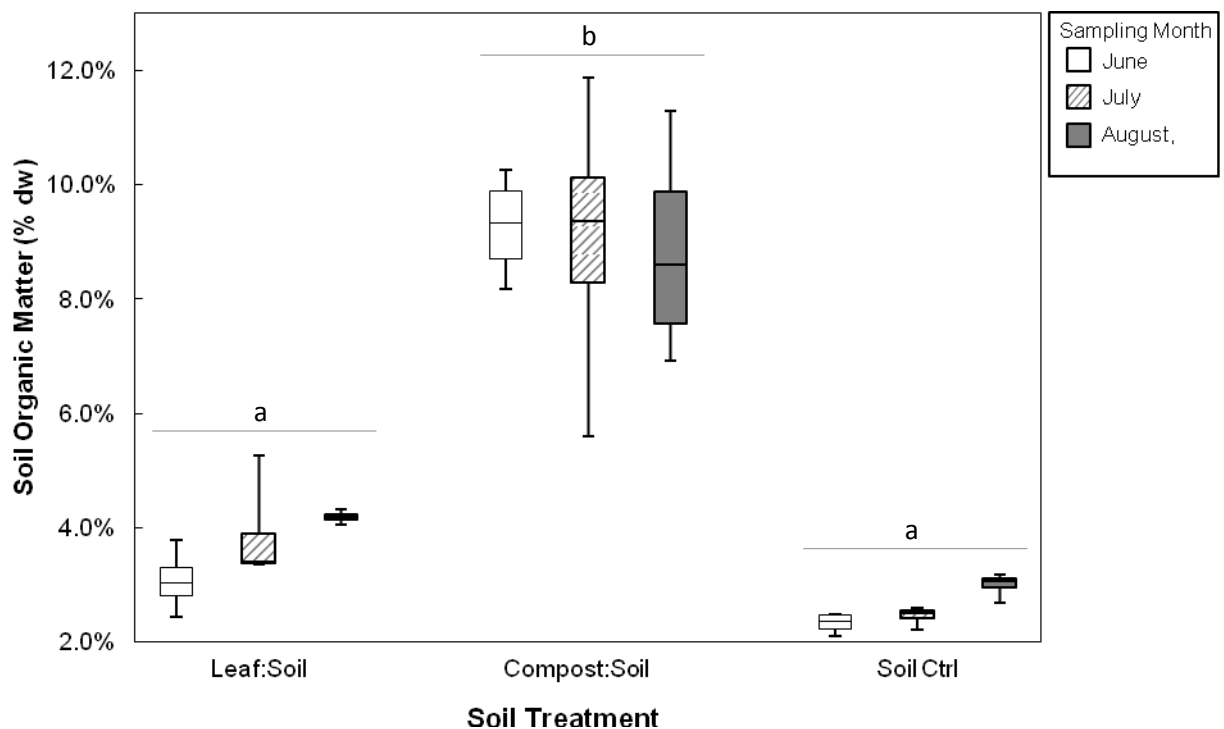


Figure 7. Comparison of the soil organic matter (SOM) content of experimental site soil amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) at the beginning of June, July, and August 2015 ( $n = 4$ ). Note: statistically significant differences between treatments are indicated with different letters above the boxplots.

For soil pH in June to August 2015, the compost treatment had 0.22 units lower soil pH ( $7.74 \pm 0.10$  [mean  $\pm$  S.D.;  $n = 4$ ]) than the control and 0.21 units lower soil pH than the leaf litter treatment. The difference in pH between compost treatment and control was significant, but the difference between leaf litter treatment and control was insignificant (two-way ANOVA,  $F_{\text{treatment}}(2,6) = 32.17$ ,  $p = 1.0 \times 10^{-3}$ ). Between months, soil pH varied significantly (two-way ANOVA,  $F_{\text{time}}(2,6) = 17.36$ ,  $p = 3.0 \times 10^{-3}$ ); soil pH's in July were significantly lower than those in June and August in all treatment factors. In experimental plots treated with leaf litter, the average soil pH varied by as much as 0.26 units. Likewise, the average soil pH fluctuated by as much as 0.28 units in control plots. With less variation, average soil pH varied by 0.16 units in experimental plots treated with compost. The effect of amendment agents on soil pH during the June-July-August 2015 period was consistent across months, with insignificant interactions between treatment factors and time (two-way ANOVA,  $F_{\text{treatment} \times \text{time}}(4,12) = 1.64$ ,  $p = 0.23$ ) (Figure 8).

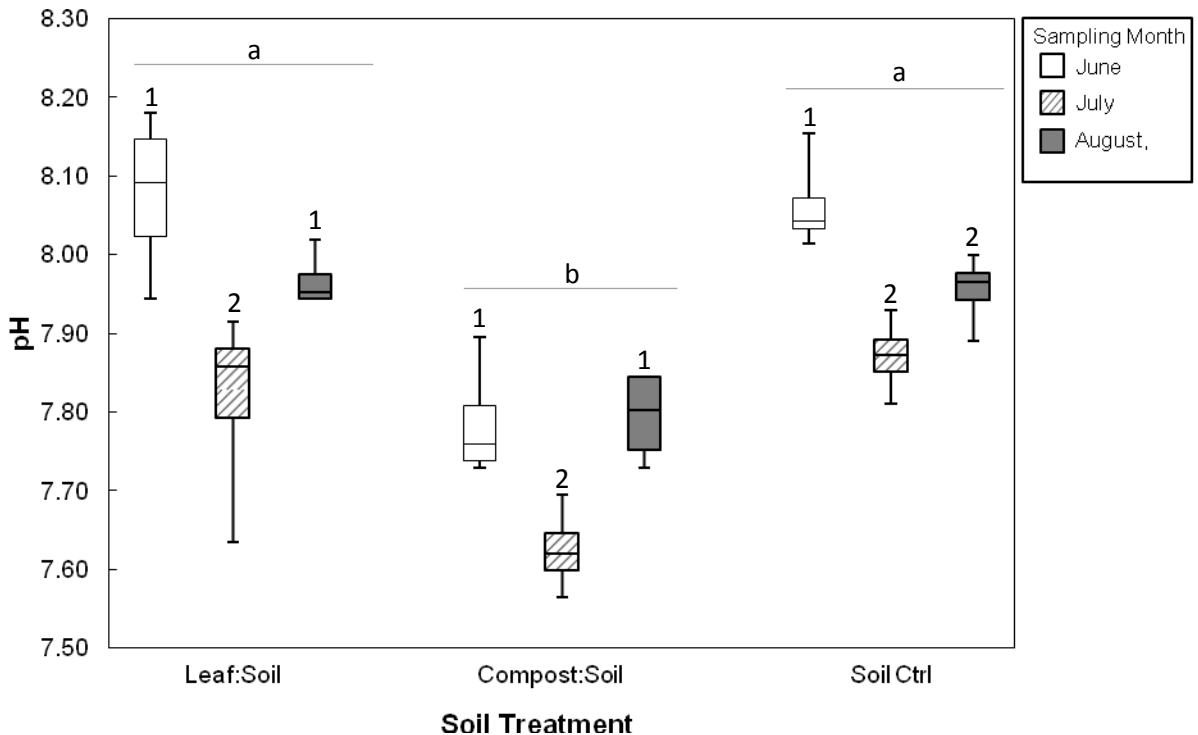


Figure 8. Comparison of the pH of experimental site soil amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) at the beginning of June, July, and August 2015 ( $n = 4$ ). Note: statistically significant differences between treatments are indicated with different letters and statistically significant differences between sampling months within each treatment are indicated with different numbers above individual boxplots.

Regarding soil nutrients, the compost treatment resulted in 1.6 times higher soil Ex-N content ( $6.0 \pm 2.7$  kg/ha air dw [mean  $\pm$  S.D.;  $n = 4$ ]) than the control and 1.8 times higher than the leaf litter treatment. The soil Ex-N content as result of leaf litter treatment was slightly lower than of the control. Compost treatment, not leaf litter treatment, gave rise to significantly different soil Ex-N content than the control (two-way ANOVA,  $F_{\text{treatment}}(2,6) = 24.06$ ,  $p = 1.0 \times 10^{-3}$ ). There was significant variation in soil Ex-N content across time (two-way ANOVA,  $F_{\text{time}}(2,6) = 7.82$ ,  $p = 0.02$ ). The average Ex-N content of compost-treated soil appeared to dropped sharply from 8.8 kg/ha air dw in June to 5.6 kg/ha air dw in July and further to 4.0 kg/ha air dw in August 2015. Meanwhile, the soil Ex-N content in leaf litter treatment plots and control plots varied around 2.8 to 4.2 and 3.3 to 4.2 kg/ha air dw, respectively. The interaction of treatment and sampling time was statistically significant due to sharp drop in Ex-N content of compost-treated soil (two-way ANOVA,  $F_{\text{treatment} \times \text{time}}(4,12) = 5.92$ ,  $p = 0.007$ ) (Figure 9).

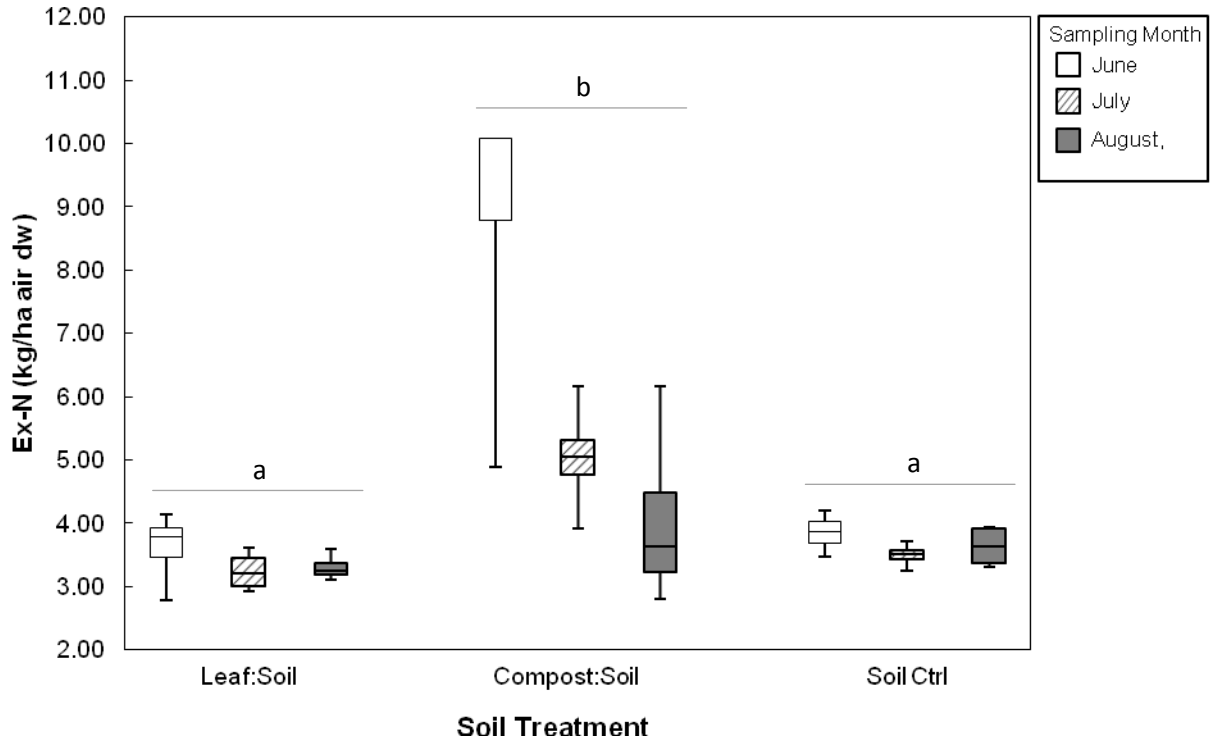


Figure 9. Comparison of the Ex-N (nitrate-nitrogen) content of experimental site soil amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) at the beginning of June, July, and August 2015 ( $n = 4$ ). Note: statistically significant differences between treatments are indicated with different letters above boxplots.

In the period of June to August 2015, soil in experimental plots with compost had 11.4 and 4.7 times higher Ex-P content ( $65 \pm 34.8$  kg/ha air dw [mean  $\pm$  S.D.;  $n = 4$ ]) than soil in the control plots and soil in leaf litter treated plots, respectively. Compost treatment resulted in significantly different soil Ex-P content from the control and leaf litter treatment; leaf litter treatment resulted in insignificantly higher soil Ex-P content ( $14 \pm 17.2$  kg/ha air dw) than the control (two-way ANOVA with  $\ln(x)$  transformation,  $F_{\text{treatment}}(2,6) = 7.19$ ,  $p = 0.03$ ). The response pattern was consistent across June, July, and August 2015 (two-way ANOVA with  $\ln(x)$  transformation,  $F_{\text{time}}(2,6) = 5.29$ ,  $p = 0.05$ ). There was no significant interaction between treatment and time period (two-way ANOVA with  $\ln(x)$  transformation,  $F_{\text{treatment} \times \text{time}}(4,12) = 1.58$ ,  $p = 0.24$ ) (Figure 10).

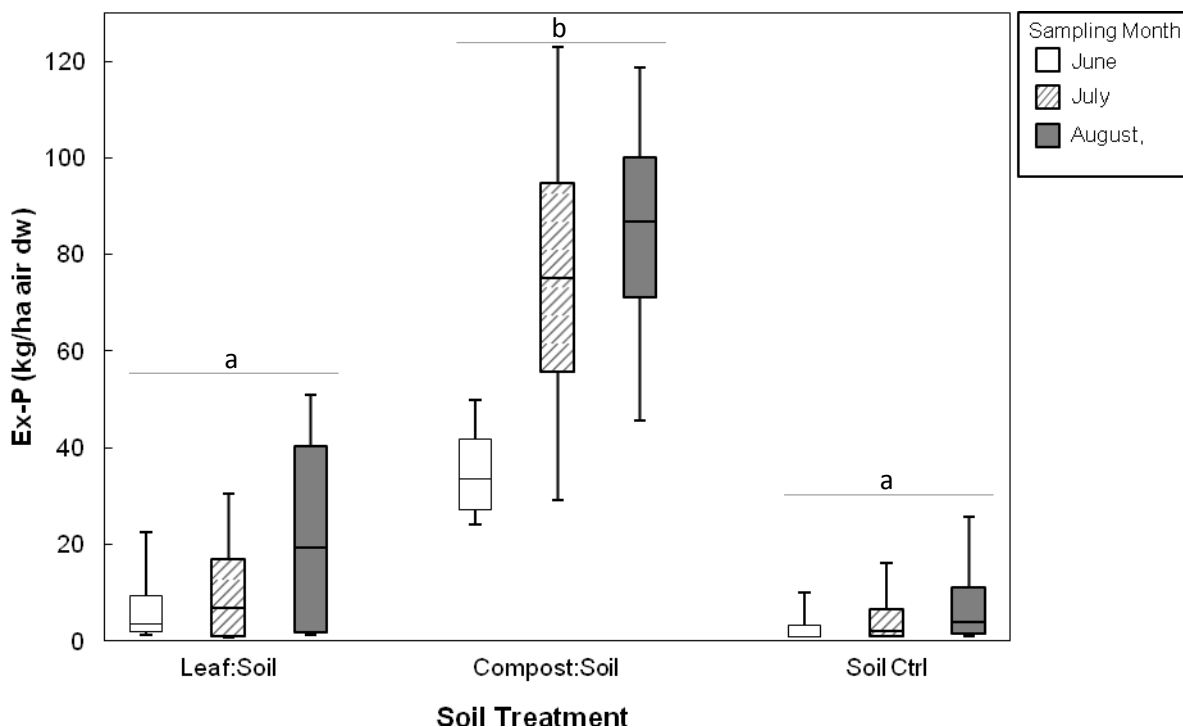


Figure 10. Comparison of the Ex-P (phosphate) content of experimental site soil amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) at the beginning of June, July, and August 2015 ( $n = 4$ ). Note: statistically significant differences between treatments are indicated with different letters above boxplots.

In terms of Ex-K, compost treatment gave rise to soil with 11.7 and 9.0 times higher Ex-K content ( $550 \pm 168.8$  kg/ha air dw [mean  $\pm$  S.D.;  $n = 4$ ]) than the control and leaf litter treatment, respectively. Leaf litter-treated soil had slightly higher Ex-K than the control. Compost treatment, not leaf litter treatment, resulted in soil having significantly different Ex-K content than control (two-way ANOVA,  $F_{\text{treatment}}(2,6) = 123.89$ ,  $p = 1.3 \times 10^{-5}$ ). Across sampling months, time did not affect the effect of treatment on soil Ex-K content (two-way ANOVA,  $F_{\text{time}}(2,6) = 2.58$ ,  $p = 0.16$ ). The interaction between treatment factor and sampling time was not significant (two-way ANOVA,  $F_{\text{treatment} \times \text{time}}(4,12) = 0.94$ ,  $p = 0.48$ ) (Figure 11).

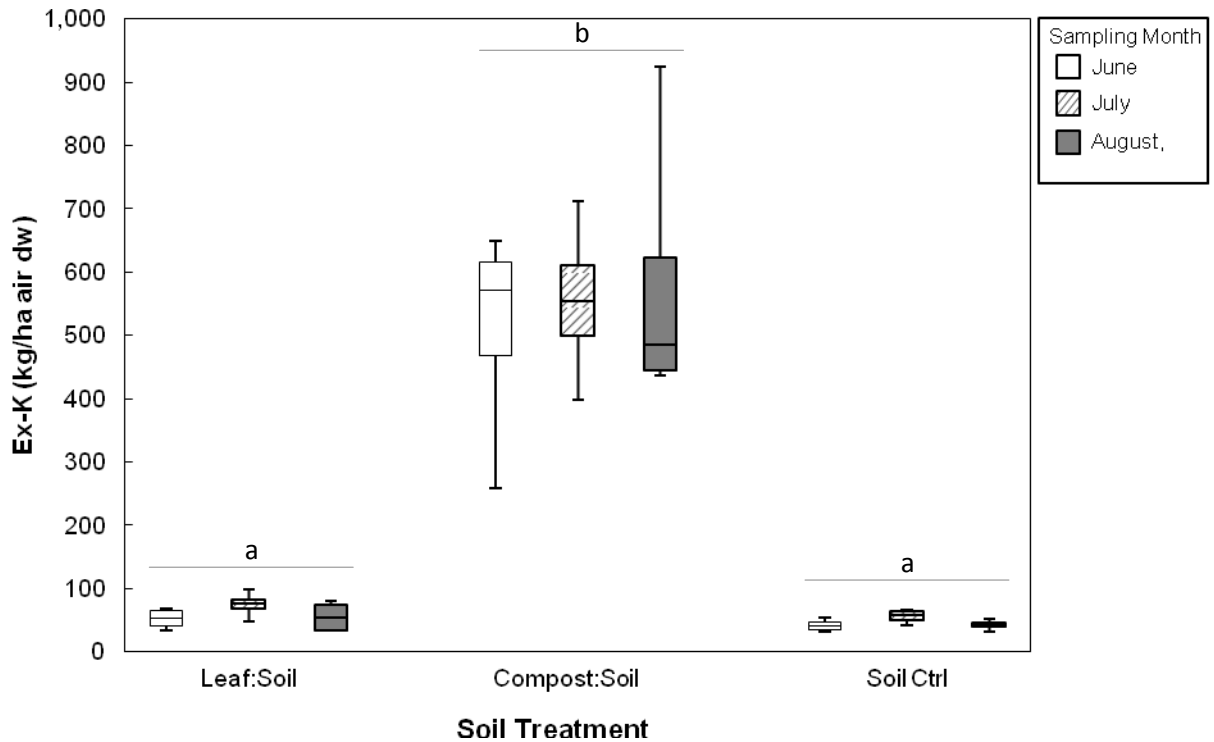


Figure 11. Comparison of the Ex-K (potassium ion) content of experimental site soil amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) at the beginning of June, July, and August, 2015 ( $n = 4$ ). Note: statistically significant differences are indicated with different letters above boxplots.

### 3.2 Plant Germination in Response to Soil Treatments

#### 3.2.1 Number and Shoot Lengths of Native Plant Seedlings in June 2015

At the middle of June 2015, the native seeds planted a month ago germinated, producing clusters of loose emerged seedlings in the centre subplots of experimental plots (Appendix II A. contains photos of experimental plots in June 2015). The number of the native plant seedlings along two diagonal plant survey transects in the centre subplot on soil treated with amendment agents was significantly different than that in the centre subplots of the control in mid-June, 2015 (one-way ANOVA,  $F(2,9) = 6.69$ ,  $p = 0.02$ ). Subplots on compost-treated soil had the highest number of seedlings along the plant survey transects ( $64.0 \pm 16.7$  [mean  $\pm$  S.D.;  $n = 4$ ]), followed by subplots on leaf litter-treated soil ( $58.3 \pm 10.8$ ). The number of native seedlings along the plant survey transects in subplots on leaf litter- and compost-treated soil did not differ significantly from each other, they were significantly higher, by 1.8 and 2.0 times, respectively, than that in subplots on the control soil (Figure 12).

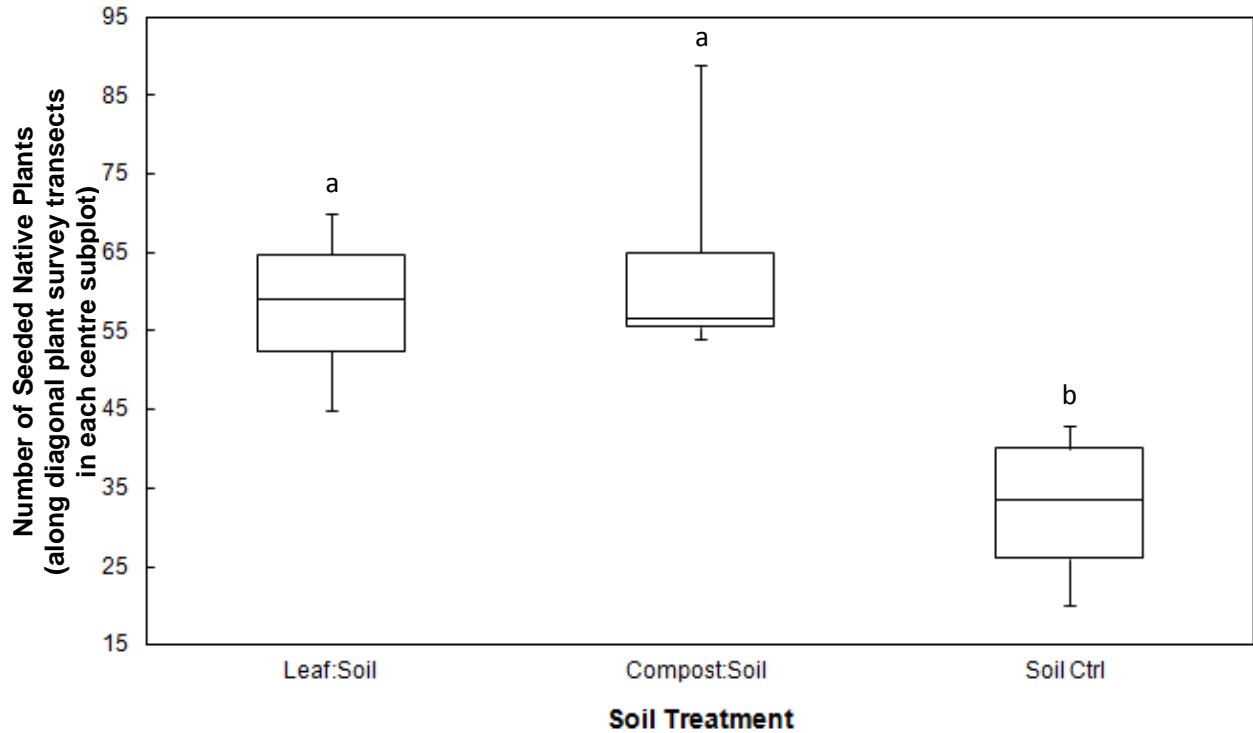


Figure 12. Comparison of the number of native seedlings (in June 2015) one month after seed planting on experimental plot soil amended with 50% (v/v) of deciduous leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) ( $n = 4$ ). Note: statistically significant differences in plant response on different soils are indicated with different letters above boxplots.

Soil treated with leaf litter and compost produced native seedlings with significantly different shoot lengths than the control (Kruskal-Wallis test,  $\chi^2(2) = 8.14$ ,  $p = 0.02$ ). The shoot length of native seedlings on both compost- (median = 7.8 cm;  $n = 4$ ) and leaf litter-treated plots (median = 6.3 cm) did not differ significantly from each other, but their values were significantly higher, by 2.1 and 2.6 times, respectively, than native seedlings in the control plots (median = 3.0 cm) (Figure 13).

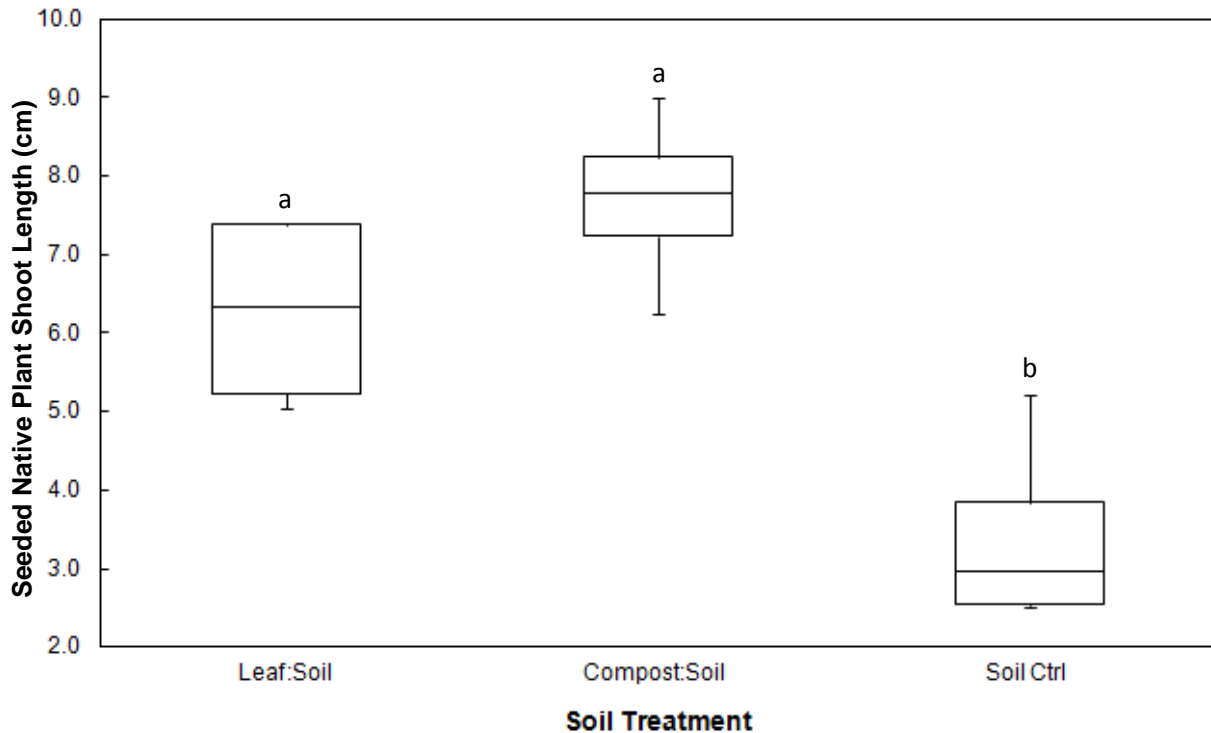


Figure 13. Comparison of the shoot lengths of native seedlings (in June 2015), one month after seed planting on experimental plot soil amended with 50% (v/v) of deciduous leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) ( $n = 4$ ). Note: statistically significant differences in plant response on different soils are indicated with different letters above boxplots.

### 3.2.2 Number and Shoot Lengths of Naturally Colonised Plant Seedlings in June 2015

On the barren soil of subplots surrounding the central subplot in each experimental plot, randomly distributed seedlings of naturally colonised plants were visible in mid-June, 2015, a month and half after site preparation (Appendix II A.). The total number of naturally colonised plant seedlings in the surrounding subplots of an experimental plot that received leaf litter treatment, compost treatment, and no treatment (control) did not differ significantly from each other (one-way ANOVA,  $F(2,9) = 1.85$ ,  $p = 0.21$ ). The total number of naturally colonised plant seedlings in the surrounding subplots on compost treatment subplots varied largely, but was overall insignificantly higher ( $76.5 \pm 34.8$  [mean  $\pm$  S.D.;  $n = 4$ ]) than those in the leaf litter-treated surrounding subplots ( $61.7 \pm 12.6$ ), both which had insignificantly higher total count of naturally colonised plant seedlings in the surrounding subplot than the control ( $43.2 \pm 20.8$ ) (Figure 14).



Figure 14. Comparison of the number of naturally colonised plant seedling in June 2015 on barren surface of experimental plot soil amended with 50% (v/v) of deciduous leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) in May 2015 ( $n = 4$ ). Note: statistically significant differences in plant responses on different soils are indicated with different letters above boxplots.

However, there was significant difference in the shoot lengths of the naturally colonised plant seedlings among the treatment factors in mid-June, 2015 (Kruskal-Wallis test,  $\chi^2(2) = 7.91, p = 0.02$ ). Naturally colonised plant seedlings on soil with compost treatment had about 3.6 and 5.1 times higher median shoot lengths (median = 7.2 cm;  $n = 4$ ) than those on the control soil and soil amended with leaf litter, respectively. Naturally colonised plant seedling shoot lengths differed significantly between compost and other treatment factors but did not differ significantly between leaf litter treatment and control (Figure 15).



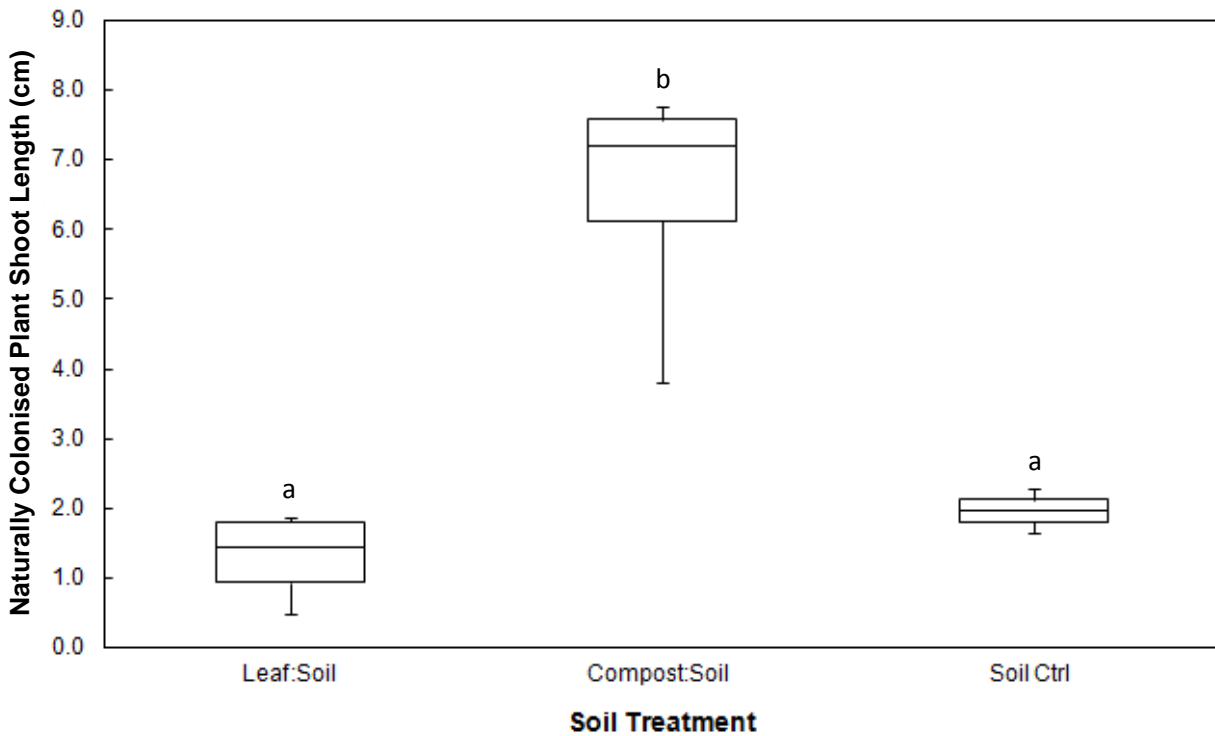


Figure 15. Comparison of the shoot lengths of naturally colonised plant seedlings in June 2015 on barren surface of experimental plot soil amended with 50% (v/v) of leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) ( $n = 4$ ) in May 2015. Note: statistically significant differences in plant response on difference soils are indicated with different letters above boxplots.

### 3.3 Response of Plants to Soil Treatments in July and August 2015

#### 3.3.1 Number and Shoot Lengths of Seeded Native Plants in July and August 2015

Over the course of summer 2015, patches of native grass species occupied the central subplots of all experimental plots. In mid-August, even a few black eyed Susan plants from the commercial native seed mixture bloomed on compost-amended soil (Appendix II B.). Three months after plant establishment by seeding, the number of seeded native plants along two random plant survey transects in each centre subplot on compost-amended soil ( $47.8 \pm 6.7$  [mean  $\pm$  S.D.;  $n = 4$ ]) was insignificantly higher than that on leaf litter-amended soil ( $41.5 \pm 5.5$ ). The number of seeded native plants along the plant survey transects in centre subplots of leaf litter and compost soil treatments were significantly higher, by 1.5 and 1.7 times, respectively, than that on the control site soil (one-way ANOVA,  $F(2,9) = 11.42$ ,  $p = 3.0 \times 10^{-3}$ ). From July to August, the number of seeded native plants along plant survey transects on leaf litter-treated soil dropped slightly (one-way ANOVA with repeated measure,  $F(1,3) = 2.98$ ,  $p = 0.18$ ) from  $50.5 \pm 8.4$  to  $41.5 \pm 5.5$ . But the number of seeded native plants along plant survey transects on compost-treated soil

increased significantly (one-way ANOVA with repeated measure,  $F(1,3) = 17.75$ ,  $p = 0.02$ ) from July ( $38.3 \pm 4.2$ ) to August ( $47.8 \pm 6.7$ ), as new seedlings emerged. Meanwhile, the number of seeded native plants along the plant survey transects on the control site soil dropped somewhat (one-way ANOVA with repeated measure,  $F(1,3) = 8.22$ ,  $p = 0.06$ ) from July ( $38.5 \pm 9.4$ ) to August ( $28.3 \pm 5.4$ ) (Figure 16).

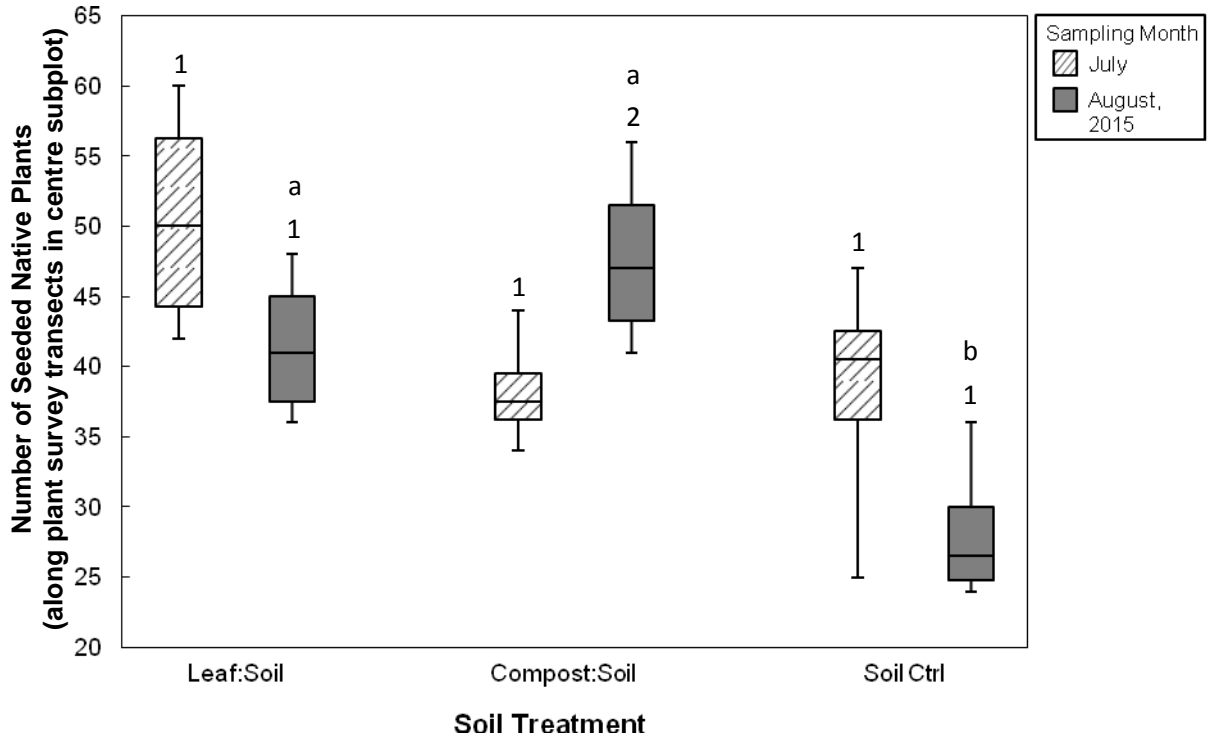


Figure 16. Comparison of the number of seeded native plants in the summer 2015, two and three months after seed planting on experimental plot soils amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) ( $n = 4$ ). Note: statistically significant differences in plant response on different soils in August 2015 are indicated with different letters, and within each soil treatment, differences between months are indicated with different numbers above boxplots.

During spring and summer 2015, there was noticeable growth of seeded native plants on all experimental plots. Three months after seed planting, different experimental treatments gave rise to significantly different shoot lengths of seeded native plants in August (Kruskal-Wallis test,  $\chi^2(2) = 9.88$ ,  $p = 7.0 \times 10^{-3}$ ). At the middle of August 2015, seeded native plants on compost-amended soil had significantly higher shoot lengths (median = 40.0 cm;  $n = 4$ ), by 1.7 times, than plants on the experimental site soil (control) and by 2.8 times than plants on leaf litter-amended soil. The shoot lengths of seeded native plants on leaf litter-amended soil were significantly lower than those of plants on control site soil by 1.6 times. On leaf litter-amended soil, the median shoot lengths of seeded native plants increased significantly (Friedman test,  $\chi^2(2) = 8.00$ ,  $p = 6.0 \times 10^{-3}$ ) from June (6.3 cm) to August (14.3 cm). Meanwhile, the median shoot lengths of seeded native plants increased significantly from 7.8 to 40.0

cm on compost-amended soil (Friedman test,  $\chi^2(2) = 8.00, p = 4.0 \times 10^{-3}$ ) and from 3.0 cm to 23.2 cm on control subplots (Friedman test,  $\chi^2(2) = 8.00, p = 5.0 \times 10^{-3}$ ) (Figure 17).

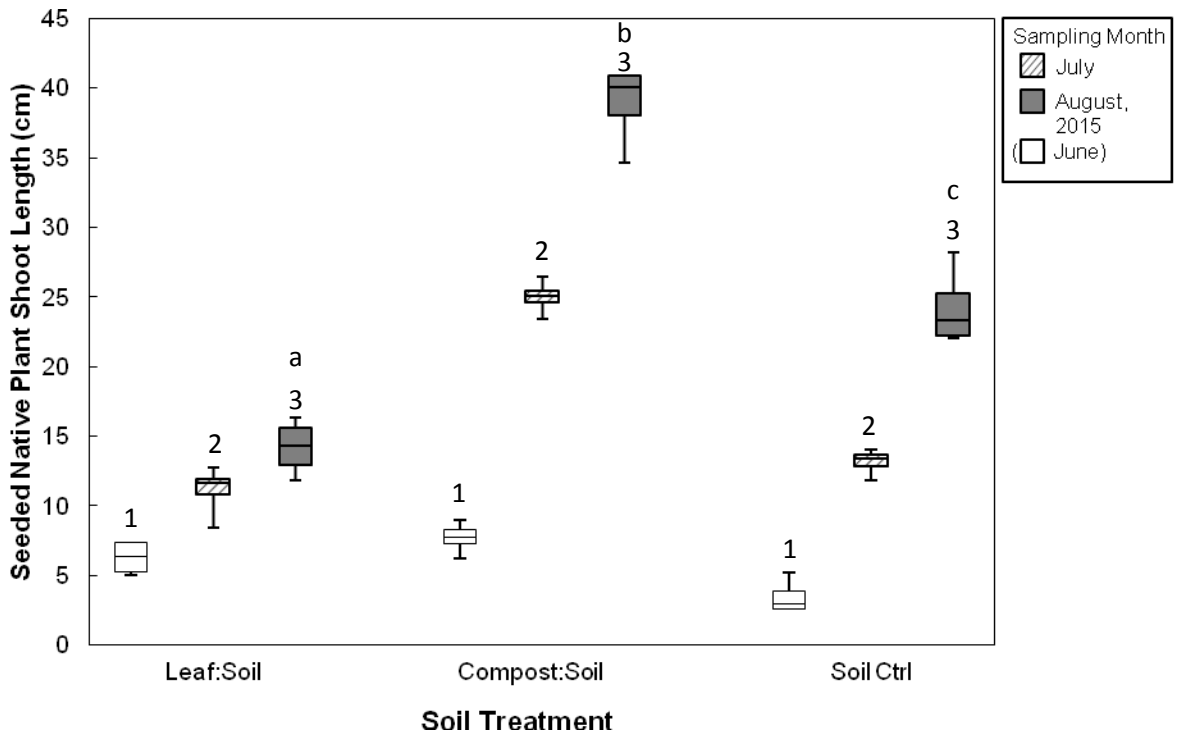


Figure 17. Comparison of the shoot lengths of seeded native plants in summer 2015, two and three months after seed planting on experimental plot soils amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) ( $n = 4$ ). Note: June shoot length data is carried from Figure 13) to here for convenience of comparison. statistically significant differences in plant response on different soils in August 2015 are indicated with different letters, and within each soil treatment, differences between months are indicated with different number above boxplots.

### 3.3.2 Number and Shoot Lengths of Naturally Colonised Plants in July and August 2015

By mid-August 2015, many plants colonised the barren soil of the subplots surrounding the central plot in each experimental plot (Appendix II B.). The naturally colonised plants were randomly distributed. In mid-August, the number of naturally colonised plants along the three random plant survey transects in the surrounding subplots of an experimental plot with and without treatments differed somewhat (one-way ANOVA,  $F(2,9) = 3.47, p = 0.08$ ). Specifically, the number of naturally colonised plants along the plant survey transects on compost-amended soil ( $19.0 \pm 5.5$  [mean  $\pm$  S.D.;  $n = 4$ ]) was significantly higher, by 1.3 and 1.6 times, than those on the control experiment site soil and on leaf litter-amended soil, respectively. From mid-July ( $14.3 \pm 6.1$ ) to mid-August 2015 ( $11.5 \pm 1.3$ ), the number of naturally colonised plants along the sampling transects on leaf litter-amended soil did not seem to change

significantly (one-way ANOVA with repeated measure,  $F(1,3) = 0.98$ ,  $p = 0.40$ ), despite a reduction in the variation of plant count over time. However, the number of naturally colonised plants along the sampling transects decreased significantly on compost-amended soil (one-way ANOVA with repeated measure,  $F(1,3) = 12.22$ ,  $p = 0.04$ ) from July ( $25.8 \pm 6.8$ ) to August ( $19.0 \pm 5.5$ ). Meanwhile, the number of naturally colonised plants along the sampling transects decreased insignificantly (one-way ANOVA with repeated measure,  $F(1,3) = 5.07$ ,  $p = 0.11$ ) from  $21.5 \pm 9.5$  to  $14.5 \pm 4.2$  on un-amended control soil (Figure 18). As August came, many plants on the once barren soil in the subplots surrounding the centre subplot in each experimental plot had seedpods and subsequently died, so the number of live plants decreased.

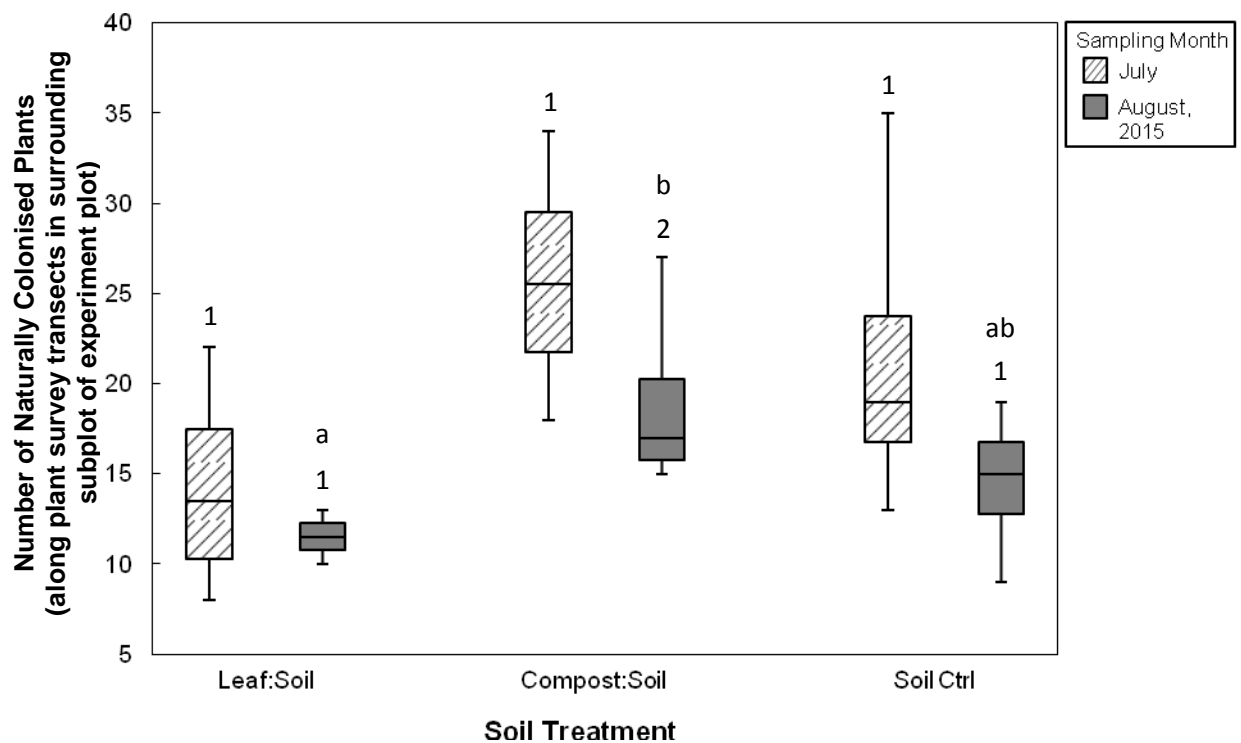


Figure 18. Comparison of the number of colonising weeds in summer 2015, two and three months after initial site preparation on experimental plot soils amended with 50% (v/v) leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) ( $n = 4$ ). Note: statistically significant differences in plant response on different soils in August 2015 are indicated with different letters, and within each soil treatment, differences between months are indicated with different number above boxplots.

The shoot lengths of the naturally colonised plants differed significantly in mid-August 2015 between soil treatments and control (Kruskal-Wallis test,  $\chi^2(2) = 9.85$ ,  $p = 7.0 \times 10^{-3}$ ). At that time, significant differences in the shoot lengths of the naturally colonised plants by soil treatment were as follows: compost-amendment had 2.55 times greater median shoot lengths of naturally colonised plants (24.8 cm;  $n = 4$ ) than control, which had 1.2 times greater median shoot length of naturally colonised

plants than the leaf litter-amendment. Across time, the growth of naturally colonised plants on leaf litter-amended soil was significant (Friedman test,  $\chi^2(2) = 6.50, p = 0.04$ ), with shoot length changing from a median value of 1.4 cm in June to 8.4 cm in August, 2015. Naturally colonised plants on compost-amended soil seemed to experience a large growth (Friedman test,  $\chi^2(2) = 6.00, p = 0.07$ ) from a median shoot length of 7.2 cm in June to a medium length of 24.8 cm in August 2015. By August, while some naturally colonised plants produced seedpods and stopped growing, others continued to grow on compost-amended soil, giving rise to a large variation in plant shoot length. On control site soil, naturally colonised plants grew (Friedman test,  $\chi^2(2) = 6.00, p = 0.07$ ) from a median length of 2.0 cm to 9.7 cm during the period from mid-June to mid-August, 2015 (Figure 19).

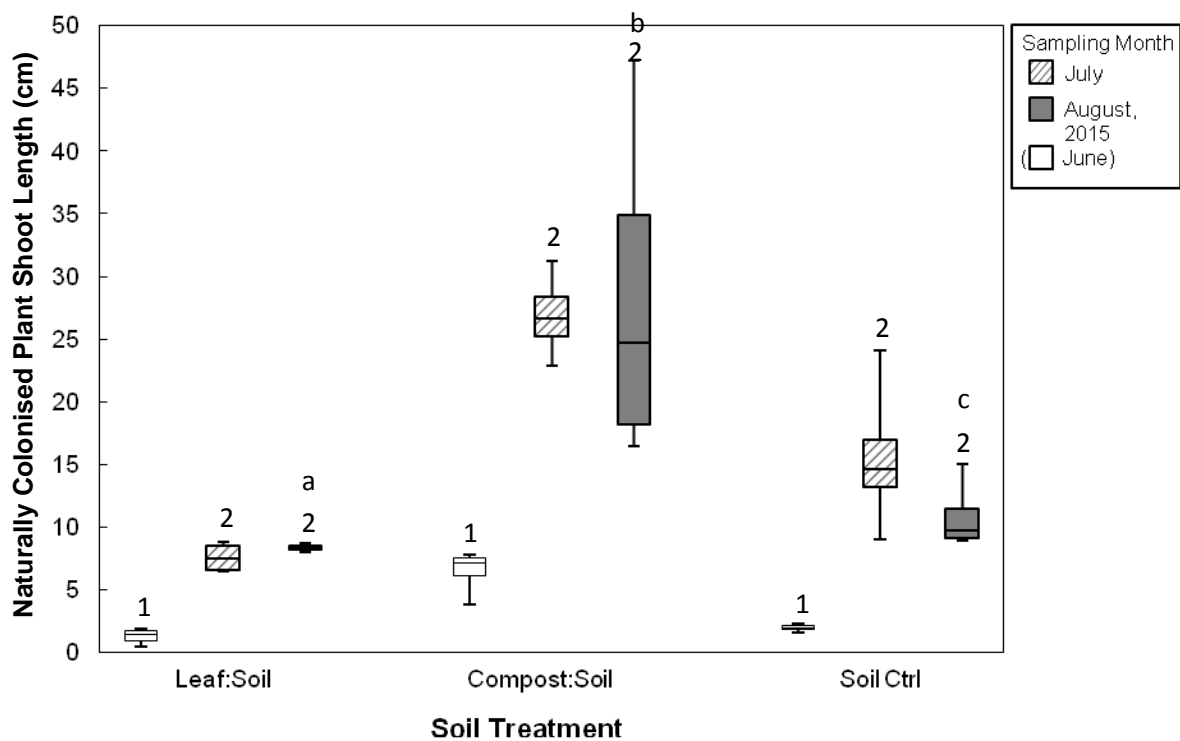


Figure 19. Comparison of the shoot lengths of naturally colonised plants in summer 2015, two and three months after initial site preparation on experimental plot soils amended with 50% (v/v) deciduous leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) ( $n = 4$ ). Note: June shoot length data is carried from Figure 15) to here for convenience of comparison. Statistically significant differences in plant response on different soils in August 2015 are indicated with different letters, and within each soil treatment, differences between months are indicated with different number above boxplots.

### 3.3.3 Characteristics and Diversity of Naturally Colonised Plants in July and August 2015

A total of 16 colonising plant species were observed during the random-transect plant surveys in July and August, 2015. Other than hairy crabgrass (*Digitaria sanguinalis*), all of the species were broadleaf plants. Half of these species were exotic, plus the naturalised species wormseed mustard (*Erysimum*

*cheiranthoides*); the rest (40%) of the species were native. The majority of the naturally colonised plant species (ca. 70%) were annual, with the rest perennial. In general, most of the naturally colonised plants in summer 2015 were the following species: henbit (*Lamium amplexicaule*), Canada thistles (*Cirsium arvense*), yellow clover (*Trifolium agrarium*), and field pennycress (*Thlaspi arvense*), followed by prostrate pigweed (*Amaranthus blitoides*), goldenrods (*Solidago spp.*), nodding smartweed (*Polygonum lapathifolium*), common ragweed (*Ambrosia artemisiifolia*), and lamb's quarter (*Chenopodium album L.*) in lower numbers, as well as ox-eye daisy (*Leucanthemum vulgare*), shepherd's purse (*Capsella bursa-pastoris*), and hairy crabgrass (*Digitaria sanguinalis*) in much lower numbers.

The most abundant species of naturally colonised plant differed slightly on soils with different amendment agents. In general, the top five most abundant species of each soil treatment was as follows: henbit (40% [mean;  $n = 4$ ]) > yellow clover (34%) > Canada thistle (29%) > field pennycress (10%) > nodding smartweed = early goldenrod (*Solidago juncea*) = and ox-eye daisy (6%) on leaf litter-amended soil, Canada thistle (48%) > henbit (24%) > field pennycress (19%) > prostrate pigweed (16%) > yellow clover (14%) on compost-amended soil, and yellow clover (41%) > henbit (39%) > field pennycress (26%) > Canada thistle (23%) > early goldenrod (13%) on control site soil. Almost all of the five most abundant naturally colonised plant species in each treatment were exotic to southern Ontario. Within the same soil treatment, while some species were presented in certain experimental plots, other species emerged in other plots, giving rise to a total of 12 to 13 species per soil treatment (Table 3).

Table 3. The average relative abundance and ecological characteristics of each naturally colonised plant species observed on the barren soil within experimental plots of different soil treatments ( $n = 4$ ; Leaf:Soil, Compost:Soil, and Soil Ctrl) in July and August 2015.

Species of Naturally Colonised Plants		Average Relative Abundance (%; $\pm$ SD)		
Name (in alphabetical order)	Ecological* Characteristics	Leaf:Soil	Compost:Soil	Soil Ctrl
amaranth, redroot ( <i>Amaranthus retroflexus</i> )	native, annual	—	1% ( $\pm$ 3)	—
clover, yellow ( <i>Trifolium agrarium</i> )	exotic, annual	34% ( $\pm$ 12) ②	14% ( $\pm$ 11) ⑤	41% ( $\pm$ 24) ①
crabgrass, hairy ( <i>Digitaria sanguinalis</i> )	exotic, annual	3% ( $\pm$ 5)	2% ( $\pm$ 3)	3% ( $\pm$ 6)
daisy, ox-eye ( <i>Leucanthemum vulgare</i> )	exotic, perennial	6% ( $\pm$ 8) ⑤	1% ( $\pm$ 3)	3% ( $\pm$ 5)
goldenrod, early ( <i>Solidago juncea</i> )	native, perennial	6% ( $\pm$ 4) ⑤	1% ( $\pm$ 1)	13% ( $\pm$ 14) ⑤
goldenrod, sweet ( <i>Solidago odora</i> )	native, perennial	—	2% ( $\pm$ 4)	—
henbit ( <i>Lamium amplexicaule</i> )	exotic, annual	40% ( $\pm$ 11) ①	24% ( $\pm$ 7) ②	39% ( $\pm$ 16) ②

lamb's quarter ( <i>Chenopodium album</i> )	native, annual	3% ( $\pm 3$ )	6% ( $\pm 9$ )	2% ( $\pm 3$ )
mustard, wormseed ( <i>Erysimum cheiranthoides</i> )	naturalised, annual/biannual	2% ( $\pm 3$ )	—	1% ( $\pm 1$ )
pennycress, field ( <i>Thlaspi arvense</i> )	exotic, annual	10% ( $\pm 3$ ) ④	19% ( $\pm 4$ ) ③	26% ( $\pm 23$ ) ③
pigweed, prostrate ( <i>Amaranthus albus</i> )	exotic, annual	5% ( $\pm 6$ )	16% ( $\pm 5$ ) ④	6% ( $\pm 9$ )
ragweed, common ( <i>Ambrosia artemisiifolia</i> )	native, annual	—	7% ( $\pm 14$ )	1% ( $\pm 1$ )
shepherd's purse ( <i>Capsella bursa-pastoris</i> )	exotic, annual	3% ( $\pm 4$ )	—	2% ( $\pm 4$ )
smartweed, nodding ( <i>Polygonum lapathifolium</i> )	native, annual	6% ( $\pm 9$ ) ⑤	—	—
thistle, Canada ( <i>Cirsium arvense</i> )	exotic, perennial	29% ( $\pm 24$ ) ③	48% ( $\pm 16$ ) ①	23% ( $\pm 6$ ) ④
thistle, yellow ( <i>Cirsium horridulum</i> )	native, perennial	—	2% ( $\pm 3$ )	—

Notes:

\* The ecological characteristics of these species in Southern Ontario (Canada) are obtained from the United States Department of Agriculture Natural Resources Conservation Service's (2016) PLANTS Database.

A dash (—) indicates that the species was not present.

An encircled number from 1 to 5 (e.g. ④) indicates the top ranking of the relative abundance of a species among all species observed in experimental plots of that soil treatment.

In terms of species richness, the average number of naturally colonised plant species along the three random plant survey transects per experimental plot did not differ significantly between soil treatment factors in July and August 2015 (one-way ANOVA,  $F(2,9) = 0.36$ ,  $p = 0.71$ ). Specifically, the number of species found along the plant survey transects in each of the four experimental plots with leaf litter- and compost-amended soil was  $6.0 \pm 1.4$  (mean  $\pm$  S.D.;  $n = 4$ ) and  $6.0 \pm 1.6$ , respectively, and in the four control plots with experimental site soil  $5.3 \pm 1.3$  (Figure 20).

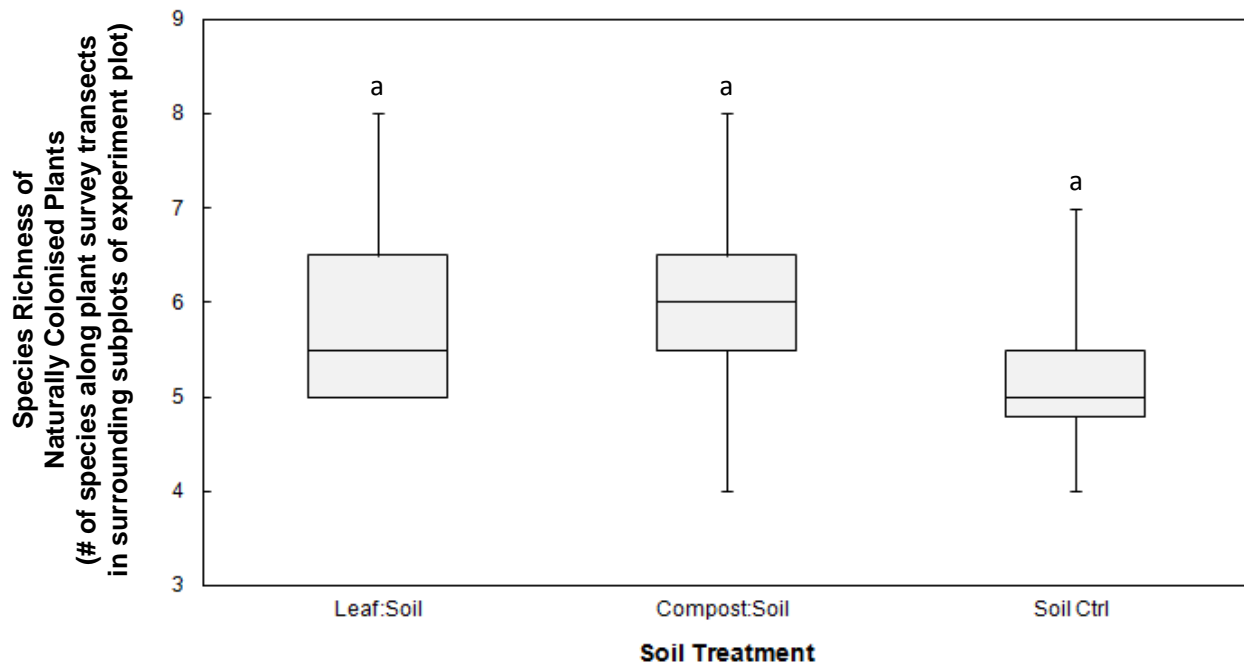


Figure 20. Comparison of the species richness of the naturally colonised plants in the summer (July and August) of 2015 on barren soil of experimental plots, whose soils received 50% (v/v) of leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) in May 2015 ( $n = 4$ ). Note: statistically significant differences in plant response on different soils are indicated with different letters above boxplots.

Similarly, there was no major difference in the distribution of the relative abundance of the naturally colonised plant species between experimental treatments. The most abundant species on leaf litter-treated soil occupied on average 40% ( $n = 4$ ) of the total number of naturally colonised plants and the least abundant species was on average 2% of the total number of naturally colonised plants. The most abundant species represented about 41% of the total number of naturally colonised plants and the least abundant species represented 1% of the total number of naturally colonised plants on the control site soil. However, the distribution pattern of the average abundances of different species was less even on plots with compost treatment: the most and least abundant species represented 48% and 1% of the total number of naturally colonised plants, respectively. Canada thistle was the most abundant species on compost-amended soil and had greater relative abundance than the most abundant weed species on other types of soil. Of the total of 12 to 13 species found within all experimental plots, at least half of the species alone represented less than circa 7% of the total number of naturally colonised plants per plot, indicating the dominance of the soil by a few species of naturally colonised plants (Figure 21, Table 3).



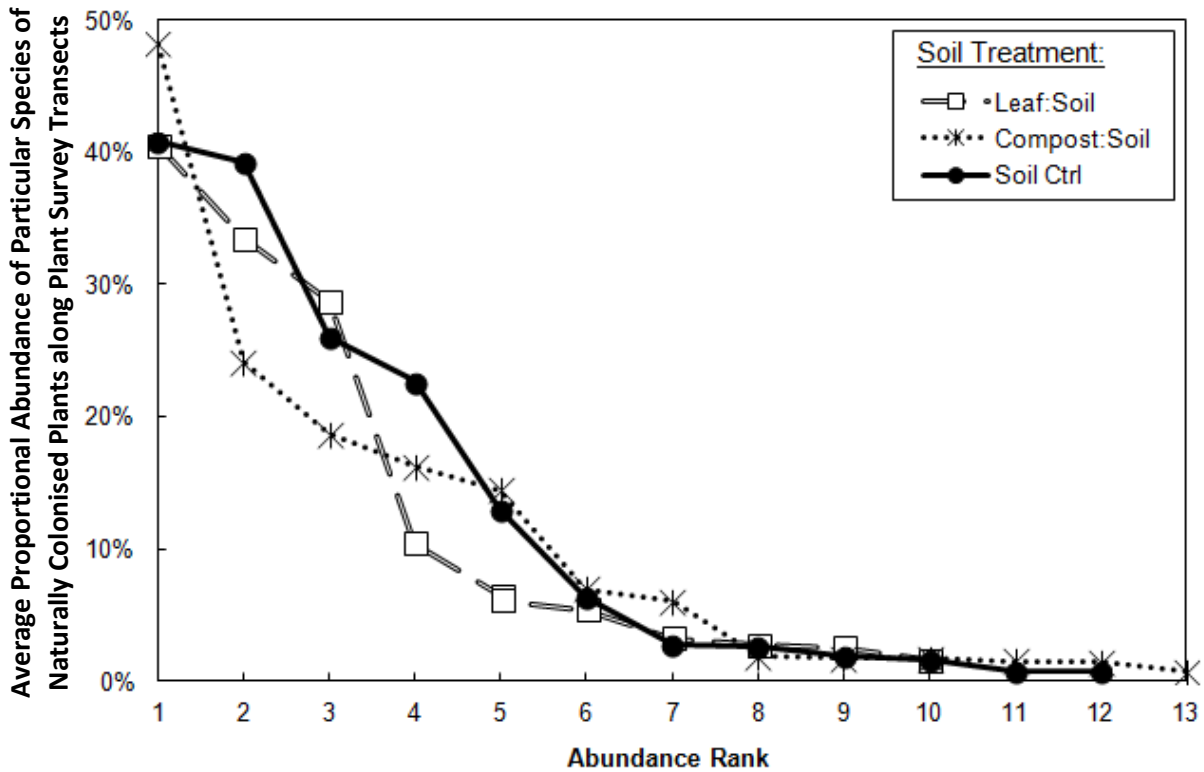


Figure 21. Comparison of the distribution of the average relative abundances of naturally colonised plants species in the summer (July and August) of 2015 on the barren surface of experimental plots with different or no soil treatments (i.e. treatment with 50% (v/v) leaf litter [Leaf:Soil], compost [Compost:Soil], or none [Soil Ctrl] in May 2015) ( $n = 4$ ).

Taking species richness and relative abundances together into consideration, the Shannon-Wiener Diversity Index ( $H'_w$ ) of the naturally colonised plants in each experimental plot with or without soil treatment did not differ significantly in summer 2015 (one-way ANOVA,  $F(2,9) = 2.39$ ,  $p = 0.15$ ). The  $H'_w$ 's of the four experimental plots received compost treatment was insignificantly higher ( $1.635 \pm 0.234$ ; [mean  $\pm$  S.D.;  $n = 4$ ]) than those of the control experimental plots ( $1.355 \pm 0.171$ ) and those with leaf litter treatment ( $1.445 \pm 0.136$ ) because of a total of 13 instead of 12 species of naturally colonised plants were found on compost-amended soil. The  $H'_w$  between leaf-litter treatment plots and control plots were also not significantly different (Figure 22).

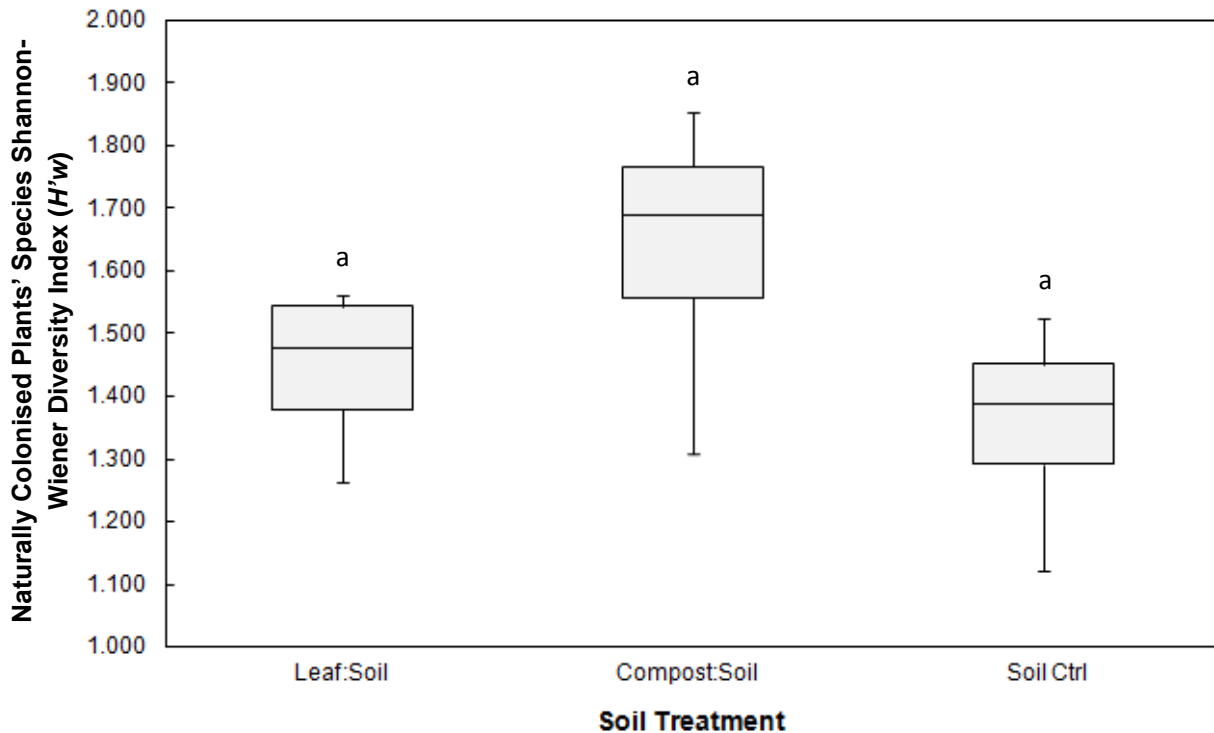


Figure 22. Comparison of the Shannon-Wiener Species Diversity of naturally colonised plants in the summer (July and August) of 2015 on the barren soil of experimental plots, whose soils received 50% (v/v) of leaf litter (Leaf:Soil), compost (Compost:Soil), or none (Soil Ctrl) in May 2015 ( $n = 4$ ). Note: statistically significant differences in plant response on different soils are indicated with different letters above boxplots.

### 3.4 Other Observations

Besides formal soil sampling, testing and plant surveys taking place during the experiment, other informal notes include photos of the experimental sites were taken (Appendix II). From June to August, small creatures were occasionally encountered during sampling visits and random site check-ups. For instance, a frog was seen several times hiding among the tall and dense plants in one of the experimental plots with compost-amended soil. At other times, butterflies were seen on the naturally colonised flowers in the compost treatment plots. No visits by any animals were ever observed on the leaf litter-treated soil and in the control experimental plots. In general, the surface soil in the compost-treated experimental plots was moist and had water puddles after rainfall, while the surface soil in experimental plots with leaf litter treatment and without treatment were dry. Over time, there was thick cover of exotic plants on the compost-amended plots and the exotic plant leaf litter was abundant on the floor of compost-amended plots as the plants senesced and died.

## 4.0 Discussion

### 4.1 Effects of Different Plant Residues on Soil Physical and Chemical Properties

Observed from the current experiment, plant residue compost dramatically changed the physical and chemical properties of the topsoil following soil incorporation. With dramatically higher moisture content, lower bulk density, higher organic matter content, and higher Ex-N, P, and K contents than the receiving soil (i.e. the experimental site soil), the compost consistently and dramatically reduced the bulk density and increased the moisture content, SOM, and plant nutrient levels of the receiving topsoil. However, plant residue compost with higher pH than the receiving topsoil in this case dramatically reduced the pH of the receiving soil. The inability to predict receiving soil pH from the compost pH might have to do with the production of carbonic acid following aerobic decomposition of organic matter from the compost (Sevilla-Perea and Mingorance 2015, Larney and Angers 2012, Pickering and Shepherd 2000).

The effects of plant residue compost on topsoil properties from the current experiment were generally consistent with past observations. After a one-time incorporation of plant waste compost made from leaves and branches into saline coastal soil, Wang *et al.* (2014) observed reduction of receiving soil bulk density, elevation of receiving soil organic carbon content, and significant increase in receiving soil moisture and NPK contents through repeated measurements a few months and one and two year after soil treatment. However, different from the current experimental observation, the authors saw increased soil pH following the application of basic pH compost into neutral soil. Similarly, Liu *et al.* (2012) found elevated soil pH, soil water content, organic carbon, and total NPK within four months after incorporating alkaline tree residue compost into slightly acidic soil (pH = 6-7). At 25% (v/v) rate of yard waste compost incorporation, Curtis and Claassen (2009) observed reduced soil bulk density, increased soil C and N, and greater soil water holding capacity, hydraulic conductivity, and other plant nutrient contents. With lower rate of soil incorporation, the authors achieved almost the same factor of bulk density reduction as the current experiment and much greater factors of soil C and N elevations. Applying compost made from yard trimming and wood residues, Kowaljowa and Mazzarino (2007) found increase in soil C, N, and P. The consistency with past observations on the effects of plant residue composts reinforced the current understanding of the ability of plant residue compost on reducing soil compaction as well as increasing soil moisture, organic matter and plant nutrients. However, compost pH did not appear to be a good predictor of soil pH after amendment agent incorporation (Pickering and Shepherd 2000), suggesting that extra carefulness may be needed when using compost to restore soil pH at least in the short-term.

Actually, the effects of plant residue compost on soil properties from this experiment were congruent with the effects of other types of compost on soil properties. Incorporating neutral pH compost made from sewage sludge plus agriculture plant residues, Sevilla-Perea and Mingorance (2015) observed a reduction in the pH of alkaline soil (pH = 8.6) while more than ten times increase in soil phosphate, nitrate, and several times increase in soil organic carbon and potassium fifteen months after soil treatment. Working with cow manure compost, Loper *et al.* (2010) tilled the compost into soil to a depth of 15 cm and saw reduced soil bulk density, higher soil SOM, phosphate, and potassium levels; adding compost with lower pH than the receiving soil reduced the receiving soil pH in this case. The incorporation of municipal solid waste composts made from food and garden residues also reduced soil bulk density and increased soil organic carbon, Ex-P, total nitrogen, and water infiltration capacity (Civeira 2010). Other studies documenting the effects of compost on soil bulk density reduction as well as soil moisture, SOM and nutrient elevation in the recent time include Evanylo *et al.* (2016), Ibrahim *et al.* (2015), Zamani *et al.* (2015), Belyuchenko and Antonenko (2014) and Waterhouse *et al.* (2014). In addition to a large body of literature documenting uniform effects of compost on soil physical and nutrient effects (Larney and Angers 2012), observations from this experiment further confirmed the capability of compost to increase soil nutrient stock and moisture while loosening the texture. Compost-amended soil with low bulk density, high moisture and nutrient contents and neutral pH has been recognised as ideal for plant survival and growth (Ondono *et al.* 2015, Haan *et al.* 2012).

However, incorporating a large volume of deciduous leaf litter into soil had little to no effects on variables of the receiving soil measured in the current experiment. This was despite the leaf litter having lower bulk density and higher moisture and organic matter content than the compost. The incorporation of acidic leaf litter had no effects on receiving topsoil pH in the current experiment. With very low readily extractable N, P, and K from the leaf litter, the receiving topsoil only had moderate increase in soil Ex-P over time. Thus, the current experiment found that the incorporation of deciduous leaf litter into topsoil could not achieve immediate noticeable modification of soil physical and chemical properties.

Despite very high rate of incorporation, the lack of large fraction of decomposed organic matter probably explained the lack of short-term effects on soil quality by the leaf litter (Environment Canada 2013: pp.3-1 and 3-15, Larney and Angers 2012). Little soluble nutrients that were released from the un-decomposed leaf litter organic matter matrix could not increase the levels of nutrients in the receiving soil. Without sufficient decomposition, the acidity of leaf litter material could not quickly modify the receiving soil pH. Although the leaf litter had high total moisture content, the surface of the leaves was dry – the lack of leaf surface moisture could have slowed down microbial decomposition of the leaf litter (Mukhopadhyay and Joy 2010). The surface of the un-decomposed oak leaves could still have waxy

coating to cause high soil water repellency (Cesarano *et al.* 2016), meaning the inability of fresh leaves to maintain soil moisture. In addition, the planar geometry of deciduous tree leaves did not allow easy homogenous mixing of the leaf litter with the receiving soil aggregates, possibly reducing the surface area for contact with soil microbial decomposers and the ability to physically alter soil texture and therefore soil bulk density. As a possible sign of leaf decomposition, increasing Ex-P content in the leaf litter-amended soil over the course of summer from the current experiment might suggest slow ramification of the effect of plant materials on soil properties.

The lack of short-term visible change in soil properties following leaf litter incorporation from the current experiment was consistent with other studies on direct use of plant material for soil alteration. Only through long-term decomposition of fresh organic matter were the effects of plant material on receiving soil visible (Owen *et al.* 1999). Incorporating plant litter continuously into soil, Burke *et al.* (1995) observed slow gradual increase in SOM after 5 to 10 years. Likewise, only with long-term annual course woody debris input, Goldin and Hutchinson (2013) observed increase in soil C, nitrate, phosphate, and decrease in soil pH in Eucalyptic forest floor soil in Australia. Similar to the lower Ex-N content of leaf litter-amended soil than the control in the current experiment, Baer *et al.* (2003) observed a short-term decrease in the soil content of plant-available nitrogen after adding sawdust to soil. With a C:N ratio higher than 30 for oak and beech leaves (Liu *et al.* 2016), very little supply of bioavailable nitrogen resulted in slow decomposition of the leaf mixture in soil and negligible short-term effects of the leaves on soil properties and nutrient supply following soil incorporation (Nguyen and Marschner 2016). Likely due to the chemical effects of phenolic compounds released from plant matter on nitrogen compounds and the uptake of nutrients in the soil matrix by soil microorganisms, the addition of plant material could not increase the supply of bioavailable forms of nitrogen for plant uptakes (Alexander and Arthur 2014, Rinkes *et al.* 2014).

## **4.2 Effects of Soil Alterations by Plant Residues on Plant Establishment**

### **4.2.1 General Effects of Soil Alterations on Plants**

Both the seeded native plants and naturally colonised plants on leaf litter-amended soil had shorter shoot lengths over the season than the control in the current experiment, suggesting that leaf litter amendment of soil might not increase plant growth in the short-term. The higher numbers of germinated seeded native plants and naturally colonised plants and the greater shoot lengths of the native seedlings on leaf litter-amended soil in mid-June, 2015 could be attributed to the release of small amount of plant nutrients following the initiation of leaf litter decomposition in soil (Rinkes *et al.* 2014). The same or lower number

of naturally colonised plants on leaf litter-amended soil than on the un-amended experimental site soil suggested that deciduous leaves could have negative effects on the establishment of certain plant species.

The short-term effects of unprocessed plant residues on plant growths from the current experiment have also been documented in the literature. For instance, Venner *et al.* (2011) found that paper birch tree seedling did not grow well on saw dust growth substrate; only with nitrogen fertiliser addition was tree seedling germination and growth on woody debris possible. In a microcosm experiment with the incorporation of ground pine, oak, and huckleberry leaves, pitch pine seedling grew less on the leaf-litter amended soil than on the control soil (Jonsson *et al.* 2006). Perhaps due to the release of phenolic compounds from the decomposing plant material (Alexander and Arthur 2014), maize plants grown in soil rich in plant-extracted phenolic compounds suffered from nutrient deficiency (Kraus *et al.* 2003). Other studies also found that plants did not grow well in medium with increasing concentration of plant-extracted phenolic substances (Li *et al.* 2010), suggesting that it would be interesting to further investigate the causal connections between plant residues, their phenolic compounds, and other soil factors on plant growth.

Different from the effects of leaf litter on plants, the number and shoot lengths of both native seeded plants and naturally colonised plants on compost-amended soil were consistently greater than those on the un-amended soil and on the leaf litter-amended soil throughout the growth season in the current experiment. With abundant nutrient supply from a moist growth matrix with loose texture, compost was shown to facilitate plant establishment.

Among the literature documenting past observations on the effects of composts on plant growth, Wang *et al.* (2014) increased pagoda tree growth by incorporating plant residue compost into soil to improve soil nutrients, SOM and moisture contents. With improvement of soil N and neutralisation of soil pH by compost addition, white cabbage yield increased (Brito *et al.* 2013). As compost addition increased receiving soil organic content and total nitrogen and phosphorus contents, there was greater rate of Chinese mustard seed germination and seedling shoot and root growth (Novo and Gonzalez 2014). Following repeated application of plant waste compost, higher soil organic content and porosity plus lower soil bulk density corresponded to higher yield of summer maize and winter wheat (Xin *et al.* 2016).

Similarly, to establish turfgrass on urban soil, one-time incorporation of compost into topsoil increased soil C, N, P, and K and decreased soil bulk density, which lead to greater growth of tall fescue (*Festuca arundinacea*), rye grass (*Lolium perenne*), and Kentucky bluegrass (*Poa pratensis*) (Evanylo *et al.* 2016). In Buenos Aires, Argentina, experimental addition of municipal solid waste compost led to lower soil bulk density and improved water infiltration capacity, soil organic carbon content, and total N

and Ex-P, in turn, corresponded to greater growth of Kentucky bluegrass (Civeira 2010). In other attempts to grow grass species, short-term application of yard waste compost by Hough-Snee *et al.* (2011) increased soil nutrients and resulted in better growth of species such as Kentucky bluegrass, Canada blue grass (*Poa compressa*), and fescue (*Schedonorus arundinaceus*); Loper *et al.* (2010) also observed increase in St. Augustine grass (*Stenotaphrum secundatum*) dry mass and tissue N and P contents in connection with improvements in soil bulk density, field capacity, SOM and nutrients by cow manure compost. Thus, compost amendment of soil appears to facilitate the establishment of grass species.

#### **4.2.2 Specific Effects of Soil Alterations on Seeded Native Plants**

With significantly higher plant numbers than on un-modified experimental site soil throughout the growth season, the modification of soil by either plant residue compost or leaf litter corresponded to successful establishment of seeded native plant groundcover in the current experiment. There was an increase in number of seeded native plants on compost-amended soil over time. Such increase suggested the positive effect of compost soil on native herbaceous plant density. Moreover, the shoot lengths of native plants on compost-amended soil were consistently the highest among all experimental treatments. This further pointed out that compost amendment of soil, while beneficial in general to many types of plants, may also favour the restoration of tall growth native meadow plants. Consistent with inhibitory effect of unprocessed plant residues on plants, the dwarfed growth of the native plants on leaf litter-amended soil signified the lack of immediate effect in regenerating tall growth native herbaceous plants by directly using plant residues in the current experiment.

In fact, past experiments with the addition of unprocessed plant residues or reduction of soil nitrogen generally confirmed the observed effects of soil modification with leaf litter on native meadow species in this experiment. Adding sawdust into soil, big bluestem and switchgrass had lower aboveground net primary productivity (Doll *et al.* 2011). In a tallgrass prairie restoration experiment, the addition of sawdust to soil resulted in reduction of soil N mineralisation and inhibited the growth of grass species including Canada wild rye and big bluestem in the first growth season (Averett *et al.* 2004). In the same study, the sawdust-amended soil inhibited, although to a lesser extent, prairie flowering species. Observation from the current experiment conformed to the authors' observation, as no identifiable mature native flowering species was spotted among the seeded native plants on leaf litter-amended soil over the first season of plant establishment. Adding the carbon source sucrose to soil every two weeks, decreasing content of soil nitrate corresponded to increasing plant cover but reducing aboveground biomass of native species (Clark and Tilman 2010). In contrary, nitrogen fertiliser addition instead of sawdust incorporation into soil resulted in higher productivity of tallgrass prairie vegetations (Baer *et al.* 2003).

With greater nutrient availability and better soil physical and pH conditions for plant growth, the positive effects of plant residue compost on native herbaceous plant establishment and growth in this experiment resonated with past observations on better establishment of native plants under favourable soil conditions. With higher SOM (i.e. humic acid) content, Traversa *et al.* (2014) observed greater germination and growth of switchgrass, a grass species that is native to eastern and mid North America, on plant residue compost-amended soil. Through a correlation analysis, Haan *et al.* (2012) found that herbaceous plants such as black eyed Susan, New England aster, wild bergamot, and hairy beardtongue, a species in the same *Penstemon* genus as the foxglove beardtongue in the current experiment, grew the best on soil with low bulk density and high water availability. The spotting of flowering black eyed Susan plants on one of the experimental plots with compost-amended soil in late August in the current experiment was consistent with Haan *et al.*'s (2012) finding. Incorporating yard waste compost at 25% (v/v), lower soil bulk density, higher soil C, N and water availability correlated with increase native plant (i.e. big squirrel tail [*Elymus multisetus*] and purple bunchgrass [*Nassella pulchura*]) biomass in California (Curtis and Claassen 2009). Similar, Séré *et al.* (2008) saw higher biomass of French indigenous plants grown on compost-amended soil that had closer-to-neutral pH. As another source of evidence on compost-facilitated restoration of plant community, compost alteration of soil properties from the current experiment could be used to initiate the establishment of native plants.

Without soil amendments, the seeded native plants also germinated and grew on un-modified soil in the current experiment, suggesting that soil change may not always be needed for the restoration of native herbaceous plants. For instance, the native grasses tussock grass (*Poa labillardieri*) and red-leg grass (*Bothriochloa macra*) established successfully on barren soil in Australia (Lindsay and Cunningham 2011). As a broad-spectrum agent that promotes plant germination and growth, it is also necessary to pay attention to potential side-effects associated with compost usage in native plant restoration exercises.

#### **4.2.3 Specific Effects of Soil Alterations on the Establishment and Species Characteristics and Diversity of Naturally Colonising Plants**

Similar to successful restoration of native plants, the number and growth of naturally colonising plants, which consisted mostly of exotic species, on compost-amended soil during the entire growth season in the current experiment was often higher than those on un-amended and leaf-amended soil. In other words, by creating plant-favourable soil conditions, compost incorporation into soil could also encouraged exotic plant colonisation. This finding was consistent with the observation of excessive weeds on soil over-fertilised through high rate of (45% (v/v)) municipal plant waste compost application (Sparke *et al.* 2011) and the wide presence of annual weeds on disturbed soil with high nitrogen content (Perry *et al.* 2010).



However, with lower number and dwarfed growth of naturally colonised plants in leaf litter-amended plots into the growth season, the current experiment showed that incorporating deciduous leaf litter into barren soil can deter exotic plants at least in the short-term.

The most dominant species of naturally colonised plants in the current experiment were exotic species that are commonly found on open fields and disturbed soils. Originated from Eurasia, Canada thistle is an aggressive invasive weed that is commonly found on abandoned grounds and barren soil (Swearingen *et al.* 2010: p.37). Brought from Eurasia and North Africa, henbit is often found on abandoned soils such as roadsides, fields, and wastelands (Linn and Linn 1978: p.11). As an agricultural winter crop (Fan *et al.* 2013), the exotic weed field pennycress is often associated with vacant lots and disturbed soil (Newcomb 1977: p.136). Originated from Europe, yellow clover is widely found along roads and on open land (Linn and Linn 1978: p.40). Within the *Amaranthus* genus, prostrate pigweed is a common weed on abandoned land (Newcomb 1977: p.414). Open disturbed fields and ground are also the habitat for the other less common weeds in the current experiment (e.g. Newcomb 1977: pp. 150, 414, 422, 430, 438). As soil disturbance and vegetation clearance opened up soil surface and brought in greater amount of solar radiation, the process could have facilitated the widespread colonisation of the exotic weeds on barren ground (Clark and Tilman 2010). This facilitation process could explain the lack of significant difference in the species composition pattern (i.e. species richness, relative abundance, and overall  $H'w$ ) of the naturally colonised plants on open soils with and without amendment agents in this experiment. In other words, the current experiment showed that colonisation of barren soil by wild plants in the urban environment was not able to effectively restore native plants and could not preferentially restore specific plant species.

As a restoration technique that can save the cost associated with seeding, wild plant colonisation of barren ground worked in the current case to facilitate the establishment of exotic species. Consistent with current experiment, past experiences showed that open barren ground aided the germination and growth of Canada thistle (Moore 1975) and field pennycress (Warwick *et al.* 2002, Blackshaw *et al.* 1994). On disturbed land surface with open canopy, Von Holle and Motzkin (2007) found greater number of non-native plants in southern New England (USA). In Colorado (USA), Bernstein *et al.* (2014) saw greater establishment of exotic weeds on disturbed soil in the first year.

As there was even greater number and growth of naturally colonised plants on compost-amended soil, it meant that short-term soil nutrient and structure improvement could further strengthen the dominance of fields by exotic weeds. Similar to observations from compost-amended soil in the current

experiment, high soil moisture promoted the seed germination of field pennycress (Royo-Esnal *et al.* 2015, Blackshaw *et al.* 2002) and henbit (Hill *et al.* 2014).

Moreover, the compost-lead enrichment of soil nutrient contents can lead to dominance of soil surface by low number of plant species. Similar to the presence of a large proportion of Canada thistle on compost-amended soil in the current experiment, Borden and Black (2011) saw dominance of a single noxious weed species (cheatgrass [*Bromus tectorum*]) on soil enriched with nitrogen from biosolid. With increase in soil fertility in semi-natural ecosystems, land can often be dominated by a few productive species (Marrs 1993), resulting in species diversity loss. On a newly established tallgrass prairie, soil N-enrichment corresponded with high plant productivity but low diversity (Baer *et al.* 2003). In Europe, there was a negative relationship between high soil nitrogen content and plant species richness on European grasslands; but rising soil phosphorus could have also accounted for the loss of plant biodiversity (Ceulemans *et al.* 2014, Ceulemans *et al.* 2011). In the current experiment, Ex-N content of compost-amended soil decreased sharply between the first and the second month after soil modification while Ex-P appeared to increase. This could potentially mean that high soil nitrogen content from compost was responsible for encouraging Canada thistle dominance. The restoration of plant community with high species diversity may thus be difficult on compost-amended nutrient-rich soil.

Although the current experiment did not examine the interactive effects between exotic weeds and the seeded native plants, exotic weed establishment might be a threat to the restoration of native plant community. Different plant species have different ways and abilities to compete for nutrients (Tang *et al.* 2014). For example, big bluestems survive well on nutrient-poor soil, where the species could acquire nutrients through mutualistic association with Mycorrhizal fungi (Hetrick *et al.* 1994). However, the addition of extra soil nutrient could reduce the association of the plant species with mycorrhizal fungi (Bever *et al.* 2010), rendering the native plants less competitive than exotic annual weeds on nutrient-rich soil (Perry *et al.* 2010, Standish *et al.* 2006). In an effort to restore native grass, Lindsay and Cunningham (2011) found a positive relationship between soil P level and exotic species cover but a negative one between soil P and native plant cover. With litter fall following quick maturity and seeding, exotic weeds could sustain their plant community by accelerating nutrient cycling (Vinton and Georgen 2006). With high abundance and growth, shade created by exotic weed growth could prevent smaller less competitive plants from acquiring sufficient amount of light (Hautier *et al.* 2009).

Unprocessed plant residues could inhibit exotic weed encroachment while favouring the germination of native grass species (Perry *et al.* 2010). The low number and growth of weed plants but high number of seeded native plants from the current experiment added a piece of evidence to this claim.

Mulching oak and maple leaves into Kentucky bluegrass lawn, Kowalewski *et al.* (2009) saw reduced number of plants of the weed dandelion (*Taraxacum officinale*). At the same time, the authors saw increased number of grass shoots, suggesting that plant materials in soil can also foster grass species germination. Similarly, the addition of sawdust into soil by Averett *et al.* (2004) resulted in high biomass of Canada wild rye and big bluestem and low biomass of exotic weeds. Blumenthal *et al.* (2003) also observed reduced biomass of colonising weeds but increased biomass of fourteen native grass species that included big bluestem, Canada wild rye, and switchgrass after tilling sawdust into soil. Therefore, the application of deciduous leaf litter may help to establish native plant ground cover.

#### **4.3 Implications of the Different Short-Term Effects of Different Plant Residues for Native Plant Restoration and Plant Waste Management**

At the first sight, the application of plant residue compost and deciduous leaf litter each appeared to give rise to “dilemmas” between soil improvement and native plant restoration. From the current short-term experiment and literature review, improvement of soil physical and chemical by compost could favour not only the establishment of native plants but also the proliferation of dominant exotic weed species. On the opposite side, leaf litter amendment of soil did not bring immediate change to commonly measured soil properties but decreased the growth of exotic weeds. While leaf litter-amended soil did not give rise to tall growth of native plants in the current short-term experiment, the soil alteration appeared to foster native herbaceous ground cover with high plant density.

This apparent dilemma is not unique to the use of deciduous leaves and plant residue compost for plant restoration. For example, the addition of biochar made from plant residues could increase plant yield, but it might also decrease soil available N to negatively affect nutrient-poor soil (Atkinson *et al.* 2010). Volatile organic matter from plant residue biochar might even be toxic to certain plant species (e.g. Deenik *et al.* 2010). Thus, finding a “perfect” soil amendment agent is impossible when the goal of restoration is vaguely defined (Saebo and Ferrini 2006).

Each native plant restoration project would take place under certain expectations and under certain level of commitment at a specific site (Kimball *et al.* 2015). The specific site would have its unique soil and vegetation conditions. The restoration project may also be tied into the greater social and ecological needs of a city (Pavao-Zuckerman 2008). Suiting the unique set of needs associated with each restoration project, restoration ecologists can make use of the unique short-term effects of different types of plant residues on soil properties to initiate differently the restoration of native plants.

Making use of the deciduous leaf litter for native urban landscaping practices can reduce the cost associated with leaf litter disposal and composting. For instance, by making use of the initial dwarfed dense growth of native grasses following leaf litter soil modification, it might be possible to start the generation of a native lawn in urban parks. With confirmation and betterment of this potential leaf-litter native lawn technology, the adoption of slow-growth native ground cover may reduce the environmental and economic cost of landscape maintenance in urban parks (Asah *et al.* 2014). Meanwhile, such landscape feature may increase urban residents' awareness and appreciation for ecosystem services provided by native plants. This can then gear greater support for local natural heritage protection and future native landscaping practices in urban parks (Standish *et al.* 2013, Holl and Aide 2011). When applied continuously onto soil or existing lawns, the leaf litter may be used as a low-cost method to prevent exotic weeds on urban soil (Kowalewski *et al.* 2009). Besides facilitating the establishment of native grasses and deterrence of exotic weeds, continuous addition of leaf litter can also supply soil nutrients through gradual decomposition (Chavez-Vergara *et al.* 2014). Thus, leaf litter should be considered to be part of low-maintenance strategies to adjust urban soil and to initiate the restoration of native plant communities.

To generate tall growth native herbaceous plants on nutrient-poor soil, compost can be an effective agent to quickly restore native plants. Composting should continue to be practiced for recycling plant residues to improve soil quality and to increase plant germination and growth. The restoration of plant communities can be a market for this recycling industry. With quick establishment of native plants on compost-amended soil, compost application can help to increase public confidence in native habitat regeneration (Klaus 2013). In parts of urban parks for naturalisation, compost may be applied to soil to restore deliberately seeded native plants in short-time. However, soil nutrient enrichment by compost may not lead to plant communities with high species diversity (Ceulemans *et al.* 2014, Ceulemans *et al.* 2011), suggesting that compost may not be used for plant restoration projects with high requirement for species diversity. Moreover, compost should be used with caution in areas facing high pressure from exotic plant invasions as the material appears to facilitate the growth and dominance of exotic weed species. The dominance of fields by exotic weeds may not provide the aesthetical expectation for urban parks and can lower the support for future ecological restoration (Standish *et al.* 2013). But under cases where soil heavy metal remediation is needed (Park *et al.* 2011, Farrell *et al.* 2010), the presence of exotic weeds might be an inevitable part of the initial stage of land rehabilitation using compost.

Depending on site soil conditions, the application of plant residues may not always be necessary to restore native plants. The current experiment showed that deliberately seeded native plants may germinate and grow well on un-amended soil. In other words, the quality of the site soil was sufficient for

initiating native plant restoration. Under such circumstance, high seeding rate into the soil alone may regenerate native groundcover (Clark and Tilman 2010) and become the local source for native seeds (Hopwood 2013). Thinking about the purpose of prospective soil modification in each individual restoration case, soil structural improvement and nutrient enrichment may not always benefit native plant restoration (Larney and Angers 2012, Perry *et al.* 2010, Standish *et al.* 2006). Intense human involvement may not always render positive restoration outcomes (Prach and del Moral 2015, Hilderbrand *et al.* 2005).

Actually, the need for human facilitation in restoring native plants can be decided by considering several restoration site-specific factors (Prach and Hobbs 2008). At sites with high degree of soil disturbance, such as vegetation stripping and soil tillage in the current experiment, active human planting should occupy the soil surface with desired native species so to avoid the dominance of exotic weeds. This also means that complete elimination of existing vegetation cover may not be the best way to start the restoration of desired plants. Instead, it might be worth to try if planting desired plants or drilling seeds into existing lawn surfaces can gradually restore native ground cover (Suding 2011). After disturbance due to exotic plant removal, native planting or deciduous leaf litter amendment should immediately follow to prevent exotic weed colonisation of exposed soil. Upon improvement of soil conditions that would favour plant establishment, native planting or continuous addition of leaf litter should proceed to prevent exotic plant colonisation. The nitrogen content of some urban soil may already be high due to industrial and transportation activities, which can facilitate exotic weed establishment (Vitousek *et al.* 1997). There, native planting plus leaf litter application may help to counteract this negative effect. When dealing with soil of extreme low nutrient contents, poor physical structure (Larney and Angers 2012), and/or heavy metal contamination (Park *et al.* 2011, Farrell *et al.* 2010), compost amendment should still be considered to aid native plant establishment. No manipulation of soil conditions would be needed if native plants have already been shown to establish adequately. When the size of soil seed bank of desired native plants is small or non-existent, intensive seeding efforts should be considered in restoring urban native habitat.

The decision on human involvement in native plant restoration also depends on the environment surrounding the restoration site (Prach *et al.* 2015, Holl and Aide 2011). With the presence of sufficient amount of native propagules in close proximity of the restoration site, Prach *et al.* (2013) documented the restoration of disturbed sites by random seed colonisation in Czech Republic. Similarly, the presence of desired plant species in the vicinity of restoration sites could effectively restore plant communities in France (Forget *et al.* 2012, Muller *et al.* 1998). However, when propagules of exotic species dominate the surrounding of a restoration site, passive restoration by weed colonisation cannot restore native plants even after manipulating the soil substrate (Prach *et al.* 2015, Prach *et al.* 2014, Hobbs and Cramer 2008).

This means that seeding desired native plants would be needed to restore native plants in high disturbed environment such as cities, where native propagules are unlikely to be found in the vicinity of restoration sites due to the lack of intact natural habitat and/or the presence of invasive species.

Despite the technical discussions on native plant restoration so far, there will certainly be cases where the social and ecological cost of restoration at an urban site would be too high to prevent native plant restoration. Financially, it may be too costly for native restoration at an urban site, where the site and its surrounding have been altered to render the establishment of desired native species very difficult (Prach and Hobbs 2008). Time-wise, permanent habitat modification in cities can mean that decades are needed to meet native plant restoration goals (Jones and Schmitz 2009). Socially, depending on the particularity of each case, Gobster (2007) found that “valueless” weedy and barren urban parklands offered recreational opportunities for residents in some neighbourhoods in Chicago and San Francisco; for example, children could once interact with nature hands-on before ecological restoration activities (i.e. playing with the dirt, flowers, plants, and stones). After native species restoration, the author found that park users could only passively view the restored landscape behind fences, which in turn, increased urban residents’ disapproval for native plant restoration. Similarly, some patches of lawns in parks may still provide important space for informal sport activities in cities. Although the sighting of small animals interacting with exotic weeds on compost-enriched soil was not part of a formal systematic assessment at the current experimental site, it hints that more research is needed to evaluate the ecological and social costs and benefits of ecosystems with exotic species versus those of native habitats (Hobbs *et al.* 2014, Standish *et al.* 2013). Taking explicitly human needs into consideration, soil improvement and native plant restoration should not be taken as the default options for urban improvement or gentrification (Rankin and McLean 2014).

#### **4.4 Future Research**

The restoration of native herbaceous plants is really an open-ended question, given that ecological restoration is always context-specific (Suding 2011). The current research only gave a short-term observation of how different types of plant residues could be potentially used to initiate urban native plant restoration. On one hand, future research around the application of plant residues should continue to develop the technical capacity to restore different types of native landscapes. On the other hand, future works on urban ecological restoration need to engage with stakeholders and the general public to use this technical capacity to achieve desirable restoration outcomes.

First, there should be more rigorous research to confirm the restoration outcomes from the current experiment. For instance, there should be long-term evidence on the use of deciduous leaf litter for

building functioning low-maintenance native groundcover. To have replicable results, experiments should be carried out in different park land settings under the influence of different urban environments. Experimentation with different intensities of seeding different combinations of native plants under increasing amount of leaf litter application should also be carried out. Similarly, experimentation with ground leaves may render greater effect on soil N-immobilisation (Rinkes *et al.* 2014) and give rise to different seedling emergence and growth phenomena. Instead of focusing on only one growth season, observations over years may give more conclusive results and can allow a species diversity assessment of the restored native grasses and forbs. Whether continuous spread of grinded leaf litter and occasional re-seeding effort can sustain a robust native lawn would be of interest to the development of native landscaping practices. Couple the experiment with soil phenolic content analysis and phenolic phytochemistry research may give a mechanistic insight into the effects of leaf litter on soil and plants (e.g. Kraus *et al.* 2003).

To make the best use of both unprocessed and processed plant residues, it would be interesting to know if combining deciduous tree leaf litter with plant residue compost at different proportions can yield dense and tall native groundcover and simultaneously deter exotic weed especially in the long term. Depending on the production procedure, compost of different qualities could be produced for uses in soil and plant restoration (Guénon and Gros 2015). Similarly, the effect of leaf litter on soil could differ by tree species (Alexander and Arthur 2014) and possibly also by season and source of tree leaves (Reed and McCarthy 1996, Feeny and Bostock 1968) for restoration uses. Besides unprocessed plant residues and plant residue composts, the effects of other types of organic residue products, such as different types of biochars made from plant residues (Atkinson *et al.* 2010), should also be tested singly or in combination. The type of plant material for investigation seems to be infinite. Moreover, different methods and frequencies of soil amendment agent application may also affect the restoration outcome (e.g. Evanylo *et al.* 2016, Dhanya *et al.* 2013, Walsh *et al.* 1996), and these effects over time should be documented for differential applications.

Technically, there are more investigations that can further enhance our knowledge and practice of native plant restoration. Weed seeds can be present in compost (Larney and Blackshaw 2003) and in the soil of an ecological restoration site (Rossiter *et al.* 2014, Holl and Aide 2011). Pay attention to the influence of pre-existing seed banks on restoration outcomes, soil amendment agents and the experimental site soil can be subject to laboratory plant germination assay. Also, there can be tests on the fitness and genetic diversity of the restored plants to inform about their long term persistence (eg. Aavik *et al.* 2014). Progress through technical understanding, successful restoration should always depend on sound science.

Of course, a “successful” restoration project is inherently defined by the needs of its stakeholders (Instone 2015, Standish *et al.* 2013, Yu *et al.* 2012, Gobster 2010, Trigger and Head 2010, Ingram 2008). Based on informal communications with some urban residents, this experiment assumed the common desire to restore native forbs and grasses in parks (Bliss and Fischer 2011, Kettenring and Adams 2011). In other cases, it might be important for the restoration ecologists to engage in a rigorous manner with the relevant decision makers and the general public in parkland restoration projects (Kimball *et al.* 2015, Wiegleb *et al.* 2013, Christoffersen 2011, Newman 2011, Herringshaw *et al.* 2010). Future research in urban ecological restoration can focus more on the following aspects to contribute to social and ecological wellbeing in cities. First, it is necessary to raise the awareness for societal needs among restoration ecologists besides technical expertise (Allison 2012: pp.123, Kimball *et al.* 2015). Second, increase the communication efficiency between restoration ecologists and non-technical stakeholders would be vital (Jorgensen *et al.* 2014). Putting research into practice, research on how to take serious logistical and financial considerations in restoration project designs would also be helpful (Kimball *et al.* 2015). With better integration of both social and ecological needs of cities and better communication with non-technical stakeholders, restoration ecologists can be more active participating members in urban restoration projects and parkland maintenance works (Kimball *et al.* 2015, Holl and Aide 2011).

Lastly, to find avenues for the reuse and recycling of different types of plant residues, collaborative research with organic waste management researchers and practitioners may be needed. Preventing the spread of plant and other types of pathogens as well as unwanted exotic plant propagules should continue to be a top priority for sound reuse of plant residues. Diverting different types of plant residues would be needed to gather sufficient amount of each type of residue for potentially different uses in urban native plant restoration. Considering the observations of the effects of raw plant residue on soil and plant from the current study for potential applications, waste managers may need to think again whether composting would always be needed to join the nutrient cycle between plants and soil. By reusing plant residues, ecological restoration and green waste management can push together for environmental protection, natural resource conservation, and social wellbeing in an ever urbanising world.

## **4.5 Conclusions**

The different short-term effects of deciduous leaf litter and plant residue compost on soil and plants from the current experiment and the literature review showed that not all plant residues should be viewed the same. Different types of plant residues may be used to amend the soil differently to initiate native plant restoration differently in urban ecological restoration projects aimed at different goals.



As a form of unprocessed plant residue, deciduous leaf litter was again shown to be poor in plant-available nutrients and had little short-term effects on commonly measured soil physical and chemical properties (i.e. moisture content, bulk density, SOM, pH, and NPK). Using the possible effect of soil N-immobilisation by deciduous leaf litter, this type of plant residue may be used to establish dense but slow-growing native grass lawn that may require low maintenance in urban parks. As an environmentally friendly alternative to prevent the spread of exotic plants, the leaf litter were shown to deter exotic weeds at least in the short-term after soil addition.

As a form of processed plant residue that was low in bulk density and high in moisture, organic matter, and plant nutrients, soil modified by plant residue compost showed properties similar to those of other types of composts. However, the pH of the receiving soil did not follow that of the compost, suggesting that compost may not be used to easily restore desired soil pH in the short-term. Different from leaf litter-amended soil, compost is a “broad-spectrum” agent that aided the establishment of not only native but also undesired exotic plants, and compost-amendment of soil appeared to cause low plant species diversity.

There is no black-and-white answer to the question on the extent that humans should involve in assisting the recovery of damaged urban ecosystems. The answer to this question is context-dependent. The restoration of native plants with compost should happen at places where pressure from exotic plant invasion is low and when the restoration of high species diversity is not a priority. In fact, soil modification may not always be required to restore native plants – high density seeding on urban soil of adequate nutrient and physical conditions alone may be sufficient to start the restoration of native plant cover. Deliberate seeding of desirable native plants should be necessary for native plant restoration efforts at locations where there is no or little presence of propagules at or around the restoration site. Soil disturbance should be avoided during the application of amendment agents to prevent the facilitation of exotic weed establishment. The lack of significant difference in weed species composition between soil treatments suggested that under high degree of soil disturbance, soil amendment may not favour the establishment of particular plant species over others.

More long-term research on the different effects of different types of plant residues on soil and plants may offer native plant restoration techniques suiting different urban ecological and social needs. Then, the restoration of terrestrial plant communities would have greater potential to reduce net organic waste output from cities by diverting different plant residues for different uses. Although the current research is a short-term observation, the research results suggested that plant materials such as deciduous leaf litter can have its own use in initiating native plant restoration. This should cause organic waste

managers to rethink about the revaluation of plant wastes, not only through compost production and application but also direct usage. Integrating ecological restoration with waste management, the restoration of native plant cover may have the potential to solve multiple ecological and social issues facing cities today.

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## Appendix I Soil and Plant Restoration Experiment Data

This appendix contains the laboratory and field data for the experimental materials and for each experimental plot that were subjected to statistical analyses in the thesis. Please see Figure 3 for soil treatment allocation to the experimental plots and the naming of the experimental plots.

### Part I Characteristics of the Experimental Site Soil and those of the Soil Amendment Agents

Five replicates of each experimental material were analysed in May 2015:

Experimental Material	Moisture (% ww)	Bulk Density (g/cm <sup>3</sup> dw)	Soil Organic Matter (% dw)	pH	Ex-N (kg NO <sub>3</sub> <sup>-</sup> /ha air dried weight)	Ex-P (kg PO <sub>4</sub> <sup>3-</sup> /ha air dried weight)	Ex-K (kg K <sup>+</sup> /ha air dried weight)
Deciduous leaf litter	54.7%	0.0171	92.63%	4.86	< 0.10 d.l.	169.5	204
	44.6%	0.0144	90.66%	4.97	< 0.10 d.l.	136.9	306
	51.9%	0.0183	92.70%	5.00	< 0.10 d.l.	94.6	245
	46.5%	0.0179	90.74%	5.19	< 0.10 d.l.	150.0	459
	57.5%	0.0214	93.51%	5.15	< 0.10 d.l.	120.6	500
Plant residue compost	50.4%	0.295	29.57%	7.77	237.63	560.8	3268
	42.3%	0.316	31.57%	7.75	200.64	603.2	4062
	46.4%	0.3137	29.96%	7.79	331.79	599.9	3595
	41.1%	0.340	8.42%	7.82	223.06	648.8	3688
	45.4%	0.307	38.54%	7.70	175.98	645.6	2848
Experimental site soil	13.3%	0.968	3.05%	7.12	4.48	88.8	73
	11.9%	0.943	3.03%	7.27	6.73	93.3	56
	13.1%	0.877	2.89%	7.35	4.48	29.4	78
	11.9%	0.857	3.12%	7.14	8.97	97.6	62
	12.6%	0.979	3.09%	7.11	7.85	108.7	45

### Part II Soil Physical and Chemical Properties in Response to Soil Treatments

A. Physical properties:

Experimental Plot	Moisture (% ww)	Bulk Density (g/cm <sup>3</sup> dw)
June 3, 2015		
L <sub>1</sub>	14.41%	1.23
L <sub>2</sub>	13.51%	1.17
L <sub>3</sub>	14.54%	1.23
L <sub>4</sub>	15.45%	1.22
C <sub>1</sub>	20.77%	0.945
C <sub>2</sub>	18.39%	0.962
C <sub>3</sub>	18.40%	0.938
C <sub>4</sub>	21.77%	0.943



S <sub>1</sub>	11.85%	1.20
S <sub>2</sub>	12.71%	1.25
S <sub>3</sub>	14.03%	1.29
S <sub>4</sub>	11.92%	1.23
<hr/>		
July 2, 2015		
L <sub>1</sub>	14.02%	1.30
L <sub>2</sub>	16.15%	1.30
L <sub>3</sub>	15.31%	1.32
L <sub>4</sub>	15.18%	1.31
C <sub>1</sub>	23.93%	0.859
C <sub>2</sub>	21.31%	0.991
C <sub>3</sub>	19.32%	1.13
C <sub>4</sub>	22.98%	1.02
S <sub>1</sub>	13.34%	1.29
S <sub>2</sub>	13.69%	1.43
S <sub>3</sub>	13.41%	1.52
S <sub>4</sub>	12.59%	1.40
<hr/>		
August 5, 2015		
L <sub>1</sub>	12.71%	1.24
L <sub>2</sub>	14.40%	1.34
L <sub>3</sub>	14.00%	1.32
L <sub>4</sub>	14.49%	1.31
C <sub>1</sub>	19.84%	1.08
C <sub>2</sub>	19.09%	1.06
C <sub>3</sub>	19.22%	1.10
C <sub>4</sub>	20.50%	0.949
S <sub>1</sub>	12.62%	1.28
S <sub>2</sub>	13.79%	1.30
S <sub>3</sub>	13.31%	1.38
S <sub>4</sub>	13.13%	1.33
<hr/>		

B. Chemical and properties:

<b>Experimental Plot</b>	<b>Soil Organic Matter (% dw)</b>	<b>pH</b>	<b>Ex-N (kg NO<sub>3</sub><sup>-</sup> /ha air dried weight)</b>	<b>Ex-P (kg PO<sub>4</sub><sup>3-</sup> /ha air dried weight)</b>	<b>Ex-K (kg K<sup>+</sup> /ha air dried weight)</b>
<hr/>					
June 3, 2015					
L <sub>1</sub>	2.31%	8.05	3.70	1.39	34
L <sub>2</sub>	2.79%	7.95	2.78	2.11	43
L <sub>3</sub>	2.95%	8.18	3.86	5.11	65
L <sub>4</sub>	4.16%	8.14	4.15	22.64	69
C <sub>1</sub>	10.50%	7.74	11.77	24.08	605
C <sub>2</sub>	9.61%	7.90	10.09	39.03	258

C <sub>3</sub>	10.00%	7.78	10.09	28.23	650
C <sub>4</sub>	7.95%	7.73	10.09	49.82	538
S <sub>1</sub>	2.43%	8.16	3.47	10.09	54
S <sub>2</sub>	2.63%	8.04	4.19	1.08	32
S <sub>3</sub>	2.66%	8.02	3.77	0.85	45
S <sub>4</sub>	2.02%	8.05	3.98	0.76	37
<hr/>					
July 2, 2015					
L <sub>1</sub>	3.78%	7.64	2.92	0.76	75
L <sub>2</sub>	3.43%	7.92	3.03	1.26	49
L <sub>3</sub>	2.65%	7.87	3.61	12.33	77
L <sub>4</sub>	5.69%	7.85	3.39	30.49	99
C <sub>1</sub>	6.74%	7.57	6.16	64.77	712
C <sub>2</sub>	10.77%	7.70	5.04	29.06	577
C <sub>3</sub>	5.67%	7.61	5.04	85.53	398
C <sub>4</sub>	12.78%	7.63	3.92	122.89	532
S <sub>1</sub>	2.35%	7.81	3.49	16.10	64
S <sub>2</sub>	2.58%	7.93	3.26	1.08	65
S <sub>3</sub>	2.73%	7.87	3.72	3.32	52
S <sub>4</sub>	2.04%	7.88	3.52	0.94	41
<hr/>					
August 5, 2015					
L <sub>1</sub>	3.00%	7.96	3.59	2.15	34
L <sub>2</sub>	4.52%	8.02	3.21	1.21	34
L <sub>3</sub>	3.58%	7.95	3.31	36.59	73
L <sub>4</sub>	3.72%	7.95	3.12	51.07	80
C <sub>1</sub>	7.59%	7.85	3.36	79.71	437
C <sub>2</sub>	11.35%	7.85	6.16	45.67	521
C <sub>3</sub>	8.42%	7.76	2.80	93.83	448
C <sub>4</sub>	13.48%	7.73	3.92	118.74	925
S <sub>1</sub>	2.93%	7.96	3.32	25.65	52
S <sub>2</sub>	2.83%	7.97	3.90	1.70	32
S <sub>3</sub>	3.36%	7.89	3.94	6.14	43
S <sub>4</sub>	2.85%	8.00	3.38	0.90	45

### Part III Plant Germination in Response to Soil Treatments in June 2015

June 18 and 19, 2015

A. Native plant seedlings:

Experimental Plot	Number of Seedlings	Shoot Length of Seedlings (cm)
L <sub>1</sub>	55	5.0
L <sub>2</sub>	70	7.4

L <sub>3</sub>	45	5.3
L <sub>4</sub>	63	7.4
C <sub>1</sub>	89	9.0
C <sub>2</sub>	54	8.0
C <sub>3</sub>	57	6.2
C <sub>4</sub>	56	7.6
S <sub>1</sub>	28	2.5
S <sub>2</sub>	39	5.2
S <sub>3</sub>	20	2.6
S <sub>4</sub>	43	3.4

B. Naturally colonised plants:

<b>Experimental Plot</b>	<b>Number of Seedlings</b>	<b>Shoot Length of Seedlings (cm)</b>
L <sub>1</sub>	50	0.5
L <sub>2</sub>	70	1.1
L <sub>3</sub>	52	1.9
L <sub>4</sub>	75	1.7
C <sub>1</sub>	128	3.6
C <sub>2</sub>	67	7.5
C <sub>3</sub>	53	6.9
C <sub>4</sub>	58	7.8
S <sub>1</sub>	28	1.6
S <sub>2</sub>	74	2.3
S <sub>3</sub>	35	1.9
S <sub>4</sub>	36	2.1

**Part IV Plant Response to Soil Treatments in July and August 2015**

A. Seeded native plants:

<b>Experimental Plot</b>	<b>Number of Plants</b>	<b>Shoot Length of Plants (cm)</b>
July 20 and 21, 2015		
L <sub>1</sub>	55	8.4
L <sub>2</sub>	60	11.6
L <sub>3</sub>	42	11.6
L <sub>4</sub>	45	12.8
C <sub>1</sub>	44	26.5
C <sub>2</sub>	34	25.0
C <sub>3</sub>	37	25.2
C <sub>4</sub>	38	23.4

S <sub>1</sub>	40	11.9
S <sub>2</sub>	41	14.0
S <sub>3</sub>	25	13.6
S <sub>4</sub>	47	13.2
<hr/>		
August 17, 2015		
L <sub>1</sub>	44	11.8
L <sub>2</sub>	38	15.4
L <sub>3</sub>	36	13.3
L <sub>4</sub>	48	16.4
C <sub>1</sub>	56	40.9
C <sub>2</sub>	44	34.6
C <sub>3</sub>	50	39.2
C <sub>4</sub>	41	40.9
S <sub>1</sub>	24	22.3
S <sub>2</sub>	28	22.1
S <sub>3</sub>	25	24.2
S <sub>4</sub>	36	28.3

B. Naturally colonised plants:

<b>Experimental Plot</b>	<b>Number of Seedlings</b>	<b>Shoot Length of Seedlings (cm)</b>
<hr/>		
July 20 and 21, 2015		
L <sub>1</sub>	16	6.7
L <sub>2</sub>	22	8.4
L <sub>3</sub>	8	6.5
L <sub>4</sub>	11	8.8
C <sub>1</sub>	34	31.3
C <sub>2</sub>	18	25.9
C <sub>3</sub>	23	22.9
C <sub>4</sub>	28	27.4
S <sub>1</sub>	20	9.0
S <sub>2</sub>	35	24.1
S <sub>3</sub>	13	14.6
S <sub>4</sub>	18	14.7
<hr/>		
August 17, 2015		
L <sub>1</sub>	11	8.4
L <sub>2</sub>	13	8.8
L <sub>3</sub>	12	8.0
L <sub>4</sub>	10	8.5
C <sub>1</sub>	27	47.3
C <sub>2</sub>	15	18.7
C <sub>3</sub>	18	30.8

C <sub>4</sub>	16	16.5
S <sub>1</sub>	14	10.3
S <sub>2</sub>	19	9.2
S <sub>3</sub>	9	15.0
S <sub>4</sub>	16	8.9

C. Naturally colonised plant species diversity:

1. See below for monthly specific counts of colonised plant species found in the experimental plot.

Number of plants of each naturally colonised plant species																			
Experimental Plot	yellow clover ( <i>Trifolium agrarium</i> )	henbit ( <i>Lamium amplexicaule</i> )	field pennycress ( <i>Thlaspi arvense</i> )	Canada thistle ( <i>Cirsium arvense</i> )	yellow thistle ( <i>Cirsium horridulum</i> )	wormseed mustard ( <i>Erysimum cheiranthoides</i> )	nodding smartweed ( <i>Polygonum lapathifolium</i> )	early goldenrod ( <i>Solidago juncea</i> )	sweet goldenrod ( <i>Solidago odora</i> )	prostrate pigweed ( <i>Amaranthus blitoides</i> )	shepherd's purse ( <i>Capsella bursa-pastoris</i> )	common ragweed ( <i>Ambrosia artemisiifolia</i> )	oxeye daisy ( <i>Leucanthemum vulgare</i> )	redroot amaranth ( <i>Amaranthus retroflexus</i> )	crabgrass ( <i>Digitaria sanguinalis</i> )	lamb's quarter ( <i>Chenopodium album</i> )	Species Richness	Total Plant Abundance	Shannon-Wiener H'
July 20 and 21, 2015																			
L <sub>1</sub>	1	9	1	1		1	1			1						1	8	16	1.537
L <sub>2</sub>	10	7	3								1					1	5	22	1.275
L <sub>3</sub>	2	3	1	1						1							5	8	1.494
L <sub>4</sub>	2	4	1	4													4	11	1.264
C <sub>1</sub>	1	8	5	9			1			4			2			4	8	34	1.852
C <sub>2</sub>		6	3	3	1					4						1	6	18	1.619
C <sub>3</sub>	3	5	5	5						3		2					6	23	1.739
C <sub>4</sub>		5	6	12						5							4	28	1.308
S <sub>1</sub>		11	1	4						4							4	20	1.122
S <sub>3</sub>	1	6	4					1			1						5	13	1.311
S <sub>2</sub>	3	6	20	3		1	1					1					7	35	1.348
S <sub>4</sub>	3	7	2	5						1							5	18	1.426
August 17, 2015																			
L <sub>1</sub>	4	3					2			1			1				5	11	1.468
L <sub>2</sub>	6	2		2			1						2				5	13	1.418
L <sub>3</sub>	4	1		4				1		1	1						6	12	1.561
L <sub>4</sub>	1	1		6				1							1		5	10	1.228
C <sub>1</sub>	1	1		14					2	3			1			5	7	27	1.456

C <sub>2</sub>	3	2	5	1		3		1	6	15	1.640
C <sub>3</sub>	5		7				5	1	4	18	1.239
C <sub>4</sub>	1	2	11			2			4	16	0.951
S <sub>1</sub>	9		3		1	1			4	14	0.803
S <sub>2</sub>	11	2	3		1		2		5	19	1.237
S <sub>3</sub>	2	1	2		3			1	5	9	1.523
S <sub>4</sub>	3	6	5		1			1	5	16	1.392

C. 2. Detailed notes on correction of naturally colonised plant lifecycle effects for colonised plant species richness, abundance, and Shannon-Wiener diversity index ( $H'_w$ ) analyses:

To test the effect of soil treatments on species richness in summer 2015 (i.e. July and August 2015), the higher monthly species richness value of each of the four experimental plots per treatment was selected for statistical analysis with a sample size of 4.

To observe for effect of soil treatments on the distribution pattern of relative species abundances in summer 2015 (i.e. July and August 2015), the following steps were taken to generate the rank-abundance curve: 1. Within an experimental plot, for each weed species, I selected the higher value of the monthly relative abundance of that species between July and August 2015; I repeated this procedure for each of the other weed species in that experimental plot. 2. I repeated step 1 for the three other experimental plots of the same soil treatment; I repeated the procedure so far for the other two experimental treatments. This generated for each species per treatment four highest possible relative abundance values within the two months of July and August 2015. 3. I took the average of the four relative abundance values for each species per treatment. 4. I ranked these average highest possible relative abundance values from the highest to the lowest, with the highest average value assigned a rank of 1 to plot a rank-abundance curve for colonising weeds on each soil treatment. 5. Keeping track of the weed names with the corresponding ranks of relative abundance, contrasts between soil treatments were made to visually inspect the effect of soil treatment on specific plant species dominance and on the evenness of colonising plant species distribution.

To test the effect of soil treatments on colonised plant species  $H'_w$  in summer 2015 (i.e. July and August 2015), the higher monthly  $H'_w$  value of each of the four experimental plots per treatment was selected for statistical analysis with a sample size of 4.

## Appendix II Photos of the Restoration Experimental Plots

Besides quantitative measurements of plant numbers and shoot lengths, photos of the experimental plots were also taken from May to August 2015 for visual observation of the experimental restoration effects. Please refer back to Figure 3 for soil treatment allocation to the experimental plots, the naming of the experimental plots, as well as experimental plot structure. In each picture, four red flags delineated the four corners of a restoration plot. For convenience of comparing the treatment effect, experimental plot photos are grouped by soil amendment treatment.

**May 14, 2015**

State of the experimental plots immediately after seed planting, watering and a week after site preparation:



**A. Seedling Germination: June 19, 2015**

Leaf:Soil treatment plots:



L<sub>1</sub>



L<sub>2</sub>



L<sub>3</sub>



L<sub>4</sub>



**A. Seedling Germination: June 19, 2015**

Compost:Soil treatment plots:



C<sub>1</sub>



C<sub>2</sub>



C<sub>3</sub>



C<sub>4</sub>

**A. Seedling Germination: June 19, 2015**

Soil Ctrl plots:



S<sub>1</sub>



S<sub>2</sub>



S<sub>3</sub>



S<sub>4</sub>

**B. Summer Time Plant Dynamics: July 20, 2015**

Leaf:Soil treatment plots:



L<sub>1</sub>



L<sub>2</sub>



L<sub>3</sub>



L<sub>4</sub>



**B. Summer Time Plant Dynamics: July 20, 2015**

Compost:Soil treatment plots:



C<sub>1</sub>



C<sub>2</sub>



C<sub>3</sub>



C<sub>4</sub>

**B. Summer Time Plant Dynamics: July 20, 2015**

Soil Ctrl plots:



S<sub>1</sub>



S<sub>2</sub>



S<sub>3</sub>



S<sub>4</sub>



**B. Summer Time Plant Dynamics: August 17, 2015**

Leaf:Soil treatment plots:



L<sub>1</sub>



L<sub>2</sub>



L<sub>3</sub>



L<sub>4</sub>

**B. Summer Time Plant Dynamics: August 17, 2015**

Compost:Soil treatment plots:



C<sub>1</sub>



C<sub>2</sub>



C<sub>3</sub>



C<sub>4</sub>

**B. Summer Time Plant Dynamics: August 17, 2015**

Soil Ctrl plots:



S<sub>1</sub>



S<sub>2</sub>



S<sub>3</sub>



S<sub>4</sub>