Assessing Land-use Legacy Effects on Soil Physico-Chemical Properties and Earthworm Biodiversity in Urban Parks

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complies with the regulations of the University and meets the accepted standards with respect to originality and quality.

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Abstract

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Human land-use alters soil properties and biodiversity differently depending on the intensity and type of use, often resulting in persistent temporal effects known as legacy effects. Cities are expected to be rich in legacy effects due to their development histories and complex socio-ecological landscapes. However, few urban ecological studies consider the role of history in shaping contemporary patterns. Therefore, we asked: do soil properties and biodiversity of our present-day urban greenspaces differ due to varied historical land-use? We surveyed 25 urban parks across the island of Montreal, Quebec, Canada with three former land-uses: forested (low intensity), agricultural (medium intensity), and industrial (high intensity). We measured soil bulk density, heavy metal concentrations, and carbon and nitrogen stocks, as well as earthworm abundance, biomass, species richness, and community composition. Most studied soil properties did not differ across historical land-uses. All properties except for heavy metal concentrations significantly increased with age, implying a legacy effect of recovery from disturbance and management post park establishment. Earthworm distribution was highest in forested sites whereas earthworm biodiversity was lower in previously agricultural sites. These findings suggest that aspects of soils in our urban greenspaces are minimally susceptible to legacy effects of historical human land-use. This demonstrates a certain effectiveness in developing municipal parks on a variety of past land-uses. This could allow for a focusing on current management choices and decisions which may have a greater influence on park ecosystem functioning.

Acknowledgements

In 1675, Isaac Newton famously wrote in a letter that "*if I have seen further, it is by standing on the shoulders of giants*". Likewise, I have, too.

I would first like to thank my friends and family for their endless encouragement in this journey. Never did they doubt my resolve. Never were they naysayers. Never did they try to persuade me or convince me otherwise from pursuing this endeavour and never did they leave my side. They stuck by me, through thick and thin, and now, at the finish line, I am happy to share my achievement with them because this thesis is as much a product of my own as it is theirs.

I would like to thank all of my lab mates for their support, motivations, and insights with my research. To all of those who, throughout the years, gave me feedback on my writing, my presentations, my posters, or anything in between: your influence has directly contributed to my success and the completion of my thesis. Of my lab mates, though, I would like to especially thank Bella who served as an inspiration to me on what it means to do good science. From experimental designs to statistical thinking, you have imparted such a breadth of wisdom onto me that I now know in which ways I am lacking; but, most importantly, in which ways I can continue to grow as a scientist myself.

Lastly, but certainly not least, I would like to thank my supervisor, Carly, for taking me on as her student. After having finished my bachelor's, and even before then, I knew that I wanted to pursue graduate research. However, life was not so kind to me and for three years I sought opportunity after opportunity. With bad luck seemingly following me around, it was not until I came into contact with Carly that I knew my despair was finally over. Her enthusiasm for soils and similar interests had renewed my hope that at last I would be able to realize my goals and ambitions. So thank you, Carly, for giving me this chance and for your mentorship. Thank you for initiating me to the world of urban research, for helping me push my reflections, for providing me with the tools and means necessary to conduct my research, for your compassion and understanding, and, above all else, for giving me the creative freedom to explore soils and worms to my heart's content.

Contribution of Authors

Conceived and designed the analysis: Michael, Isabella, Carly

Collected the data: Michael

Contributed additional data or analysis tools: Isabella

Performed the analysis: Michael

Wrote and reviewed the paper: Michael, Carly

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Land acknowledgement

Tiohtià:ke, otherwise known as "Montreal", sits on the stolen and unceded lands of the Kanien'kehá:ka (Mohawk) Nation. In my research, though I studied the legacy effects of land-use post-colonialism, this is not meant to forget nor erase the presence and stewardship that was afforded to the land by Indigenous peoples prior to the arrival of European settlers.

General Introduction

Land-use cover, change, and intensity are all ways human practices can fundamentally alter soils. Whether it is agriculture, deforestation, mining exploits or urbanization, all land-uses have the potential to modify soil nutrients (Chaurasia et al., 2023), soil bulk density (Pouyat et al., 2007) and soil heavy metal concentrations (Ghorbani et al., 2015). Some modifications may even persist in time and "carry over" to a subsequent land-use, defined as a legacy effect (Foster et al. 2003, Ossola et al. 2021).

Incorporating a historical temporal aspect into ecology, dubbed historical ecology, can serve to explain changes in abiotic conditions or species compositions by acknowledging that past human events and decisions can affect the dynamics of the present-day (Lunt & Spooner, 2005). For example, ancient Roman agriculture explained the variation in soil properties nearest settlements such that the current-day tree composition differed with distance to these settlements (Dambrine et al., 2007); or, similarly, proximity to historical disturbance explained overall earthworm patterns of invasion in a nature reserve (Beauséjour et al., 2015). Thus, considering a historical perspective can be a powerful means to understand how current land-uses may influence soil properties, functions, and services in the future.

Today, as more people flock towards urban centres, land-use is intensifying globally (Foley et al., 2005) as our demand for resources grows. Cities are dynamic, ever-changing landscapes whose development has been witness to multiple co-occurring trajectories. Consequently, cityscapes are rich in historical land-uses and, by extension, legacies. Legacy effects may threaten underlying ecosystem functions vital for the sustainability of cities by modifying soil properties and biodiversity. One way to mitigate this and promote sustainability could be through promoting soil characteristics that support soil biodiversity and ecosystem functioning (Guilland et al. 2018, Sun et al. 2023). Soil biodiversity mediates many soil functions that interact with aboveground components in a tight feedback loop (Wardle et al., 2004). This feedback loop is vital down-the-line for ecosystem services, defined as the benefits humans derive from ecosystems (MEA, 2005). For example, decomposition by soil biota recycles organic matter and nutrients. Without decomposition, trees, known for their multiple ecosystem services (Roeland et al., 2019), could not thrive. Soil biodiversity, however, ultimately depends upon soil characteristics. Despite the critical importance of soils as underpinning urban ecosystem functions and services, urban soils (and particularly the role of land-use legacies), is understudied. Thus, there is a need to understand the ramifications of our past urban land-uses.

In the following two chapters, I explore the links between historical land-use, land-use intensity, soil properties, and soil biodiversity in an urban environment. Broadly, I asked the following two research questions in Montreal, Quebec, Canada:

1. How does historical land-use affect current soil physico-chemical properties in urban parks?
2. How does historical land-use affect current soil biodiversity in urban parks?

In both chapters, I determined that the dominant historical land-uses of my study sites preceding their conversion to urban parks were remnant forests, agricultural, or industrial land-uses. In my first chapter, I measured soil bulk density, heavy metals concentrations, carbon (C) stocks, and nitrogen (N) stocks. In my second chapter, I used earthworms as model soil organisms because of their implications as ecosystem engineers for ecosystem functions and services. For this chapter, I sampled a subset of the sites from my first chapter and measured earthworm abundance, biomass, species richness, and community composition.

Chapter 1

Title: Assessing Land-use Legacy Effects on Physico-Chemical Soil Properties in Urban Parks

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1. Introduction

Land-use type, intensity, and change are three ways humans can influence soil physico-chemical characteristics. For example, studies comparing the effects of various land-uses on soil properties have shown how soil organic carbon and total nutrients (Chaurasia et al., 2023), as well as bulk density, calcium, and potassium concentrations (Pouyat et al., 2007) differ significantly across land-uses. Golubiewski (2006) showed how residential turfgrass land-use in Colorado, U.S.A., resulted in higher carbon pools than the native grassland counterparts. Land-uses that impact soil physico-chemical characteristics span a gradient of relative intensity, and intensity itself can also directly affect soil properties. For example, agricultural practices, such as tillage, physically pulverizes soil aggregates and particles. Consequently, soils under intensive tilling have increased soil compaction which reduces water holding capacity, porosity, water infiltration, and the soil's capacity to sequester carbon (Bogunovic et al., 2020). Soil properties can also be affected by land-use change, defined as a change in both land-use type and intensity. For example, soil organic carbon stocks generally increase when croplands are reclaimed by vegetation into grasslands or forests (Deng et al., 2016). Conversely, deforestation generally decreases soil organic carbon stocks (Deng et al., 2016). Increase in bulk density is also common when pastures are cleared for croplands where heavy machinery is used for ploughing (Haghighi et al., 2010). The extent that the soil properties are affected ultimately depends on what the previous and new land-use are. With global trends towards intensification (Foley et al., 2005) and frequent land-use conversions, continuing to assess how land-use change can affect our soils is crucial for understanding the relationship between soil properties and belowground ecosystem functioning vital for producing ecosystem services.

Legacy effects, defined as past events that persist through time and that "carry over" (Foster et al., 2003), can be caused by historical land-uses. Soils may be particularly susceptible to legacy effects because the pedological forces responsible for shaping belowground functions, such as soil biota activity and the addition, removal, translocation, and transformation of material, take significant time to occur (Walker et al. 2012, Ziter et al. 2017). Since ecological functions are already slow moving and slow acting, this means that soils can be recipients of legacies and thus strong systems to assess legacy effects. For example, in a current forest in France that was used as agriculture in ancient Roman times, the effects of agricultural land-use persisted such that soil P, pH, and the C:N ratio were distinctive nearest former settlements (Dambrine et al., 2007). Lead contamination in urban centers has also been shown to be a legacy effect of industrial land-uses. For example, neighborhoods that sit on historical industrial sites such as refineries (Clarke et al., 2015) or that were built at a time when lead paint and leaded gasoline were prevalent (Paltseva et al., 2022) have significantly higher levels of lead. In Canada, lead paint and leaded gasoline were only banned in 1976 and 1990, respectively, leaving ample time for sources to accumulate and persist. Soil nutrients, too, are subject to legacy effects of historical land-uses. For example, carbon, nitrogen, and phosphorous stocks can be greater in more developed land such as residential areas and lowest in semi-natural land such as grasslands and forests (Raciti et al. 2011, Ziter & Turner 2018). Of course, to assess if a given soil is affected by a legacy of a historical land-use, the history of a site must be known. Oftentimes, data scarcity and inconsistencies (Tomscha et al., 2016) or missing and undocumented data (Bürgi et al., 2017) serve as impediments. Despite the difficulties in retracing the historical land-use of a site, it is important to include an element of historical ecology in our soil sciences. If we want to understand the extent to which our anthropogenic activities might affect contemporary soils, then we must consider the past since the land-uses of today are tomorrow's legacy.

Urban ecosystems offer an underexplored opportunity to assess legacy effects (Ziter & Turner, 2018). Cities are not built in a day. Urban land-use change occurs at differing rates and magnitudes and not uniformly across time. Multiple development trajectories co-exist simultaneously: some former agricultural lands are being turned into a housing project in the east, a former forest is being turned into a tech industrial sector in the north, or a former landfill is being turned into a neighbourhood park in the south. The dynamic nature of city landscapes (Cadenasso et al. 2007, Pavao-Zuckerman & Byrne 2009, Pickett et al. 2021) results in urban centres with rich land-use histories of varying type and intensity. And, since many North American cities are relatively young compared to cities elsewhere in the world, it could be that we are just now realizing how lingering the many consequences of our earlier land-use activities are. Thus, cities offer the potential to explore temporal dynamics, a component that is often poorly quantified in ecological studies (Ossola et al., 2021), as legacy effects of historical land-use in a high heterogeneity setting.

Moreover, if we want sustainable cities for the future, or the ability for our cities to meet the needs of the present without compromising the needs of the future (UN Brundtland Commission, 1987), then we cannot ignore urban soils, an underrepresented area in both the broader soils literature (Guilland et al., 2018), and the urban ecology literature (Knapp et al., 2021). We already know that urbanization has led to a loss of soil horizons (Herrmann et al., 2018), a widespread increase in soil compaction (Edmondson et al., 2011), and that impervious surfaces deeply alter the water cycle through soil sealing (O'Riordan et al., 2021). There is concern about whether such degraded soils can adequately support our growing physical infrastructure and other urban needs, especially as more and more people move to urban centres where half the world's population is already found (UN DESA, 2018). Thus, there is a need for better managing our intensive urbanized land-uses. One way to achieve sustainability could be through assuring that urban soils can provide vital ecosystem services by promoting soil properties. Soil physical, chemical, and biological properties are highlighted in frameworks as the basis for soil ecosystem functioning (Dominati et al., 2010), with indicators such as soil texture, pH, and organic matter content commonly weighed and rated into soil quality indices (Blanchart et al., 2018). Ultimately, if it is of great interest to stakeholders to maintain the provision of multiple soil-based ecosystem services, research is needed to better understand the temporal processes – including past land use – that may influence soil properties.

Here, we investigated how different historical land-uses affected urban soil properties that have implications for ecosystem functioning. Our goal was to identify common historical land-uses of current greenspaces within a typical North American city and link these past land-uses to current-day variation in soil properties. Specifically, we asked: how do the soils of present-day urban greenspaces characterized by historically forested, agricultural, and industrial land uses differ in their current physico-chemical properties? Across urban parks in Montreal, QC, we measured soil bulk density (defined as the mass of soil in a given volume of earth), concentrations of eight heavy metals, and C and N stocks. We hypothesized that more intensive historical land-uses would result in greater changes in soil properties. We characterized intensity on a relative gradient between historical land-uses (since we did not have a baseline or "reference"), whereby previously forested sites were considered the least intensive past land-use, followed by previously agricultural and then by previously industrial sites, the most intensive.

We predicted that the soil bulk density of previously forested sites would be lowest (Pouyat et al., 2007) as they were the least intensively managed historical land-use in our study whereas we predicted that the soil bulk density of previously industrial sites would be the greatest since historical industrial land-uses are known to compact the soil significantly (Janz et al., 2019).

We predicted that heavy metals would be most concentrated in previously industrial sites. Industrial historical land-uses such as surface open-pit mining can weaken the siding walls of quarries. This means that any subsequent land-use, like landfills, can potentially release gases and contaminated leachate upwards through the fractures of the weakened siding walls (El-Fadel et al., 2001). This is especially exacerbated by water percolating through the soil profile, something that our study sites, urban parks, can be particularly susceptible to as they are some of the few permeable surfaces in an impermeable concrete and asphalt dominated landscape. Additionally, we predict that previously agricultural sites would have a higher concentration of heavy metals than previously forested sites as some chemical and organic fertilizers, modern and historical, contain traces of heavy metals (Arao et al. 2010, Zwolak et al. 2019).

We predicted that carbon stocks would be the lowest in either previously industrial or previously agricultural sites. Previously industrial sites generally entail the destruction of the previous land-use, resulting in massive losses of stored carbon (Rooney et al., 2012). However, it is possible that sites with agricultural history would have the lowest C stocks albeit being a lesser intensive land-use than industrial. Agriculture is known to reduce C stocks through the removal of biomass and the pulverization of the topsoil via tilling and ploughing (McLauchlan, 2006). Under this alternative hypothesis, we predicted that historically forested and industrial sites would exhibit greater levels of C than historically agricultural land-uses.

Lastly, we predicted that nitrogen stocks would be lowest in previously industrial sites by the same mechanisms as proposed in our aforementioned carbon predictions. We also predicted that agricultural historical land-uses would have the greatest N stocks; however, we predicted this not because of our stipulated land-use intensity gradient, but rather because of fertilizer use in past agricultural practices. Fertilizers are known to increase N rates in agricultural soils (McLauchlan, 2006).

2. Methods

2.1 General

2.1.1 Study area

Our study was conducted on the Island of Montreal, Quebec, Canada (45.5019° N, 73.5674° W). This includes the 19 boroughs that make up the City of Montreal as well as the 14 adjacent boroughs not under the administration of the City of Montreal. Montreal is characterized by a temperate climate as part of the Saint Lawrence Lowlands region of southern Quebec where temperatures annually range from -13ºC to 26ºC with a mean of 7ºC and receiving an annual precipitation mean of 48.3mm (Environment and Climate Change Canada). Soils across Montreal are mainly the products of two historical events: glaciation and the Champlain Sea. Upon the advance and retreat of the Laurentide Ice Sheet, glaciers ground and left behind deposits known as glacial till. When the final glaciers retreated for the last time during the Wisconsin ice age some 12000 years ago, the land was depressed, causing the Atlantic Ocean to flood the area, thereby creating the Champlain Sea (Moore, 2021). As the Earth's crust was undergoing isostatic rebound, the sea, by wave action and through marine sediments, deposited alluvial materials (Moore, 2021). As such, the bedrock for pedogenesis has mainly gravel and clay origins (Lajoie & Baril, 1952). Within Montreal we focused our efforts on a single contemporary land use – urban parks. These urban greenspaces were accessible for soil sampling and spanned a range of historical land-uses that we were able to reconstruct through publicly available records.

2.1.2 Historical land-use determination

We classified parks by their historical land-use based on the immediate previous land-use prior to their official designation as a city park. We considered the year of the official designation as the time-since-establishment, or age, of the site. We expected this to be the legacy which would have the strongest effect on contemporary soil parameters as it was the most recent land-use. By combing the City of Montreal's archives and the national archives of Quebec, we were able to reconstruct the historical land-uses of parks through old newspaper clippings, historical photographs, and aerial images. In this way, we were able to determine three dominant historical land-uses in Montreal: forest remnants (forested), farms (agriculture), and quarries turned landfills (industrial). In cases where a single park had multiple historical land-uses, the respective area per historical land-use was delineated to avoid detecting mixed legacy effects. For example, a park may have been formed on two separate parcels of land, one which could have been historically forested and the other which could have been historically used for agriculture – this would be considered as two sites in our analysis.

2.1.3 Site selection

Finding clear documentation in the archives such that we were confident of the reconstructed historical land-uses of our chosen parks meant that we were limited in the number and scope of our sampling sites. Due to the challenges and time commitment inherent in working with historical ecological records (see Tomscha et al. 2016), and the intensity of field sampling, we limited our study to ten sites per historical land-use, for a total of 30 sites, across a total of 25 contemporary parks (see table 1 and figure 1).

Table 1: Selected Montreal study sites and their historical land-use prior to being designated as parks. Parks are listed alphabetically within each historical land use. Year that sites received official park designation is listed in brackets.

Forested	Agricultural	Industrial
Angrignon (1927)	Angrignon (1927)	Arthur-Therrien (1964)
L'Anse-à-l'Orme (1987)	L'Anse-à-l'Orme (1987)	Baldwin (1901)
Bois-de-Liesse (1987)	Benny (1965)	Botanical Gardens (1936)
Bois-de-Saraguay (1987)	Cap-Saint-Jacques (1985)	Félix-Leclerc (1990)
Boisé-du-Saint-Sulpice (1990)	Centennial/Memorial Hall (1945)	Frédéric-Back (2016)
Cap-Saint-Jacques (1985)	Fritz (1979)	Lafond (1952)
Coulée-Grou (1988)	Lafontaine (1901)	Lalancette (1922)
Jean-Drapeau (1908)	Marguerite-Bourgeoys (1913)	Père-Marquette (1953)
Pointe-aux-Prairies (1992)	Pointe-aux-Prairies (1992)	Pointe-aux-Prairies (1992)
Thomas Chapais (1960)	Saint-Gabriel (1887)	Promenade Bellerive (1978)

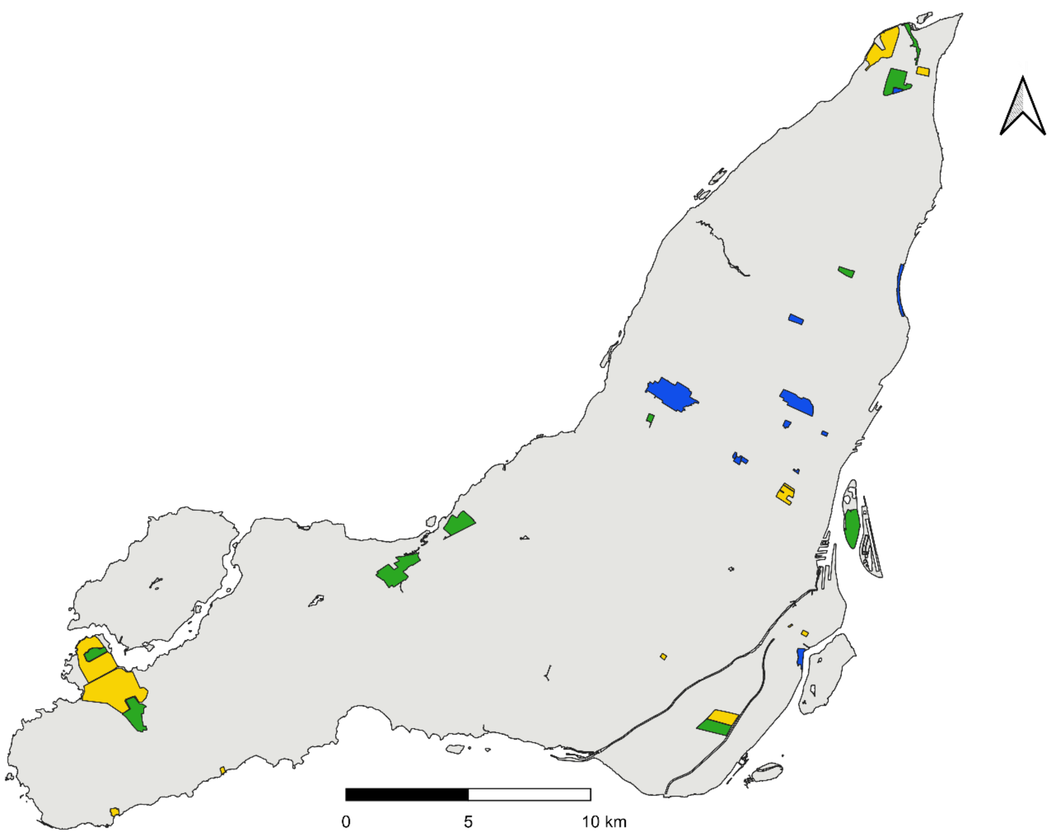

Figure 1: Map of study sites and their respective historical land-use. Sites were geographically distributed across the island to the extent possible. However, certain legacies tend to cluster spatially due to historical patterns in land use

and development. Colour scheme: previously forested (green), previously agricultural (yellow), previously industrial (blue).

2.1.4 Experimental design

This study was a part of a greater research effort investigating legacy effects of historical land-uses on both urban soils and urban forest composition and structure. As such, the sampling design was meant to account for multiple goals. To ensure sampling locations were representative of each site as a whole, at each site we characterized three classes of canopy cover based on satellite imagery: low canopy (10% - 30% coverage), medium canopy (31% - 80%), and high canopy (> 81%). For each class of canopy cover, we randomly generated three sampling plots of $800m^2$, for a total of nine plots per site. In this manner, our plots captured the heterogeneity of differing levels of vegetation throughout our sites, for example, mowed turfgrass, meadows, and forests (Figure A1.1-A1.3). At some sites, nine sampling plots could not be generated due to lacking canopy cover within certain classes. In such rare cases, we sampled the available number of plots (i.e. six, seven, or eight plots instead of nine; Figure A2.1-A2.3). In total, 255 sampling plots were established across our 30 sites.

2.2 Soils analysis

2.2.1 Field collection

We used an AMS bulk density sampler to take a single 90.59 cm^3 core at the center of each sampling plot to determine bulk density. For heavy metals, C, and N, we used a footrest soil auger (diameter = 14mm) to core and collect samples at a depth of 15cm, where possible, at each quadrant of the plot; that is, the north-eastern, south-eastern, north-western, and south-western quadrants (Figure 2). When the soil was too compact and a 15cm depth core could not obtained, we collected a sample core as close to a depth of 15cm as possible. The four cores were subsequently pooled together to make a composite soil sample to ensure that we captured soil variation across the plot. All soil samples were kept in plastic Ziploc® bags and stored in a 3^oC refrigerator for further analysis.

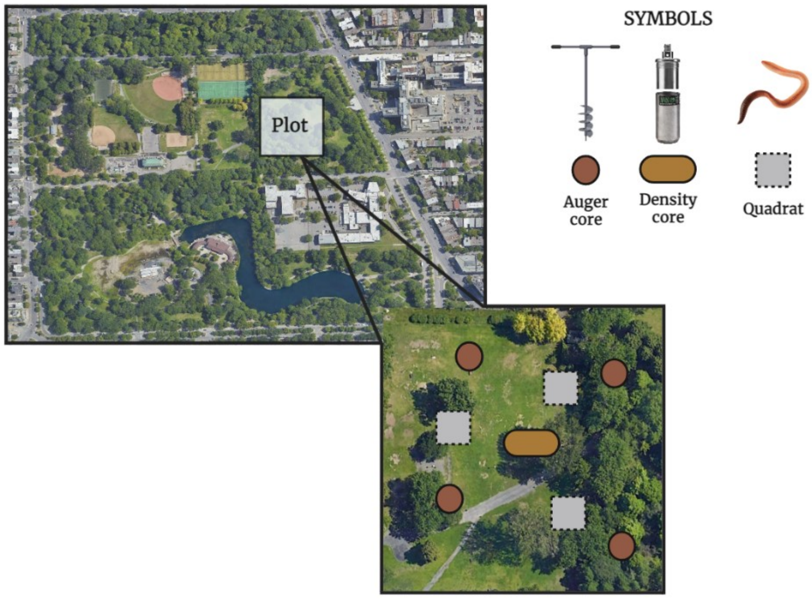

Figure 2: Experimental design for sampling the soil and earthworms in urban parks. Note that plot and quadrat dimensions are not to scale.

2.2.2 Bulk density

We crushed the samples from the AMS bulk density sampler thoroughly by hand using a mallet, and then sieved at 2mm. Particles greater than 2mm such as pieces of wood or root tips were kept separately. Similarly, we separated coarse rock particles, and their volume was measured using Archimedes' water displacement principle. All samples and particles greater than 2mm were dried at 105ºC for 24 hours in paper bags and then weighted. Bulk density was calculated using the following three different measurements according to Throop et al. (2012):

$$BD_{core} = \text{Mass of entire core / Volume of entire core} \quad (1)$$

$$BD_{FE} = \text{Mass of fine earth fraction (<2mm) / Volume of fine earth fraction} \quad (2)$$

$$BD_{hybrid} = \text{Mass of fine earth fraction/Volume of entire core} \quad (3)$$

According to Throop et al. (2012), conventional measures of bulk density bias their measurements. They argue that most cores are small and thus exclude coarse fragments larger than the diameter of the core. This can be a considerable source of error when sampling soils that are expected to contain large fragments, e.g., glacial till, alluvial deposits, etc., such as those found across the Island of Montreal. Therefore, Throop et al. (2012) developed a hybrid calculation which would help reduce inflated values. In our dataset, we verified that ρ_{core}, the conventional measure of soil bulk density, was correlated with ρ_{hybrid} visually (Figure A3.1) and via Pearson correlation tests ($R = 0.9993$), illustrating that coarse fragments were not greatly present in our soil bulk density samples. Going forward, we used the hybrid calculation, and henceforth refer to it simply as soil bulk density.

2.2.3 Trace elements

We sieved the composite soil samples at 2mm. We then processed the sieved samples at the lab of Dr. Nicolas Bélanger of TÉLUQ University. Samples were ground in a Retsch PM400 ball mill for four minutes. Then, we placed the ground soil composite sample into a Reflex Press pneumatic machine to compress the powdered sample down into a hockey-puck-shaped pellet. Using the average of three readings of a handheld X-ray fluorescent device (model type: Vanta Work Station A020-V), we determined the concentrations of the following eight heavy metals: arsenic (As), cadmium (Cd), mercury (Hg), copper (Cu), chrome (Cr), lead (Pb), nickel (Ni), and zinc (Zn).

2.2.4 Carbon and nitrogen measurements

Once the trace element analysis was completed, the pellets were brough to a secondary lab at l'Université du Québec à Montréal (UQÀM) for processing. There, a Laser Induced Breakdown Spectroscopy (LIBS) analyzer was used to assess the total carbon and total nitrogen percentages per pellet of soil sample. The laser swept across the tops of each pellet five times within the boundaries of a concentric circle atop the pellet's surface. The calibration curves of the instrument's measurement sensitivity, indicated by a linear regression R^2, were of 0.97 and 0.99 for carbon and nitrogen, respectively. Both carbon and nitrogen stocks to a depth of 15cm were calculated according to equation (4).

$$\rho = \frac{\left(\mathrm{A}\ x\ \mathrm{d}\ x\ (\mathrm{BD}*1000)\ x \left(\frac{\mathrm{NC}}{100}\right)\right)}{\mathrm{A}} \tag{4}$$

Where ρ is nutrient density (kg/m^2), A is area of a plot, d is sampling depth, BD is bulk density of a plot (g/cm^3), and NC is nutrient concentration (%)

2.3 Statistical analysis

2.3.1 Analysis pipeline

In all cases except otherwise specified, we analyzed our data by fitting general and generalized linear mixed-effects models (LMMs) using the package `lme4` (Bates et al., 2015). Sites were used across all models as a random effect to account for the non-independence of plots nested within sites. We determined model goodness-of-fit by calculating the conditional pseudo-R^2 values using `MuMIn` (Bartoń, 2023). To test the significance of the fixed effects, we used the "anova" function of the `lmerTest` package (Kuznetsova et al., 2017) with arguments specifying a Type III Analysis of Variance using the Kenward-Roger approximation as this was the recommend choice given our relatively small dataset (Luke, 2017). When the historical land-use predictor was found to be significant in a given model, we carried out post-hoc pairwise comparison of the model marginal means using the `emmeans` package (Lenth, 2023) to determine between which historical land-uses the predicted variable of interest significantly differed. Model diagnostics were performed to validate linear model assumptions. Normality, homogeneity of the residual variance, and independence were assessed visually using `performance` (Lüdecke et al., 2021) and `DHARMa` (Hartig, 2022) and, when unsure, via hypothesis testing (Levene's test and Durbin-Watson test). All data analyses were carried out in R version 4.3.2 (R Core Team, 2021). Prior to analysis, we explored the data following Zuur et al. (2010). Model effects were considered significant at an alpha of 0.05. Outcomes were considered marginally significant when $0.1 > \text{p-value} > 0.05$.

2.3.2 Bulk density

To test the effect of historical land-use on soil bulk density, we fitted a LMM following a Gaussian error structure. Historical land-use, time-since-establishment (hereafter referred to as age), and an interaction term between historical land-use and age were considered as fixed effects. For all other steps, see section 2.3.1.

2.3.3 Trace elements

Both mercury (Hg) and cadmium (Cd) were found at very low levels ($\mu < 0.001$) such that we determined performing any statistical analyses on them would be unnecessary. To test the effect of historical land-use on heavy metals, we fitted LMMs to each of the remaining six metals of interest individually following a Gaussian error structure. For copper, lead, and zinc, we log-transformed the response variable to ensure that residuals did not violate model assumptions. In every model, the only fixed effect was historical land-use. For all other steps, see section 2.3.1.

2.3.4 Carbon stocks

To test the effect of historical land-use on carbon stocks, we first removed all zero-valued observations, of which accounted for nearly 48% of instances. We deemed these to be "false zeros" not biologically relevant for two reasons. First, it is impossible that our sampled soils contained no traces of carbon whatsoever. We suspected, instead, that our zeros were caused by a limited detection and insufficient sensitivity in the LIBS machine used to measure our samples (pers. comm.). This could be corrected for in the future with new standards at very low C concentrations for calibration prior to re-measurement. Second, the mean value of total carbon at our sites with the false zeros removed are in line with the reported means and ranges for studies conducted in other urban environments like New York City (Pouyat et al., 2002), Beijing (Luo et al., 2014), and for soil orders (Batjes, 1996) typically found across the Saint-Laurence Lowlands, the ecoregion where our study took place. In the end, this cut our sample size down to $n = 133$ for this variable. Once the zeros were removed, we log-transformed the response variable and fitted a general LMM following Gaussian error structure. We considered historical land-use, age, and an interaction term between historical land-use and age as fixed effects. For all other steps, see section 2.3.1.

2.3.5 Nitrogen stocks

To test the effect of historical land-use on nitrogen stocks, we log-transformed the response variable and fitted a LMM following a Gaussian error structure. We considered historical land-use, age, and an interaction term between historical land-use and age as fixed effects. For all other steps, see section 2.3.1.

3. Results

3.1 Bulk density

In total, 255 soil samples were collected across 30 sites. Age had a significant positive relationship with soil bulk density (p = 0.0018) irrespective of historical land-use, implying that the older the sites, the denser the soil (Figure 3). Neither historical land-use (p = 0.31) nor the interaction between historical land-use and age (p-value = 0.89) were significant predictors of soil bulk density. The model had a pseudo conditional $R^2 = 0.29$.

Table 2: Output of post-hoc "anova" function on soil bulk density LMM. Previously forested sites are set as the default contrast. Asterisk (*) indicate significance of fixed effects at α = 0.05 and crosses (†) indicate marginal significance.

Predictors	Sum of Squares	Mean square	Numerator df	Denominator df	F value	p-value
Legacy	0.1317	0.0658	2	24.278	1.2385	0.3075
Age	0.6479	0.0648	1	24.849	12.1904	0.0018*
Legacy x Age	0.0122	0.0061	2	24.881	0.1143	0.8924

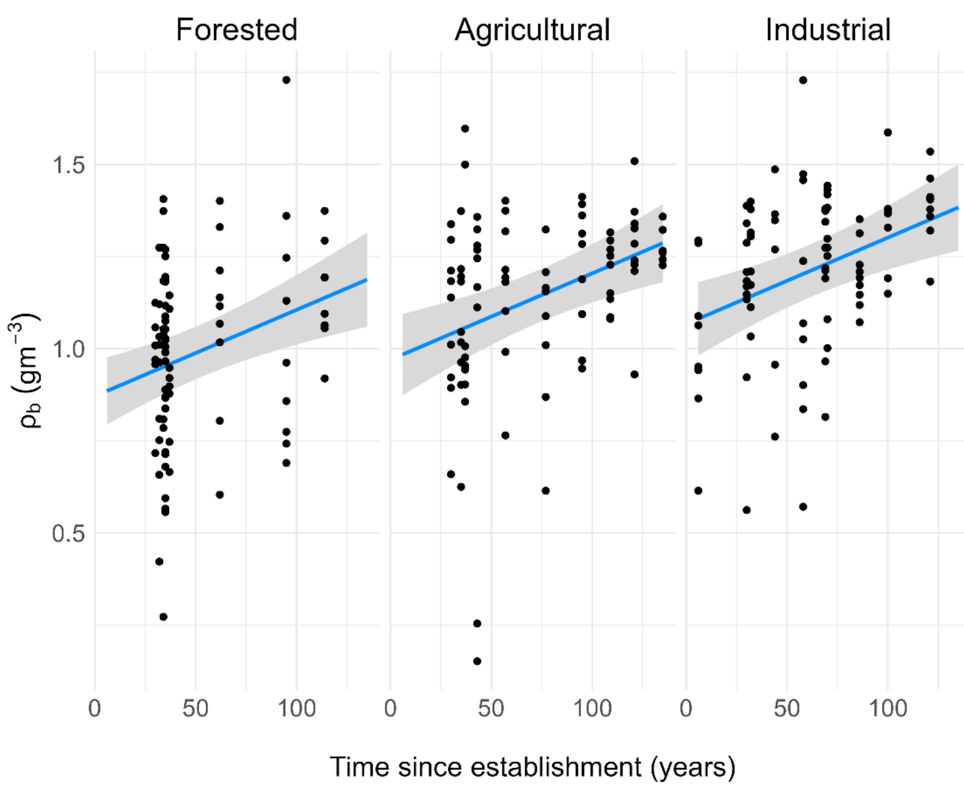

Figure 3: A plot contrasting the effect of age on soil bulk density across three historical land-uses when all other variables in the linear model are held constant. Here, the shaded area represents confidence bands at α = 0.05.

3.2 Trace elements

Historical land-use did not significantly affect the concentration of any heavy metal concentrations analyzed (Table 3).

Table 3: Overview of model results of soil heavy metal LMMs.

Response	Transformation	Error structure	Model assessment	Test of effects
Arsenic	None	Gaussian	Conditional $R^2 = 0.53$	Legacy $p = 0.233$
Chromium	None	Gaussian	Conditional $R^2 = 0.21$	Legacy $p = 0.441$
Copper	Log	Gaussian	Conditional $R^2 = 0.49$	Legacy $p = 0.130$
Nickel	None	Gaussian	Conditional $R^2 = 0.04$	Legacy $p = 0.119$
Lead	Log	Gaussian	Conditional $R^2 = 0.62$	Legacy $p = 0.349$
Zinc	Log	Gaussian	Conditional $R^2 = 0.44$	Legacy $p = 0.225$

3.3 Carbon stocks

Except for previously forested sites, we see that older sites contained greater stocks of carbon (Figure 5), suggesting an accumulation over time. In fact, post-hoc tests revealed that historical land-use (p-value = 0.04) and age (p-value = 0.046) were significant predictors of carbon stocks whereas the interaction term between the two was marginally significant (p-value = 0.051). Specifically, carbon density at previously agricultural sites marginally significantly differed from forested sites (p-value = 0.0583). Carbon density was also highly variable in historically agricultural sites (Figure 4), likely due to variation in historical agricultural practices. The fitted model had a conditional R^2 of 0.35.

Table 4: Output of post-hoc "anova" function on soil carbon density LMM. Previously forested sites are set as the default contrast. Asterisk (*) indicate significance of fixed effects at $\alpha = 0.05$ and crosses (†) indicate marginal significance.

Predictors	Sum of Squares	Mean square	Numerator df	Denominator df	F value	p-value
Legacy	5.6345	2.8173	2	26.755	3.6461	0.0398*
Age	3.4311	3.4311	1	23.385	4.4434	0.0456*
Legacy x Age	5.2293	2.6146	2	23.001	3.3851	0.0515†

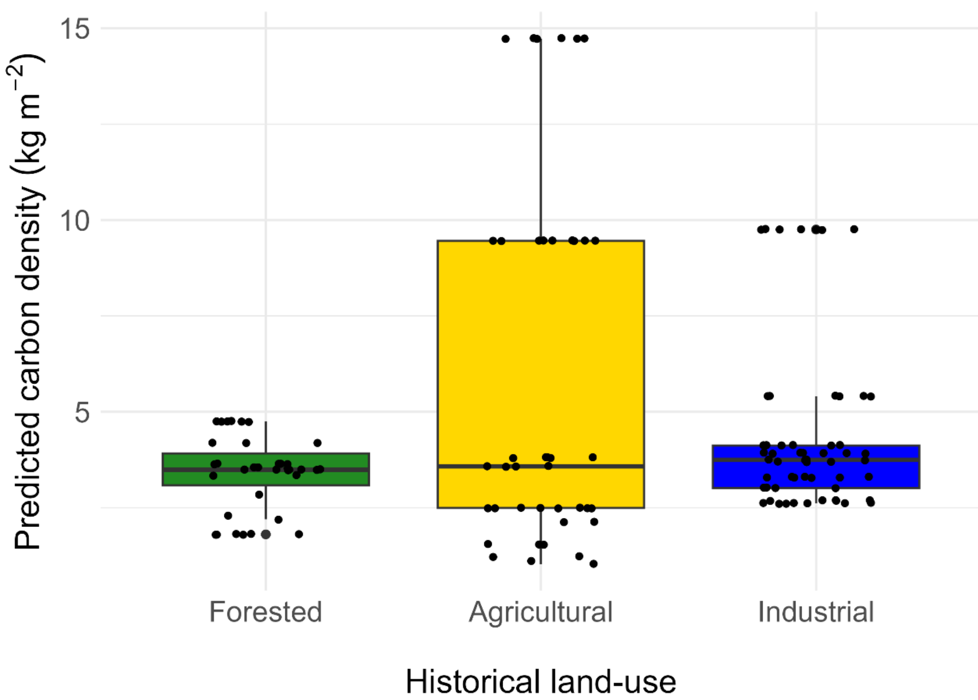

Figure 4: Boxplot of model predictions of soil carbon density across three historical land-uses.

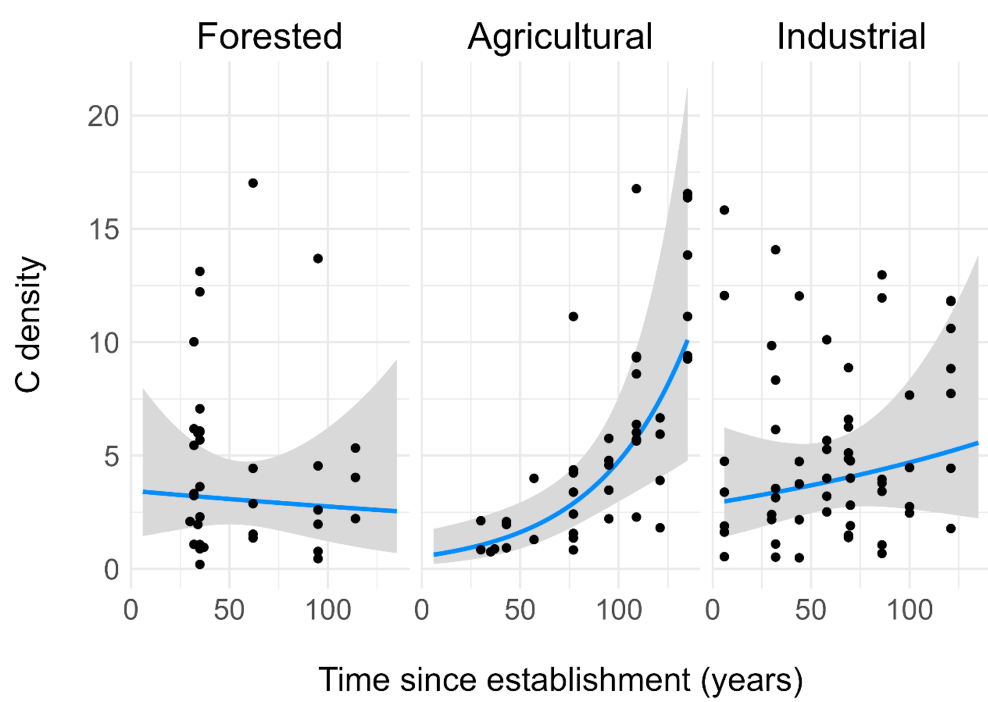

Figure 5: A plot contrasting the effect of age on carbon density (in kgm^{-2}) across three historical land-uses when all other variables in the linear model are held constant. Here, the response variable is exponentiated to back-transform the initial log-transformation. The shaded area represents confidence bands at $\alpha = 0.05$.

3.4 Nitrogen stocks

Nitrogen was half as dense, on average, in previously forested sites than in previously industrial sites (Figure 6). Age (p-value = 0.00019), but not historical land-use (p-value = 0.2), was a significant predictor of log-transformed nitrogen density. In all historical land-uses, the older the site, the denser the nitrogen stocks (Figure 7), also suggesting an accumulation over time. The fitted model had a conditional R^2 of 0.42.

Table 5: Output of post-hoc "anova" function on soil bulk density LMM. Previously forested sites are set as the default contrast. Asterisk (*) indicate significance of fixed effects at α = 0.05 and crosses (†) indicate marginal significance.

Predictors	Sum of Squares	Mean square	Numerator df	Denominator df	F value	p-value
Legacy	0.5996	0.2998	2	24.251	1.7500	0.1950
Age	3.2759	3.2759	1	24.683	19.1223	0.0002*
Legacy x Age	0.8764	0.4382	2	24.711	2.5579	0.0978†

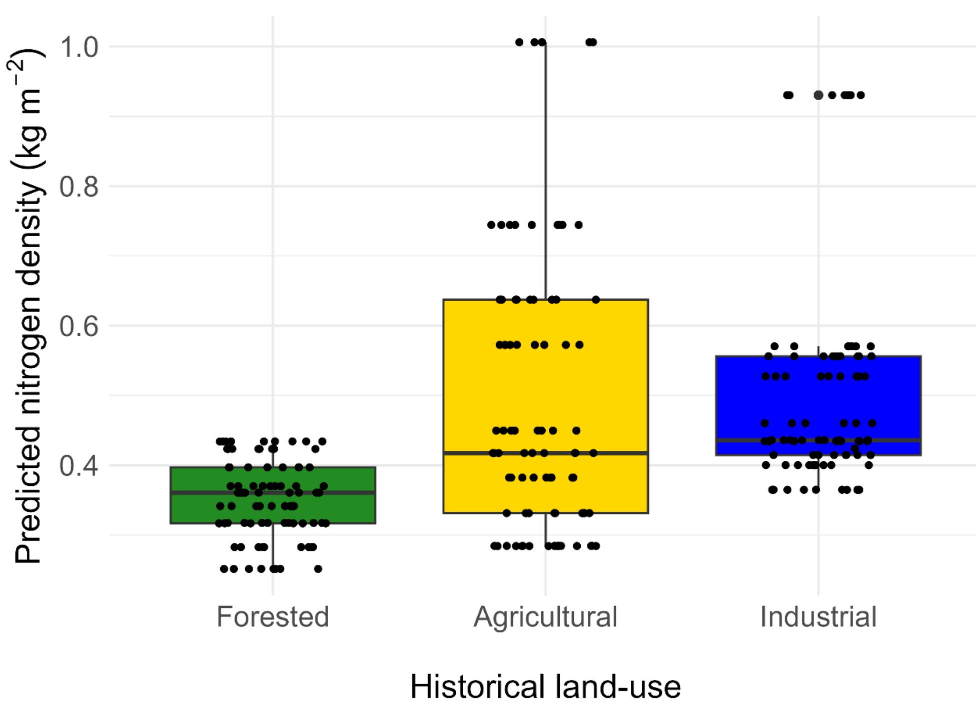

Figure 6: Boxplot of model predictions of soil nitrogen density across three historical land-uses.

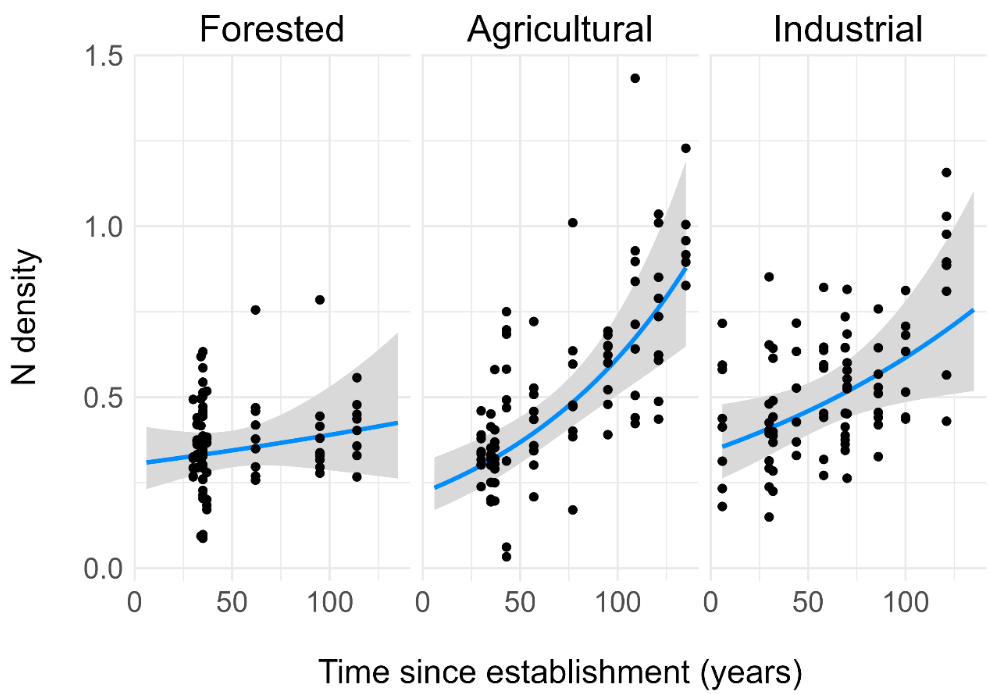

Figure 7: A plot contrasting the effect of age on nitrogen density (in kgm^{-2}) across three historical land-uses when all other variables in the linear model are held constant. Here, the response variable is exponentiated to back-transform the initial log-transformation. The shaded area represents confidence bands at $\alpha = 0.05$.

4. Discussion

Our study assessed whether historical land-uses have resulted in legacy effects on soil physico-chemical properties in contemporary green spaces. We sampled soil bulk density, soil heavy metals, and C and N stocks in green spaces across the Island of Montreal with three different historical land-uses (previously forested, previously agricultural, and previously industrial). We hypothesized that greenspaces with different historical land-uses would differ in all soil properties of interest, such that higher-intensity past land uses would have the largest impact. Our results showed that impacts of historical land-use on contemporary soils differed depending on the soil property studied. Irrespective of historical land-use, soil bulk density increased with age, suggesting an effect of contemporary management post park establishment rather than a legacy effect. Our nutrients analysis revealed that there was significant effects of historical land-use on carbon density, but no significant effects of historical land-use on nitrogen stocks. Specifically, C stocks in previously agricultural sites were marginally significantly different from that of previously forested sites. For both carbon and nitrogen, age was significant and the interaction term between historical land-use and age was marginally significant, implying a nuanced relationship between historical land-use and recovery from disturbance.

4.1 A brief discussion of historical land-uses of Montreal

All lands making up Montreal were once partitioned to lords via the seigneurial system of New France in the 17^{th} – 19^{th} century (The Canadian Encyclopedia). Lords would typically rent their land to tenants. In exchange, the tenants would pay their dues by working the land, a passive form of income serving as royalty to the lords. If land was not converted into agriculture, then it was kept natural and would be used, for example, as summertime residences (Ville de Montréal, 2007) or for hunting parties (Culture et Communications Québec; figure A4.1). Inadvertently, by serving an alternative purpose to agriculture, the "natural" areas, i.e., forests, were spared from land clearing. Upon the abolition of the lordship system in 1854, the lords' domains gradually went up for sale, fragmenting all of their lands – including the historical forests – into parcels. This land grabbing process created the remnant forests that were later purchased by the City of Montreal to be turned into city parks. Today, remnant forests maintain the same land cover and the spirit of land-use as it was historically, that is, as recreational areas.

Under the lordship system, agriculture was historically practiced for commercial gains though many toiled the earth for subsistence. Often, work-animals would be employed for ploughing, tillage, carrying, and load-bearing (Figure A4.2). Farmers were only able to intensify their practice beyond subsistence at the advent of mechanisation following 19^{th} century industrialisation. For example, in 1950, 54% of censused farmers in Quebec adopted mechanisation, in the form of tractor use, in their daily operations (Dawson & Fortier, 1954).

In the wake of the Second World War, Montreal began undergoing modernisation socially, economically, and industrially (Caron, 2016). The city began to rework much of its infrastructure to accommodate a growing population and an increasing dependency on the automobile. As such, highways and other major roads were being built. For example, the Metropolitan (Autoroute 40) that crosses the island east-west, was completed in 1958. Luckily, Montreal had ample and suitable bedrock at the surface to mine (Boyer et al., 1985), meaning that the large demand for raw rock and stone led to the establishment of many quarries. Throughout the 1960s, 1970s, and 1980s, these historical industrial sites were reappropriated into municipal landfills when the demand for construction materials was no longer needed, or the quarries were exhausted. When landfills hit maximum capacity, they were converted into city parks since no proper water lines could be installed for residential development due to possible contamination.

4.2 Soil bulk density

Our previously forested sites have long been used as nature spaces and refuges away from the bustling city life. As such, they likely have not had a recent land-use change, meaning that the low land-use intensity of the historical forests endured in the remnant forests. Our previously agricultural sites saw the traditional use of work-animals like horses, oxen and other cattle in subsistence farming. This could have historically compacted the soil via a trampling effect (Smith et al., 2016). Consulting the provincial law Q-2, r.19 provides glimpses on what historical waste management practices in the former quarries-turned-landfills were like at the time of legislation in 1972. Operators would create mounds and layers of soil over incoming waste at the end of every day throughout the lifespan of the landfill (Figure A4.3). They would deliberately compact the soil to effectively seal it to avoid any public health concerns. Across all of our studied historical land-uses, choices made in the past may have influenced the soil physical structure for years to come. However, despite these choices and contrary to our prediction, we did not detect a significant effect of historical land-use on current soil bulk density. Rather, across all three historical land-uses, soil bulk density increased significantly with park age (Figure 3). This is contrary to the findings of Golubiewski (2006), who found that younger soils were more bulky than older ones. We suspect that the opposite relationship present in our sites may be a result of human trampling from parkgoers compacting the surface soil at our sites. Parks that have been around longer, i.e., that are older, would have had more time for parkgoers to use them and thus trample the soil. This mechanism is reflected in Edmondson et al. (2011) who showed that soil bulk density was significantly lower in tree and shrub land-covers as well as non-domestic greenspaces, possibly due to lower public usage. Thus, a legacy effect has occurred, though not by historical land-use but from disturbances post the establishment of each historical site into municipal parks.

Moreover, we suspect that the age effect on soil bulk density in previously forested sites may be nuanced due to two factors: (i) the special designation of some of our previously forested sites and (ii) the way in which we aged our sites. First, half of our previously forested sites are not only municipal parks, but they are further designated as "nature parks." These sites receive a different type of management, one that is conservation-based. One such management practice is a trail network throughout the extent of the park that parkgoers are expected to abide by via a leave-no-trace policy. This would likely result in a reduced trampling effect in our study plots (which were off trail). Secondly, all of the nature parks we studied were officially established in the mid 1980s. As such, when determining the age of our study sites, we find that half of our previously forested sites are relatively young, or some 40 years old only. This confluence of our "youngest" previously forested sites also being least likely to experience heavily trampling may contribute to the positive relationship we see between age and bulk density in previously forested sites.

4.3 Soil heavy metals

Contrary to our expectations and the literature surrounding urban soil contamination (Clarke et al. 2015, Paltseva et al. 2022) soil heavy metals – As, Cr, Cu, Ni, Pb, and Zn – did not significantly differ between parks with varied historical land-uses. However, our sites exceeded the background levels of Southern Quebec (Figure A5.1), implying that our soils could be considered enriched. This may be partially explained by atmospheric deposition. Small particulates of heavy metals in the atmosphere are deposited on the soil surface and incorporated into the soil profile over time. Multiple anthropogenic sources exist, such as motor engine combustion (lead), galvanized fences and treated wood products corroding (copper + zinc) and metallurgy foundries (Alloway, 2010), and these point sources can generate volatile particulates in the atmosphere. These sources, which tend to be localized, are all sources that can be found in any urban environment, including Montreal, and can enrich the soil above background concentrations. For example, Fong et al. (2008) showed that urban heavy metal contents in Kuala Terengganu were both enriched in Pb, Cd, and Zn, and that these had to be due to human activity since no industrial activities exist in the Malaysian town. Luo et al. (2015) and Hu et al. (2018) have both shown soils enriched in heavy metals, with relative contributions of 51% and 63.8% from human-related activities, respectively, with 30.8% specifically arising from atmospheric deposition in the case of Hu et al. (2018). Moreover, proximity to roads has been demonstrated to correlate with an increase in soil heavy metal concentrations (Clarke et al., 2015). All of our sampled urban park sites were in proximity to roads of varying size and use frequency. Stormwater runoff can carry enriched road waste and pollution onto adjacent unsealed soils (Lorenz & Lal, 2009) such as parks. Assessing the effect of surrounding landscape variables, such as proximity and density of roads and other point sources, is an interesting area for future research.

4.4 Soil carbon

Contrary to our expectations, carbon density was marginally significantly greater in previously agricultural sites than in previously forested sites (Figures 4), supporting neither of our predictions. This finding matches the results of Pouyat et al. (2009) who found significantly higher percent SOC between residential turfgrass (which are comparable to both our previously agricultural and industrial sites) and urban forest remnants (which are comparable to our previously forested sites). In general, it is accepted that agriculture lowers carbon stocks due to the removal of biomass, i.e. cultivation, and the pulverization of organic matter in the upper 20-30cm of soils through tillage and ploughing (McLauchlan, 2006). Indeed, we know historically that many Canadian farms would habitually exhaust their lands prior to moving onto the next (Castonguay, 2018). However, other farms tended more to their soils by managing the carbon regime of their fields via fertilizers such as manure and other organic amendments (Castonguay, 2018). As fertilizer application was not standardized, amounts could have varied widely. Moreover, following the industrial revolutions of the 19th century, agricultural intensification took place. The shift from subsistence farming to mass-grown conventional farming would have introduced further variability in agricultural practices as farmers would have transitioned their practices towards mechanisation at varying rates (Dawson & Fortier, 1954). The large spread in the time of land-conversation of our previously agricultural sites may explain why the interaction between historical land-use and age was marginally significant for both measures of carbon. We speculate that some of the former agricultural sites were converted into urban parks prior to intensification, when agricultural sciences were not formalized and practices were based upon individual wisdom, whereas others were converted into urban parks after intensification, when farmers were perhaps more consistent in their approach to agriculture. In essence, the evolution of agriculture saw varied types and intensity of agricultural practices, resulting in an equally varied, but notable, legacy effect on carbon stocks in previously agricultural sites.

Carbon stocks increased with age in previously agricultural and industrial sites, implying that there is an accumulation (Figure 5). This result is concordant with other research in the literature, for example, that found significant increases of soil C concentrations with increasing greenspace age in the upper 10cm (Livesley et al., 2016) or that soil C increased with time since development in urban residential areas (Ziter & Turner, 2018). Ziter & Turner (2018) did not find any significant differences in soil C in open spaces, sites most comparable to ours; however, the researchers did not assess the effects of various historical land-uses, as we did, assuming that all of their sites shared a common agricultural historical land-use. In a meta-analysis, Phillips et al. (2023) conclude that SOC concentrations in turfgrass grasslands increased between 30 to 50 years following establishment but leveled-off and declined thereafter. Many of our sites are turfgrass dominated, though we did not see the same leveling-off or declining trend as described by Phillips et al. (2023). In their study of land-use impacts on SOC stocks, Edmondson et al. (2014) found that stocks were greater in domestic gardens than in public greenspaces, but that urban greenspaces still had greater SOC stocks than surrounding agricultural lands. Edmonson et al. (2014) raise the point that current intensive urban management practices such as fertilizing and mulching could explain the patterns of their results. Similarly, our results might not be explained by a legacy of a specific historical land-use, but rather by a legacy of previous disturbances at our study sites, and an accumulation of C post-establishment. For example, time of park establishment could represent a significant disturbance (land-cover change) in previously agricultural sites where cultivation suddenly ceased, so biomass no longer was removed from the system and thus carbon could recover with time. Previously industrial sites similarly would also have experienced a dramatic change in land-cover when landfilling and quarrying were replaced by a municipal park, presumably one that had to be fully conceived and managed post-establishment. Only in previously forested sites could a park have been established and then maintained with minimal disturbance – in line with our results that soil C remained consistent over time in previously forested parks, unlike in their previously agricultural or industrial counterparts.

4.5 Soil nitrogen

Contrary to our expectations, nitrogen stocks (Figure 6) did not significantly differ between parks with varied historical land-uses. We suspect that the lack of a strong legacy effect may be in part a result of our experimental design. In an urban environment, nutrient inputs are often buried and resurface due to human activity (Lorenz & Lal, 2009), so assessing changes in soil nutrients across varying depths could be an important factor in detecting land-use legacy effects. In a study much like our own, Raciti et al. (2011) compared the C and N stocks of residential lawns with various historical land-uses at four differing depths: 0-10cm, 10-30cm, 30-70cm, and 70-100cm. Raciti et al. (2011) found significant differences in carbon between the soils of residential lawns and their reference soils from forested areas at depths of 10-30cm and 30-70cm but not at the shallowest (0-10cm) nor deepest (70-100cm) parts. Falkengren-Grerup et al. (2006) showed that C and N pools were significantly different between two historical land-uses at depths of 10-30cm, but not past 30cm. This sounds contradictory; however, that nutrient stocks were significantly different in surface soils but not in subsurface soils was reasoned to be proof of a legacy effect of tillage practices since historical agriculture would only reach depths of 30cm. If not for this contrasting pattern in the soil profile, then Falkengren-Grerup et al. (2006) could not have established that there was any distinction in soil nutrient stocks between previously cultivated sites and continuously forested sites. What these studies illustrate is that considering a gradient of layers across the soil profile may increase the ability to detect effects of historical land-use on nutrients.

Age was a significant predictor of N stocks (Figure 7), which we speculate can be partly explained by atmospheric deposition. Urban centres are known to be hotspots for N deposition (Huang et al., 2015). In Baltimore, U.S.A, urban lawns continued to accumulate more atmospheric

N than did forests after 10, 70, and 365 days (Raciti et al., 2008). A literature review revealed that many anthropogenic sources such as vehicular emissions result in elevated local sources of various forms of nitrogen, for example, ammonia NH_3 (Decina et al., 2019). There have been studies showing a clear positive relationship between local traffic intensity, NH_3 emissions, and local deposition (Decina et al., 2017). As previously discussed, all of our sites were surrounded by roads, implying that the topsoil could be susceptible to N deposition from vehicles. In other words, sites converted earlier would have been exposed longer to atmospheric N deposition, thereby accumulating more nitrogen with age. Additionally, the management history of irrigation, mowing, and fertilization after park establishment, as well as animal urine and feces (Lorenz & Lal 2009, Hobbie et al. 2017), could also partly explain our result. Parks that have been converted and established earlier would have been exposed longer to these anthropogenic sources and accumulate nitrogen with age. This would be indicative of a legacy effect not of historical land-use, but of recovery from disturbances. Again, future work would require sampling soil across a depth gradient to test this hypothesis. For now, we can content ourselves with knowing that the urban parks of Montreal are seemingly not limited in nitrogen, so there would not be a need for any fertilizer use in our current-day management practices. This could help prevent any overloaded nutrients not captured in the soil from running off into the watershed (Hobbie et al., 2017).

5. Conclusion

Knowing what effects historical land-uses can have on soil properties can help city managers and urban planners better assess the state of urban greenspaces like public parks. In our research, we showed that historical land-uses can cause a legacy effect on soil carbon, where carbon stocks significantly increased with age, suggesting that the cessation of historical agricultural practices led to the system recovering in carbon. Though historical land-use was not a significant driver of soil bulk density, we see a trend with age such that older sites have denser soils. We know that dense soils can inhibit plant root growth and lessen nutrient, water, and air circulation due to reduced porosity (Kissling et al., 2009). If urban greenspaces grow to and continue to see high and frequent human traffic, then eventually this root-inhabited threshold might be reached. Contrary to our expectations, our results of no significant differences between six heavy metals support that conversion of historically industrial lands to parks did not result in long-lasting contamination of heavy metals, at least in the topsoil. Seemingly, previously industrial sites appear to have been adequately reclaimed: revegetation and conversion efforts could be considered a success and a fantastic legacy to leave for future generations. So, if city planners and managers wish to foster access to equitable greenspaces for residents in historically industrial neighborhoods, then they may be able to implement solutions like converting previously industrial sites to green spaces.

Carbon and nitrogen stocks both increased with time since park establishment. This implies that the history of land management following park establishment may be more important than the historical land-use, per se. In other words, how we choose to manage our urban greenspaces likely has important implications for the nature of its nutrient stocks. Knowing this, we can focus on finding the best practices in our mowing frequency, irrigation strategies, fertilizer use, and other current-day management in hopes of better governing our C and N stocks. Further, research into urban soil nutrients should consider site age – particularly for sites that have undergone past land use conversion – as it is likely to partially explain differences between sites. That fact that carbon and nitrogen are increasing over time in many of our urban greenspaces may have positive implications for ecosystem services provided by these parks. For example, carbon is linked to organic matter which is linked to decomposition whereas nitrogen is linked to nutrient cycling and is important for plant life that can provide us with even more ecosystem services (Roeland et al., 2019). Ultimately, controlling our present-day nutrient inputs in urban greenspaces means mitigating far-reaching future impacts such that urban-dwellers-to-come would inherit a legacy of sustainable land-use.

Chapter 2

Title: Assessing Land-use Legacy Effects on Earthworm Biodiversity in Urban Parks

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1. Introduction

Earthworms, considered ecosystem engineers (Jones et al. 1994, Jouquet et al. 2006), are important organisms to understand soil biodiversity, function, and ecosystem services. This is particularly true in regions of the world where earthworms are non-native – including in our study region in eastern Canada, where the dominant European peregrine species have been accidentally introduced as stowaways by colonials via the ballasts of their ships (Addison 2009, Tiunov et al. 2006, Wenzlhuemer 2020). On one hand, earthworms can fractionate organic matter such as leaf litter and render food items more accessible for microbe populations (Blouin et al., 2013) or increase soil aggregate size and porosity (Blouin et al., 2013), contributing to better water infiltration and flood mitigation. On the other hand, earthworms can also reduce native plant species richness and community composition in favour of exotic species (Craven et al., 2017), decrease the thickness of organic horizons and carbon stocks in boreal forest floors (Lejoly et al., 2021), and can even diminish Collembola and oribatid mites diversity, especially when endogeic and anecic earthworm ecotypes are present in high densities (Ferlain et al., 2017). Invariably, then, the presence of earthworms and their associated potential impacts are largely context-dependent (Hendrix & Bohlen, 2002). However, if earthworm biodiversity is variable across a given area – be it agricultural fields, forests, or urban centres – that could mean certain spaces would be disproportionally receiving the influence of earthworms. In contexts where earthworms confer a positive impact, an increased diversity and abundance of earthworms can be expected to exhibit increased soil functioning and services. Whereas in contexts where a negative impact is expected, spaces high in earthworm diversity and abundance can be expected to lead to declines in soil functioning and services. Thus, understanding the drivers of earthworm diversity and abundance in their non-native range is an important step in understanding the complex relationship between soil biodiversity and ecosystem functions and services.

Soil biodiversity, known for their contributions to multiple ecosystem functions (Delgado-Baquerizo et al., 2020), can be driven by historical land-uses. For example, in a Californian riparian ecosystem, Culman et al. (2010) showed that there was an overall consistent trend in reduced soil biodiversity along an agricultural intensification gradient. However, the researchers note that the majority of the variation in their data remained unexplained, and they speculated that historical management could play an important role in determining the community composition of nematodes and microbes in their study system. In a nature reserve in Southern Québec, Beauséjour et al. (2015) showed that the best predictor of earthworm invasion occurrence, intensity, and distribution of ecological types was distance to historical human disturbance. Historical land-use and historical development have even been proposed as possible filters that shape species pools in cities (Aronson et al., 2016). The effects of historical land-use on soil biodiversity can in turn influence important soil functions related to productivity (Barrios, 2007) and human health (Wall et al., 2015). Therefore, research at the intersection of land-use change and soil biodiversity has important ramifications for human wellbeing.

Land use intensity, an element often missing in biodiversity assessments (Dullinger et al. (2021), can also be a driver of soil biodiversity. For example, with increasing agricultural intensity, there was a decrease in Collembola functional richness (Yin et al., 2020), differences in the proportions of Acari and Collembola ecomorphological groups (Joimel et al., 2017), and a replacement of native epigeic earthworm species in the Columbian Andes by exotic counterparts better able to handle greater disturbance regimes (Feijoo et al., 2021). Postma-Blaauw et al. (2010) showed that larger-sized soil biota such as earthworms, enchytraeids, and nematodes were reduced in greater amounts relative to smaller-sized soil biota. In a network correlation analysis, Felipe-Lucia et al. (2020) showed that the positive correlation of provisioning services with both biodiversity and ecosystem functions declined at higher intensity levels. When conducting research on land-uses, especially land-use change, it is important to evaluate the intensity of the land-use, otherwise we risk missing a crucial driving factor of soil biodiversity and its connections to ecosystem functions and services.

Relationships between land-use change and soil biodiversity are particularly understudied in urban ecosystems, where biodiversity plays an important role in initiating people to environmental stewardship, improving human physical and mental wellbeing, and providing ecosystem services like mitigation against the heat island effect or buffering against flooding (Dearbon & Kark, 2008). Not all urban greenspaces are equal in terms of their biodiversity (Lepczyk et al., 2017), and urban biodiversity assessments typically focus on charismatic species, like birds and trees. Assessments of other organisms, such as soil microorganisms and soil invertebrates are rare (Knapp et al., 2021) partly due to the difficulty of working on a study system that is impractical, i.e., no ease of access to things that are belowground (Nielsen et al., 2011). Earthworms, for example, have been largely studied in natural and agricultural settings, but rarely in an urban context (but see Amossé et al. 2016, Smetak et al. 2007, and Toth et al. 2020). Studies within urban areas have found conflicting results on earthworm abundances and biomass (Guilland et al., 2018). Consequently, there is a need to better understand the drivers of soil biodiversity, especially those that are ecosystem engineers like earthworms, in urban greenspaces so that city planners and managers can better link biodiversity, ecosystem processes, and ecosystem services (Aronson et al., 2017). This includes the effects of historical land uses that are common across urban ecosystems and may represent an overlooked driver of current species richness and abundance.

Here, we investigated whether historical land-use type resulted in legacy effects on earthworm biodiversity within urban parks across Montreal, Quebec, Canada. We measured earthworm abundance, biomass, species richness, and community composition in urban green spaces that differed in their land-use history – specifically previously forested, previously agricultural, and previously industrial land uses historically common across this region. We hypothesized that historical land-uses would result in a legacy effect on all earthworm properties of interest due to differences in a relative degree of disturbance and land-use intensity prior to park establishment. Specifically, we predicted that previously forested sites, our sites with the least disturbance and the lowest land-use intensity, would exhibit the greatest abundance, biomass, species richness, and the most diverse community, followed by previously agricultural sites, then previously industrial sites.

2. Methods

2.1 General

2.1.1 Study area

Our study was conducted on the Island of Montreal, Quebec, Canada (45.5019° N, 73.5674° W). This includes the 19 boroughs that make up the City of Montreal as well as the 14 adjacent boroughs not under the administration of the City of Montreal. Montreal is characterized by a temperate climate as part of the Saint Lawrence Lowlands region of southern Quebec where temperatures annually range from -13ºC to 26ºC with a mean of 7ºC and receiving an annual precipitation mean of 48.3mm (Environment and Climate Change Canada). Soil formation regionally is mainly derived from glacial activity (see Chapter 1, Methods 2.1.1). Within Montreal we focused our efforts on a single contemporary land use – urban parks. These urban greenspaces were the most accessible for earthworm sampling and spanned a range of historical land-uses that we were able to reconstruct through publicly available records.

2.1.2 Historical land-use determination

We classified parks by their historical land-use based on the immediate previous land-use prior to their official designation as a city park by referring to many historical documents across a variety of outlets (see Chapter 1, Methods 2.1.2). In this way, we were able to determine three dominant historical land-uses in Montreal: forest remnants (forested), farms (agriculture), and quarries turned landfills (industrial). In cases where a single park had multiple historical land-uses, the respective area per historical land-use was delineated to avoid detecting mixed legacy effects.

2.1.3 Site selection

To limit vegetation effects on earthworms, where earthworm abundance, freshweight, and diversity could be driven by plentiful (high canopy-cover plots) or reduced (low canopy-cover plots) leaf litter food items, we restricted our sampling to medium canopy-cover plots at each site (see Chapter 1, Methods 2.1.4). These plots were the most consistent in terms of canopy-cover level across our sites, which allowed us to attribute differences in earthworms more confidently to land use legacy rather than current vegetation cover or vegetation effects on soil temperature and moisture (House et al. 1987, Fountain and Hopkin 2004). Moreover, we opted to omit sampling nature-parks as they differed most strongly in vegetation structure and diversity compared to other sampling sites. In wanting to maintain a balanced design, we randomly removed an equivalent number of sites from our two other historical land-uses, for a total of five sites per category (Table 1). In the end, we sampled three replicate quadrats of 25cm by 25cm per plot, amounting to 135 sampling points (see Figure 2 of Chapter 1, Methods 2.2.1).

Table 1: Selected Montreal study sites and their historical land-use prior to being designated as parks. Year that sites received official park designation are listed in brackets. Sites were chosen at random except for previously forested sites where nature-parks were specifically excluded.

Forested	Agricultural	Industrial
Angrignon (1927)	Angrignon (1927)	Botanical Gardens (1936)
Boisé-du-Saint-Sulpice (1990)	Centennial/Memorial Hall (1945)	Félix-Leclerc (1990)
Coulée-Grou (1988)	Fritz (1979)	Frédéric-Back (2016)
Jean-Drapeau (1908)	Marguerite-Bourgeoys (1913)	Père-Marquette (1953)
Thomas Chapais (1960)	Saint-Gabriel (1887)	Promenade Bellerive (1978)

2.2 Earthworm analysis

2.2.1 Collection

To sample earthworms, we used mustard extraction using a homemade vermifuge (Clapperton et al. 2008, East & Knight 1998). Alternative solutions such as formalin are environmentally damaging and dangerous to handle, and hand sorting the upper 20 – 30 cm of soil was ruled by land managers as too destructive an approach in public urban green spaces. Our homemade vermifuge was an aqueous solution mixed with 20g of dry mustard powder in 2L of water, for a desired concentration of 10g/L. This solution was then poured twice over a 15-minute period; half of the solution was poured initially, and the remainder at the halfway mark. Any earthworm individuals that surfaced within the boundaries of the quadrat and within this time frame were placed into plastic containers for transportation to the lab where they were weighted and identified.

2.2.2 Biomass determination

Immediately upon collection, we placed the earthworm specimens in a clear plastic container with a perforated lid with wet paper towel placed at the bottom to maintain moisture. Earthworms were left in these conditions in the lab for 48 hours to void their guts of food. After this time, we lightly rinsed individuals in running water and then weighed them, constituting their freshweight. Fresh biomass, in kgm^{-2}, was then calculated using the value of freshweight. Specimens were then stored in 100mL of 95% ethyl alcohol. Fresh biomass will hereafter be referred to as biomass.

2.2.3 Species identification

We sorted adults and juveniles based on an individual's sexual differentiation, i.e., if a clitellum was present or absent. Since juveniles are undifferentiated, they cannot be reliably taxonomically identified. Juveniles were therefore excluded from our biodiversity assessment, reducing our total pool of earthworm specimens from 616 down to 49. We identified the remaining 49 adult specimens according to Reynolds (1977) with emphasis on the type of prostomium, the segments that which the male pore and tubercula pubertatis are found, and the setal arrangement (Figure A6.1, A-D).

2.2.4 Covariates of interest

We measured several covariates of interest in the field whilst collecting earthworm specimens. Soil pH and temperature were measured with a PH8500 Portable kit by Apera Instruments. Soil moisture, measured in percent volumetric water content (VWC), was determined with a HydroSense II to a depth of 12cm. Additionally, other soil properties previously measured were considered as covariates of interest, namely total carbon and total nitrogen (see Chapter 1, Methods 2.2.4) as well as copper and zinc levels (see Chapter 1, Methods 2.2.3) which were deemed to exist at concentrations potentially exhibiting effects on earthworms in our study sites (CSQ, 2018).

We also developed a numerical index to attempt to quantify leaf litter palatability. First, all trees present in the sampling plots were identified and counted, for a total of 868 individuals making up 51 unique species. Based upon the works of Côté & Fyles (1994), Fyles (2022, unpubl.), and Hendriksen (1990), the palatability of almost all 54 species were noted categorically, from very low, low, moderate, and high. For buckthorn palatability, Heneghan et al. (2007) and Klionsky et al. (2011) were used as references. The palatability of eleven species could not be determined (Figure A7.1) and were thus omitted. Categories of palatability were transformed numerically, where very low = 1, low = 2, moderate = 3, and high = 4. We then calculated a weighted average by the number of individuals per sampling plot. We recognize that this index is only an approximation of leaf litter palatability and that our categorical scale was transformed into a numerical one.

2.3 Statistical analysis

2.3.1 Analysis pipeline

We analyzed our data by fitting general and generalized linear mixed-effects models (LMMs) using the package `lme4` (Bates et al., 2015). Sites were used across all models as a random effect to account for the non-independence of plots nested within sites. We determined model goodness-of-fit by calculating the conditional pseudo-R^2 values using `MuMIn` (Bartoń, 2023). To test the significance of the fixed effects, we used the "anova" function of the `lmerTest` package (Kuznetsova et al., 2017) with arguments specifying a Type III Analysis of Variance using the Kenward-Roger approximation as this was the recommend choice given our relatively small dataset (Luke, 2017). When the historical land-use predictor was found to be significant in a given model, we carried out post-hoc pairwise comparison of the model marginal means using the `emmeans` package (Lenth, 2023) to determine between which historical land-uses the predicted variable of interest significantly differed. Model diagnostics were performed to validate linear model assumptions. Normality, homogeneity of the residual variance, and independence were assessed visually using `performance` (Lüdecke et al., 2021) and `DHARMa` (Hartig, 2022) and, when unsure, via hypothesis testing (Levene's test and Durbin-Watson test). All data analyses were carried out in R version 4.3.2 (R Core Team, 2021). Prior to analysis, we explored the data following Zuur et al. (2010). Model effects were considered significant at an alpha of 0.05. Outcomes were considered marginally significant when 0.1 > p-value > 0.05.

2.3.1 Abundance

To test the effect of historical land-use on earthworm abundance, we fitted a GLMM with a negative binomial error structure. For this analysis, besides historical land-use, we included several other predictors that we suspected may influence earthworm abundance: soil moisture, soil pH, soil temperature, leaf litter palatability, soil heavy metal concentrations, and soil nutrient level. However, given our small dataset ($n = 45$), running a GLMM with all of these covariates of interest would lead to convergence issues likely due to model overfitting. Consequently, we set up a base model, M0, which contained worm abundance as a function of historical land-use, soil moisture, soil pH, and soil temperature. Subsequently, we set up five models based on a priori hypotheses, each containing the same fixed effects as the base model with the addition of one additional fixed effect (Table 2). Model 1 included the addition of litter palatability. Models 2 and 3 included the addition of soil copper and soil zinc, respectively. Models 4 and 5 included the addition of total soil carbon and nitrogen, respectively. In wanting to avoid any collinearity issues, we could not include soil copper and zinc together in the same model since they were highly correlated (Pearson's $r = 0.84$), which was also the case for soil total carbon and nitrogen (Pearson's $r = 0.89$).To determine whether the additional predictors of models 1-5 would lead to a significantly better fit than the base model, a likelihood ratio test was carried out between M0-M1, M0-M2, M0-M3, M0-M4, and M0-M5. None of the additional models performed significantly better than the base model; therefore, we selected this model (M0) as the best model. Model goodness-of-fit was evaluated through a Pearson's chi-squared test and by calculating the model deviance (Hilbe 2011, Zwilling 2013). Exceptionally, the significance of fixed effects was determined using the function "Anova" in the package `car` (Fox & Weisburg, 2019) under a Wald χ^2 distribution.

Table 2: Various models of interest in predicting earthworm abundance.

Model	Predictors	Model comparison
M0 (base)	Legacy (fixed) Soil moisture (fixed) Soil pH (fixed) Soil temperature (fixed) Sites (random)	N/A
M1	M0 Litter palatability (fixed)	LRT (M0, M1) p-value = 0.9203
M2	M0 Soil copper (fixed)	LRT (M0, M2) p-value = 0.1688
M3	M0 Soil zinc (fixed)	LRT (M0, M3) p-value = 0.8916
M4	M0 Soil total carbon (fixed)	LRT (M0, M4) p-value = 0.462
M5	M0 Soil total nitrogen (fixed)	LRT (M0, M5) p-value = 0.2695

2.3.2 Biomass

To test the effect of historical land-use on earthworm biomass, we fitted a LMM with a Gaussian error structure. Historical land-use and soil moisture were considered as fixed effects. For all other steps, see section 2.3.1.

2.3.4 Diversity

We chose to look at both species richness and community composition as two potential aspects of biodiversity that may be affected by historical land-uses. To assess whether historical land-use resulted in distinct earthworm communities, a distance (dissimilarity) matrix was constructed for a Non-metric Multi-dimensional Scaling (NMDS) ordination using `vegan` (Oksanen et al., 2022). This allowed us to visually assess if clustering was occurring in any of the three historical land-uses; in other words, to determine if a historical land-use resulted in a unique earthworm community. To further ascertain if there was clustering, we performed a permutation analysis of variance (PERMANOVA) and an analysis of similarities (ANOSIM) using Bray-Curtis distances. The PERMANOVA would confirm if the centroid of groups, i.e. historical land-use, are the same for all groups, whereas the ANOSIM would confirm if the similarity between sites given a historical land-use is greater than or equal to the similarity within each site given a historical land-use.

To determine if earthworm species richness differed among historical land-uses, we constructed sample-based abundance accumulation curves which we then rarified. Rarefaction allows for an appropriate comparison of species richness across our three historical land-uses given that we observed a different number of individuals and species collected at each historical land-use (Gotelli & Colwell 2001, Willis 2019). We can extrapolate the rarefaction curves (Chao et al., 2014) to determine if our sampling effort was sufficient by noting if the curves reach an asymptote. This would suggest whether we collected the total expected earthworm species richness per historical land-use or not. Rarefaction and extrapolation were done using `iNEXT` (Hsieh et al., 2022). Significant differences were determined visually when confidence intervals were not overlapping (Colwell et al., 2012).

3. Results

3.1 Abundance

On average, previously forested sites had the greatest earthworm abundance and greatest variance (Figure 1). Historical land-use was a marginally significant predictor of earthworm abundance (p-value = 0.052), implying a preference for this habitat. Moreover, soil moisture (p-value = 0.0003) and pH (p-value = 0.037) were also significant predictors of earthworm abundance. Soil moisture had a positive effect on earthworm abundance (Figure 2) whereas soil pH had a negative effect on earthworm abundance (Figure 3). The effects of both soil moisture and soil pH did not vary across historical land-uses. Model deviance was 41.70 and $1 - \chi^2 = 0.27$.

Table 3: Comparison of descriptive statistics, mean (± standard deviation), of earthworm abundance and biomass in our study relative to other urban studies across the globe.

Study	Sampling site	Sampling location	Earthworm abundance (indiv.m^{-2})	Earthworm biomass (gm^{-2})	Species richness
This study	Urban parks	Montreal, Canada	13 (± 12.4)	37 (± 29)	Between 1.53 and 5
Smetak et al., 2007	Urban parks	Moscow, USA	437	94.12	1 to 3
Pizl et al., 2007	Urban parks	Brno, Czechia	121 (± 32)		6
Tiho and Josens, 2000	Grassy road median	Brussels, Belgium	152.24	91.44	–
Xie et al., 2018	Urban parks	Beijing, China	44.6 (± 39.1)	15.6 (± 14.0)	3.08

Table 4: Output of post-hoc "Anova" function on earthworm abundance LMM. Previously forested sites are set as the default contrast. Asterisk (*) indicate significance of fixed effects at $\alpha = 0.05$ and crosses (†) indicate marginal significance.

Predictors	χ^2	Degrees of freedom	p-value
Intercept	5.3020	1	0.0213*
Legacy	5.9200	2	0.0518†
Soil moisture	13.2301	1	0.0003*
Soil pH	4.3553	1	0.0369*
Soil temperature	1.6021	1	0.2056

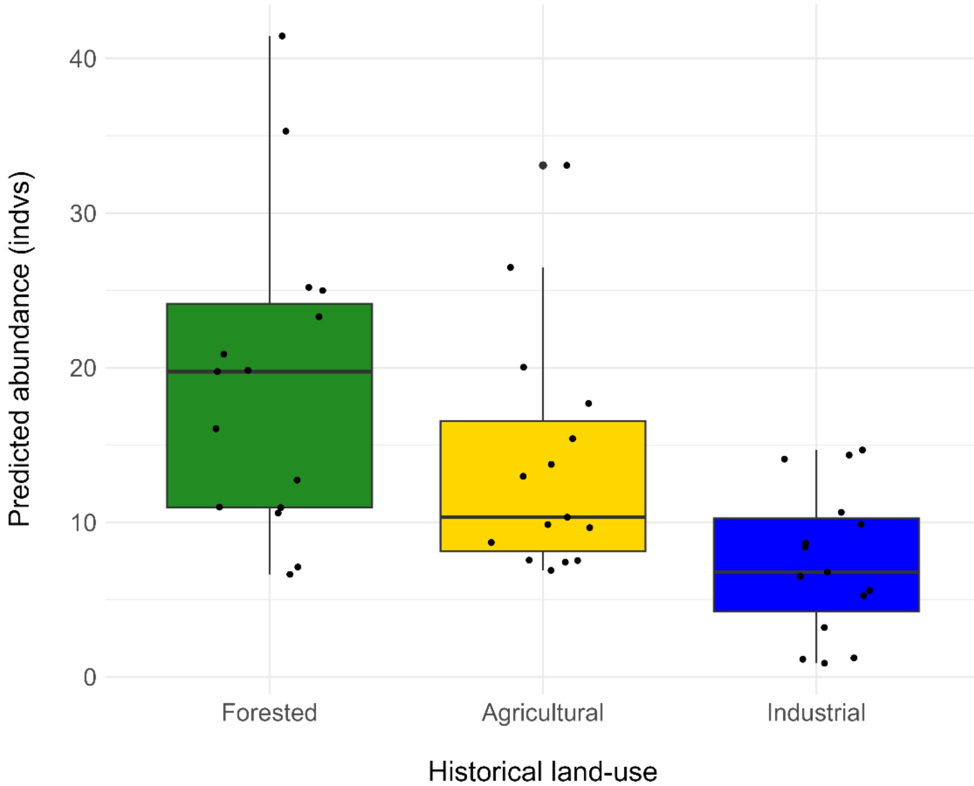

Figure 1: Boxplot of model predictions of earthworm abundance across three historical land-uses.

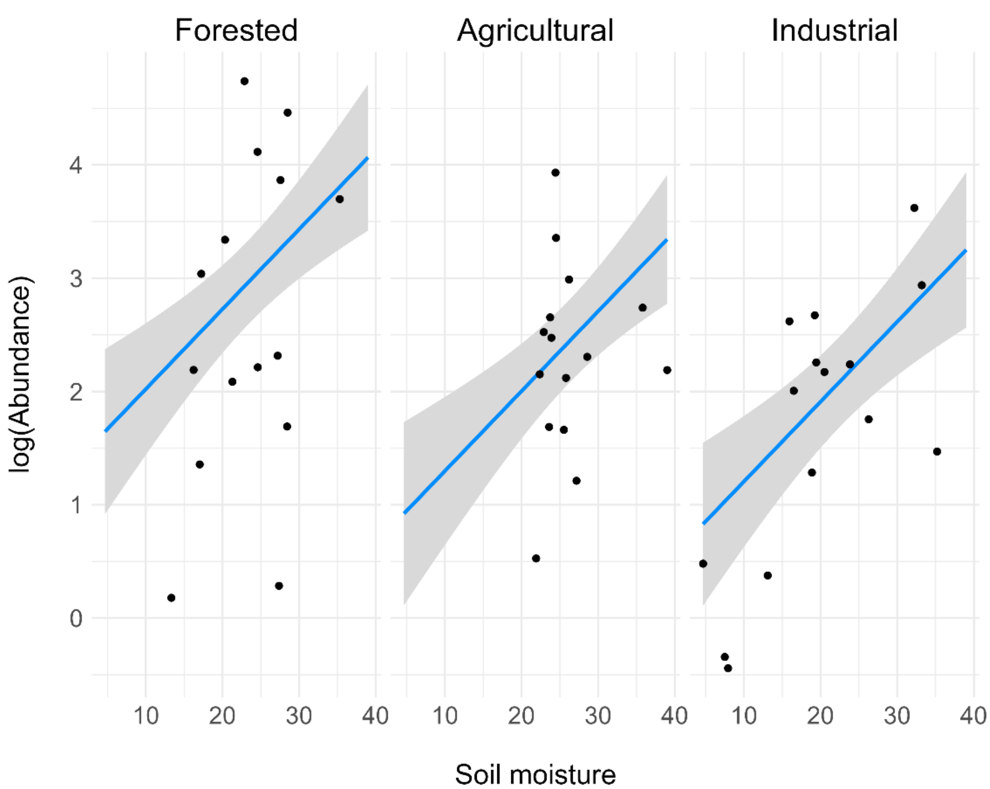

Figure 2: A plot contrasting the effect of soil moisture, measured as volumetric water content (in percent), when all other variables in the linear model are held constant. Soil moisture has a positive effect on earthworm abundance that does not differ across historical land-use. Here, the response variable is log-scaled because of the inherit link function in the GLMM. The shaded area represents confidence bands at α = 0.05.

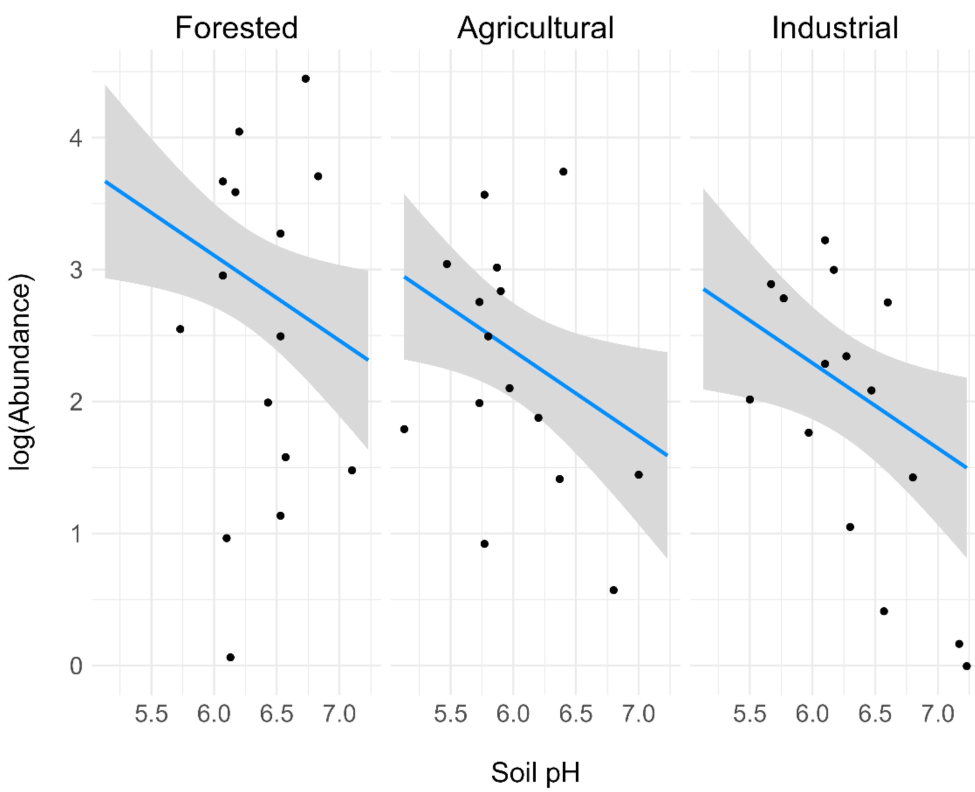

Figure 3: A plot contrasting the effect of soil pH when all other variables in the linear model are held constant. Soil pH has a negative effect on earthworm abundance that does not differ across historical land-use. Here, the response variable is log-scaled because of the inherit link function in the GLMM. The shaded area represents confidence bands at α = 0.05.

3.2 Biomass

The predicted biomass of previously forested sites was, on average, nearly twice that of the predicted biomass of earthworms collected from previously industrial sites (Figure 4). However, historical land-use was not a significant predictor of earthworm biomass (Table 5). Soil moisture was marginally significant (Table 5) and presented a similar trend across all three historical land-uses (Figure 5). The model had a pseudo conditional $R^2 = 0.48$.

Table 5: Output of post-hoc "anova" function on earthworm biomass LMM. Previously forested sites are set as the default contrast. Asterisk (*) indicate significance of fixed effects at α = 0.05 and crosses (†) indicate marginal significance.

Predictors	Sum of Squares	Mean square	Numerator df	Denominator df	F value	p-value
Legacy	0.0007	0.0004	2	11.959	0.8362	0.4572
Soil moisture	0.0016	0.0001	1	37.233	3.6178	0.0065†

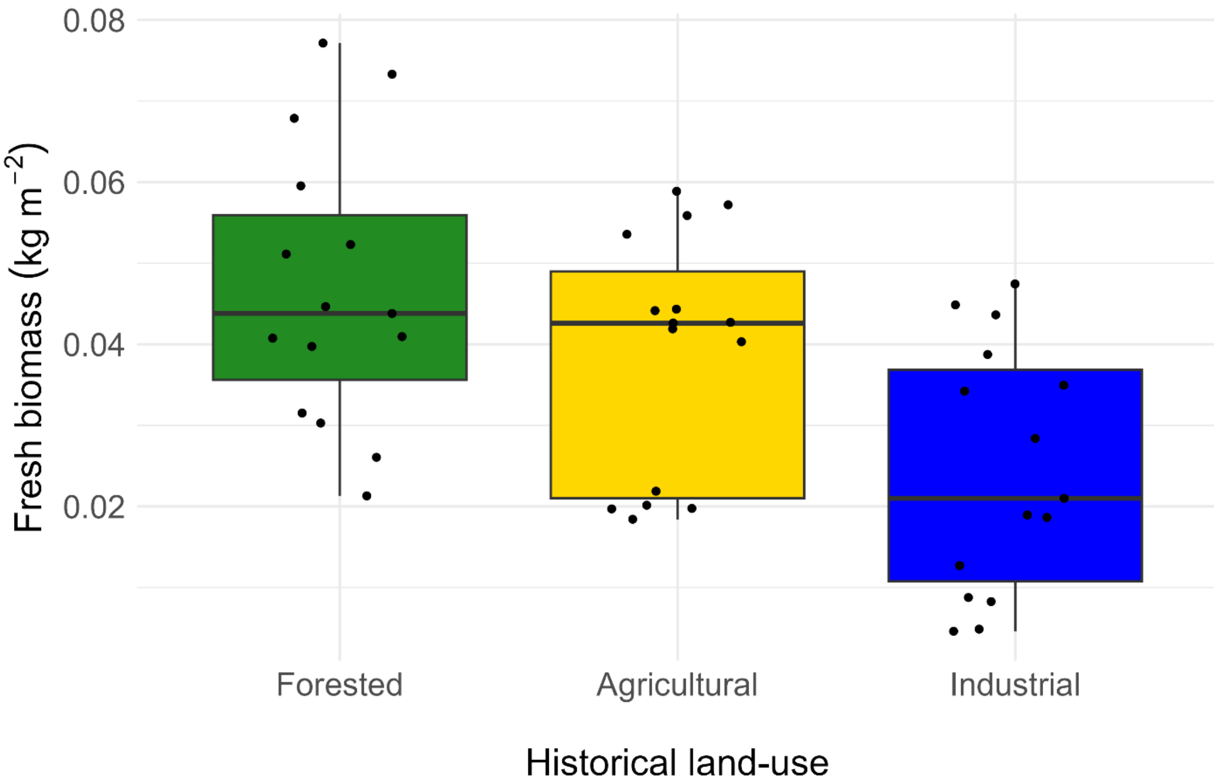

Figure 4: Boxplot of model predictions of earthworm fresh biomass across three historical land-uses.

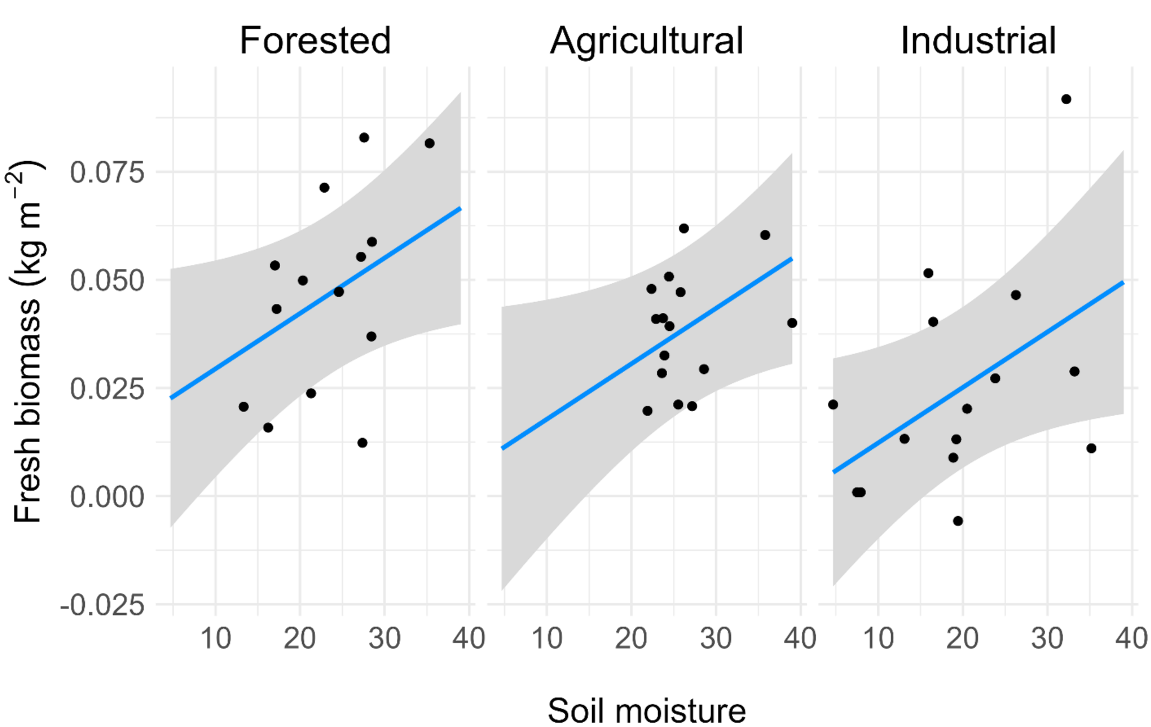

Figure 5: A plot contrasting the effect of soil moisture, measured as volumetric water content (in percent), when all other variables in the linear model are held constant. Soil moisture has a positive effect on earthworm fresh biomass that does not differ across historical land-use. Here, the shaded area represents confidence bands at α = 0.05.

Biodiversity

In total, six species were identified with most earthworms collected being *Lumbricus terrestris* (n = 30, 61% of adult specimens) and of the anecic ecotype (Figure 6). Six species of earthworms collected is relatively low in comparison to similar studies conducted in urban environments. For example, Amossé et al. (2016) collected 16 earthworm species in Neuchâtel, Switzerland. Moreover, when also considering similar studies conducted elsewhere than urban environments, our number of collected species still remains low in comparison. For example, Eggleton et al. (2009) collected 15 species in pasture woodlands of southern England, Feijoo et al. (2011) collected 26 species in agroecosystems in Columbia's Andean region, and Szlavecz & Csuzdi (2007) collected 12 earthworm species in forests of eastern Maryland, USA. Both a PERMANOVA (p-value = 0.58) and an ANOSIM (p-value = 0.42) confirmed that there were no statistically significant differences in the earthworm communities between historical land-uses.

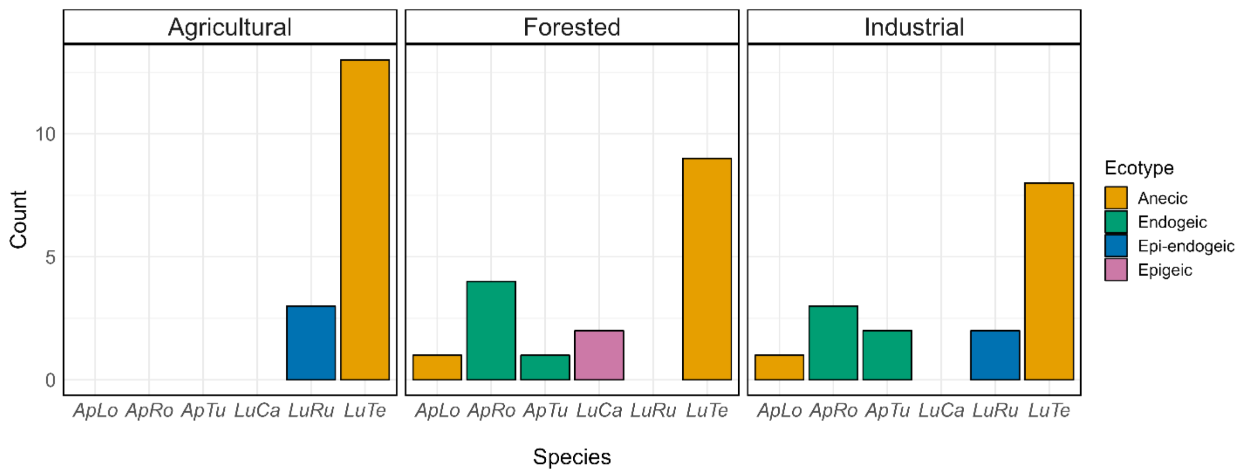

Figure 6: Earthworm demographics (species and ecological types) collected per historical land-use. Species codes: *Aporrectodea longa* (ApLo), *Aporrectodea rosea* (ApRo), *Aporrectodea tuberculata* (ApTu), *Lumbricus castaneus* (LuCa), *Lumbricus rubellus* (LuRu), *Lumbricus terrestris* (LuTe).

At minimum sampling coverage, previously industrial sites observed a high species richness. From rarefaction curves (Figure 7), it was determined that species richness in previously industrial sites were, in fact, nearly equivalent to that of previously forested sites, with species richness's of 4.68 and 5.00, respectively. The species richness in previously agricultural sites was significantly lower than previously forested and industrial sites. Earthworm diversity in agricultural sites was sufficiently sampled, as seen by the rarefaction curve reaching its asymptote (Figure 7). Earthworm diversity in industrial sites began to reach asymptote in the extrapolation (dashed line), implying that we were close to fully sampling the expected species present. Earthworm diversity in forested sites, however, continues to rise in the extrapolation, demonstrating that these sites would have required more sampling as more individuals would likely have yielded to new species being discovered and thus increasing diversity.

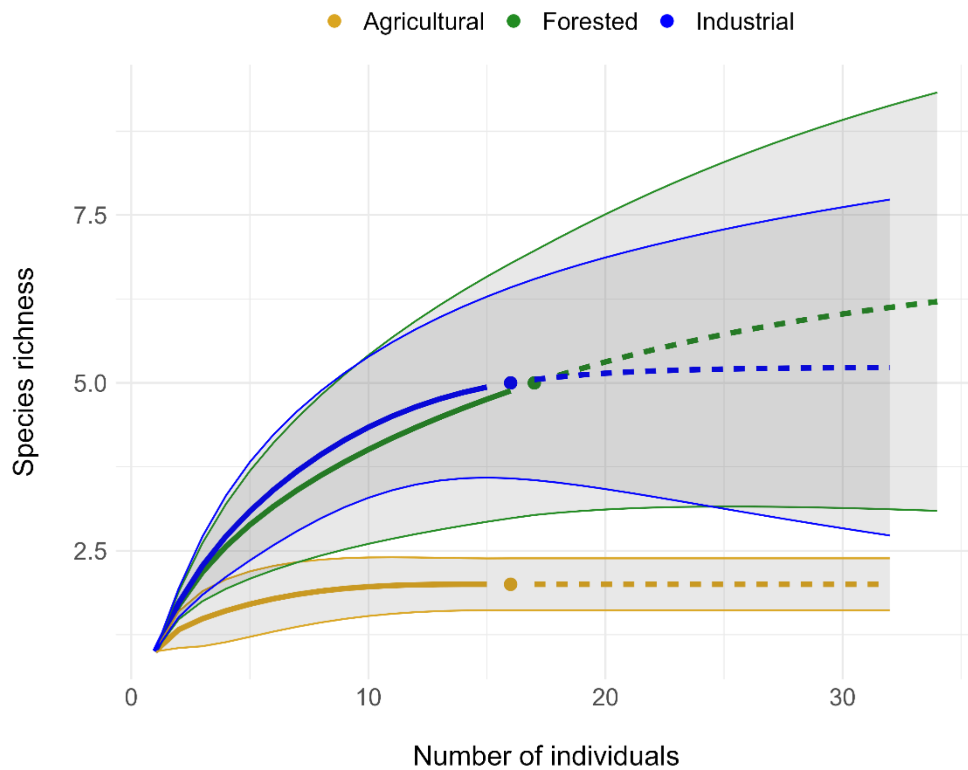

Figure 7: Rarefaction curves with 95% shaded confidence intervals of earthworm biodiversity across three historical land-uses standardized at minimum sample coverage.

4. Discussion

Our study assessed whether historical land-uses resulted in legacy effects on earthworm community and population indicators. Earthworms are considered ecosystem engineers able to profoundly alter ecosystem processes in both positive and negative ways. We sampled sites within contemporarily similar urban parks with three different historical land-uses (previously forested, previously agricultural, and previously industrial) to assess legacy effects on earthworm abundance, biomass, species richness, and community composition. Historical land-use was marginally significant for earthworm abundance but not for earthworm biomass. The community composition of earthworms across historical land-uses was not significantly different but previously agricultural sites had a significantly lower earthworm species richness. Though most of our findings are non-significant, we argue that our hypothesis is partially supported as some biological patterns are present within our results that are indicative of legacy effects, especially when in the context of the literature (Table 3).

4.1 Abundance and biomass

Some soil edaphic properties, such as soil moisture and soil pH, were found to be significant predictors of earthworm abundance. Soil moisture did not vary across historical land-use and was positively related to earthworm abundance (Figure 2). Soil pH also did not vary across historical land-use and was negatively related to earthworm abundance (Figure 3), though the pH range of the soils in our study were in the reported ranges known to be favourable for earthworms (Reynolds 1994, Curry 2004). These finding are strongly supported by the literature (see Tiunov et al. 2006 and references therein, Eggleton et al. 2009, Xie et al., 2018). In our study, soil moisture was similarly found to be marginally significant in predicting earthworm biomass, where biomass (Figure 5) was positively related to soil moisture. This is unsurprising as earthworms require moisture to survive and so water would rightfully be a large component determining their bodyweight and by extension their biomass.

Historical land-use, however, was not found to be a significant predictor of earthworm abundance; though, we argue that there is a trend partly indicative of a legacy effect by historical land-use via land-use intensity, with previously forested sites maintaining higher earthworm abundance than agricultural and industrial. This trend is supported by a meta-analysis showing that land-use extensification has a positive effect on earthworm populations, with declining land-use intensity resulting in larger earthworm community sizes (Spurgeon et al., 2013). Another study showed how distance to disturbance was an overall driver of earthworm invasion occurrence, intensity, and distribution of ecological types (Beauséjour et al., 2015). Specifically, endogeic and anecic ecotypes such as *Apporectodea* and *L. terrestris* were significantly predicted by proximity to historical human disturbance and presence of leaf litter. While the dominance of anecic ecotypes in our data (Figure 6) and the greater abundance of earthworms in our site with the least historical intensive land-use aligns with the literature, we feel that there another element might also explain our results. Ziter et al. (2021) showed that invasive jumping worms *(Amynthas spp.)* preferred forested sites but not grasslands. This pattern aligns somewhat with our results showing previously forested sites – which have often maintained more forest-like characteristics – were more abundant in earthworms, on average, than previously agricultural and industrial sites (Figure 2). Previously agricultural and industrial sites are also currently dominated by turfgrass grassland, even at similar levels of canopy cover (pers. obs.). Moreover, the identity of trees in previously forested sites might have historically differed from that of previously agricultural and industrial sites since previously forested sites would have maintained forest-like characteristics for longer. This, rather than tree diversity, might have favoured more appetizing and palatable leaf litter for earthworms (Schwarz 2015, de Wandeler 2018) and thus have driven historical patterns of abundance, too.

Historical points of invasions can be legacies that explain current-day earthworm abundance. For example, Ziter et al. (2021) alluded to a legacy effect of points of invasion in explaining the distribution patterns of the recently invasive Asian jumping worm. Historically, we know that European exotic earthworms present today in Montreal might have been introduced accidently via the ballasts of ships (Wenzlhuemer, 2020). Ship embarkments would have taken place along the coasts of the island, where we happen to find many of our previously forested sites (see Figure 1 of Chapter 1). Coupled with human fishing activities that may also have historically taken place along the shores and spread earthworms via abandoned bait (Cameron et al., 2007), we assume that historical point of invasions in Montreal might similarly explain some of the patterns observed in our earthworm abundance results, especially in our previously forested sites. Therefore, a legacy effect would be present, but it would be due to spatial factors than to historical land-use per se.

Though previously forested sites did maintain the highest average earthworm abundance, we speculate that earthworm abundances in previously agricultural and industrial sites were not significantly lower than previously forested sites, as we had initially predicted, because of possible importations. Historically, some farmers may have intentionally brought anecic and endogeic species over from Europe (Crumsey et al., 2014) as these species provide benefits in tilled and ploughed agroecosystems, such as mixing nutrients and improving water infiltration from their burrowing activity (Spurgeon et al., 2013). Thus, historically agricultural fields could have also served as initial points of invasions. Likewise, in previously industrial sites, higher-than expected earthworm abundances could have occurred from human mediated-dispersal. In a city, where urban planning, construction, and landscaping all take place, there are large volumes of soil being transported, translocated, and buried. Earthworms residing in these transferred soils could therefore passively disperse in two ways. First, as cocoons, they may stick in between the tire threads of vehicles, a dispersal mechanism known as anthropochory (Cameron 2007, Addison 2009). Second, as adults, they may physically piggyback in the soil profile and emerge elsewhere once displaced. It is known that historical industrial land-use in Montreal surely entailed much soil importation to be in conformity with environmental laws at the time (see Chapter 1, Discussion 4.1). In so doing, the machinery used for moving soil could inadvertently introduce earthworms in the landfills. Additionally, given the right topography, runoff water could have also carried surface cocoons, casts, and even individuals into the giant former open-pit quarries as they were being filled (Eijsackers, 2011). This same mechanism could also explain the surprisingly high species richness (Figure 7) in previously industrial sites. By virtue of historical land-use management decisions, we expect that earthworm abundances in previously agricultural and industrial sites are both the inadvertent and intentional products of our past choices, i.e., specimen and soil importations, and thus likely a legacy effect.

Ultimately, irrespective of the underlying factors accounting for earthworm distributions in Montreal, we see that there is a greater abundance on average of earthworms in previously forested sites than previously agricultural and industrial sites. This can be foreboding and cause for some alarm because earthworms are deemed invasive in similar ecosystems, such as temperate and boreal forests, because of their ability to modify forest floor dynamics. For example, they can significantly reduce the humic layer, bury organic matter, and change soil conditions such that they are less preferable for other soil organisms (Hendrix & Bohlen, 2002 and references therein). Or, earthworms can facilitate the introduction of other exotic species, such as buckthorn, which itself can promote soil conditions more apt for earthworms, and before long an invasional meltdown occurs (Heneghan et al., 2007). Therefore, earthworms could potentially negatively impact the functioning of soils in our urban forests. By extension, these effects could trickle back up to aboveground communities (Wardle et al., 2004) if invasion species like buckthorn are facilitated, and potentially impact human health if other soil organisms, such as those that regulate soil-born pathogens (Wall et al., 2015), are pushed out by earthworms. Future studies should explore the impacts of earthworms on urban forest soil properties and biodiversity so that we can assess whether the ecosystem services provided by these forests are affected.

4.2 Earthworm diversity

Historically forested sites had an earthworm species richness similar to that of historically industrial sites (Figure 8) but comprised of communities with lower variation across sites (Figure 7). We suspect that there are multiple mechanisms at play which might explain our results: (i) land-use type and intensity, (ii) vegetation effects, and (iii) limitations of a small sample size. First, the past – and continued – low intensity pressure of forested land-use could favour a greater diversity of earthworms, especially epigeic ones (Talavera et al., 2020). For example, the only epigeic species that we detected, *Lumbricus castaneous*, was found at previously forested sites (Figure 6). Moreover, historical industrial sites were sites prone to much soil transfer and movement (see Chapter 1, Discussion, Section 4.4). Various types of earthworms might have been accidentally introduced in between quarrying backfill and landfill exploitation. Second, plants could be significant predictors of earthworm diversity. For example, the successional status of forests can determine the proportion of earthworm ecotypes (Szlavecz & Csuzdi, 2007) because of shifting plant community composition over time. A mature, climax forest would have leaf litter of lesser palatability, like oak, whereas a forest in transition would have mid-successional species like ash, linden, and beech. Similarly, we see that leaf litter quality of specific tree species, like the Norway spruce and European larch, could negatively affect earthworm biomass, density, and species richness (Schwarz et al., 2015). These two species were particularly present at previously agricultural and industrial sites (pers. obs.). Third, we note that our rarefaction results were based on a small sample size ($n = 49$). The interpretations we can make are, therefore, limited. We expect that with a larger sample size, a larger effect would have been detected where previously industrial sites would not have had similar earthworm species richness to previously forested sites as seen by the growing extrapolation lines in our rarefaction curves (Figure 7). Thus, on one hand, our first (i) supposition would be the result of past land-use and thus a legacy effect. On the other hand, our second supposition (ii) would be a contemporary effect. However, the current-day vegetation itself could be driven by historical land-use (Roman et al. 2018, Munteanu et al. 2015, Pregitzer & Bradford 2023, Yang et al. 2017). Thus, direct and indirect mechanisms of historical land-use may have shaped earthworm species richness and community composition in both historically forested and industrial sites.

We speculate that historical agricultural management decisions are responsible for the present-day low species richness and anecic dominance of earthworms in previously agricultural sites. Our results support this legacy effect since we only detected two species in previously agricultural land-use: two *Lumbricus* species, one anecic and one endo-anecic (Figure 6). This type of earthworm coincides with ideal ecotype that farmers at the time might have deliberately introduced in their fields for their ability to mix nutrients and improve water infiltration (Crumsey et al. 2014, Spurgeon et al. 2013). Anecic ecotypes are also well suited to endure the disturbance regime of tilling and ploughing by escaping below the disturbance lines, and thus can adequately compete in such an environment. Farmers may thus have filtered the earthworm species richness and composition pools historically such that we still observe lower diversity today.

5. Conclusion

Here, we set out to assess the effects of historical land-use on earthworms in an urban setting, where such invertebrates are often underrepresented (Knapp et al., 2021). Specifically, we described components of earthworm distribution, abundance, and biodiversity. Our results were indicative of potential land-use legacy effects. Earthworm species richness and community composition may be explained by unchanged forested historical land-use, by historical agricultural practices, and by industrial backfilling practices. Earthworm abundance was greater than expected in previously agricultural and industrial sites, though remained greatest on average in previously forested sites. Historical forests seemed to be exotic earthworms' preferred habitats, though the spatial arrangement of past forests in Montreal, being mostly riverain, could have coincidentally made them more susceptible to earthworm introductions. Having earthworms present in higher numbers in remnant forests could have a negative impact, though this would have to be further verified. Continuing investigations in historical ecology will strengthen the science of our urban studies by acknowledging the impacts that our past land-uses can have on the land and the life the land harbours. If urban centers are projected to expand with growing urban populations, then our cities must be equipped in the future with the knowledge that our land-uses today could determine future soil biodiversity.

General Conclusions

Both chapters of this thesis explored the effects of historical land-use prior to urban park establishment on soil physico-chemical properties and earthworm biodiversity. Our results demonstrated that there were legacy effects of historical land-use on soil carbon stocks and earthworm species richness. Albeit not being statistically significant, we saw patterns indicative of legacy effects by historical land-use on earthworm abundance and earthworm community composition. Only heavy metals did not notably differ across historical land-uses. Overall, aspects of soils of our urban greenspaces seem to be minimally susceptible to legacy effects of historical land-uses.

Soils were not more contaminated in heavy metals across sites with historically more intensive in land-uses. This implies that that remediation efforts to convert highly intensively used historical sites, like those previously industrial, to urban green spaces worked well. However, soil bulk density increased with time, potentially due to a human trampling effect by parkgoers. If urban greenspaces continue to be heavily used and enjoyed recreationally in the future, then city planners and managers should be mindful of growing compacted soils and explore management practices to reduce this compaction. Similarly, carbon and nitrogen stocks both increased with age. Accumulation of carbon over time is highly desirable, especially if the rate and volume of sequestration can be shown to help climate regulation amidst a global warming crisis. Increasing nitrogen in urban centres, however, can be cause for concern. Since soils are some of the last permeable areas in our cities, any anthropogenic emissions will result in atmospheric deposition and become incorporated into the soil. Eventually, urban soils might become overloaded in nitrogen. Consequently, soils could leach out and lose nutrients, which could cause down-the-line issues for the overall soil chemistry and, by extension, soil ecosystem functioning. Our work has therefore shown that, when measuring urban soil properties, it is not enough to simply capture the value of a property in a single snapshot to then make some statement about it. It is important to consider how these properties change and evolve over time, especially because humans are a major driving force of these properties via current land-uses, which are intensifying, and past land-uses, which can be persistent and cause legacy effects. Without accounting for land-use history, studies may miss important drivers of soil nutrient dynamics crucial for down-the-line ecosystem functions and services.

In addition to soil properties, our results suggest impacts of historical land use on earthworm biodiversity. Previously agricultural sites had significantly lower numbers of earthworm species, but previously industrial sites had a species richness equally high as that of previously forested sites. This implies that historical industrial activities such as quarrying, landfilling, and construction all potentially introduced a variety of species to sites through human activities. Knowing this has two-fold implications. First, industrial land-uses will always be a part of a city's reality. A city could not maintain itself if, for example, there were no construction projects to build new developments or upkeep older ones. Thus, current industrial land-uses, just like historical ones, can potentially introduce multiple species of an exotic organism such as earthworms. We should be mindful of this because we would not want to blindly proliferate the spread of an ecosystem engineer able to profoundly – both negatively and positively, context dependent – affect soil ecosystem functions. Second, future studies on urban biodiversity, especially those attempting to broaden the taxonomic scope like ours, should consider the role of historical land-use in their assessments. Often, urban biodiversity is quantified and explained in ecological studies without consideration for the historical context surrounding the environment in which the study was taking place. In an urban area, where land-uses are varied both in space and time, acknowledging the importance of historical land-use could only lead to ecological studies better explaining patterns in our urban biodiversity with multi-disciplinary perspectives that match the socio-ecological landscapes that are our cities.

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Appendix I: Field sampling plots

Figure A1.1: Example of a low canopy sampling plot dominated by mowed turfgrass.

Figure A1.2: Example of a medium canopy sampling plot dominated by meadow grasses.

Figure A1.3: Example of a high canopy sampling plot dominated by trees.

Appendix II: List of sites

Table A2.1: Sites with historical agricultural land-use.

Angrignon	Cap-Saint-Jacques	L'Anse-à-l'Orme	Lafontaine	Marguerite Bourgeoys
1-HIGH01	2-HIGH02	4-HIGH03	5-HIGH03	6-HIGH01
1-HIGH03	2-HIGH03	4-HIGH05	5-HIGH05	6-HIGH02
1-HIGH04	2-HIGH05	4-HIGH06	5-HIGH06	6-HIGH03
1-LOW01	2-LOW03	4-LOW01	5-LOW03	6-L0W02
1-LOW02	2-LOW04	4-LOW02	5-LOW04	6-LOW03
1-LOW04	2-LOW05	4-LOW05	5-LOW06	6-LOW04
1-MED01	2-MED02	4-MED02	5-MED01	6-MED02
1-MED02	2-MED04	4-MED05	5-MED04	6-MED03
1-MED06	2-MED05	4-MED06	5-MED05	6-MED05

Angrignon	Cap-Saint-Jacques	L'Anse-à-l'Orme	Lafontaine	Marguerite Bourgeoys
1-HIGH01	2-HIGH02	4-HIGH03	5-HIGH03	6-HIGH01
1-HIGH03	2-HIGH03	4-HIGH05	5-HIGH05	6-HIGH02
1-HIGH04	2-HIGH05	4-HIGH06	5-HIGH06	6-HIGH03
1-LOW01	2-LOW03	4-LOW01	5-LOW03	6-L0W02
1-LOW02	2-LOW04	4-LOW02	5-LOW04	6-LOW03
1-LOW04	2-LOW05	4-LOW05	5-LOW06	6-LOW04
1-MED01	2-MED02	4-MED02	5-MED01	6-MED02
1-MED02	2-MED04	4-MED05	5-MED04	6-MED03
1-MED06	2-MED05	4-MED06	5-MED05	6-MED05

Table A2.2: Sites with historical forested land-use.

Angrignon	Bois-de-Liesse	Bois-de-Saraguay	Boisé-du-Saint-Sulpice	Cap-Saint-Jacques
1-HIGH01	2-HIGH01	3-HIGH01	4-HIGH02	5-HIGH02
1-HIGH03	2-HIGH03	3-HIGH03	4-HIGH03	5-HIGH03
1-HIGH04	2-HIGH06	3-HIGH06	4-HIGH04	5-HIGH05
1-LOW01	2-LOW01	3-LOW03	4-LOW01	5-LOW01
1-LOW02	2-LOW02	3-LOW04	4-LOW02	5-LOW02
1-LOW05	2-LOW04	3-LOW05	4-LOW04	5-MED01
1-MED01	2-MED01	3-MED03	4-MED01	5-MED02
1-MED02	2-MED03	3-MED04	4-MED02	5-MED04
1-MED05	2-MED04	3-MED05	4-MED05	

Coulée-Grou	Jean-Drapeau	L'Anse-à-l'Orme	Pointe-aux-Prairies	Thomas-Chapais
6-HIGH01	8-HIGH02	9-HIGH01	10-HIGH02	11-HIGH01
6-HIGH02	8-HIGH03	9-HIGH02	10-HIGH04	11-HIGH02
6-HIGH03	8-HIGH04	9-HIGH03	10-HIGH05	11-HIGH06
6-LOW02	8-LOW02	9-LOW01	10-MED01	11-LOW01
6-LOW03	8-LOW03	9-LOW02	10-MED03	11-LOW04
6-LOW05	8-MED01	9-LOW04	10-MED05	11-LOW06
6-MED01	8-MED02	9-MED01		11-MED03
6-MED03	8-MED03	9-MED02		11-MED04
6-MED05		9-MED03		11-MED05

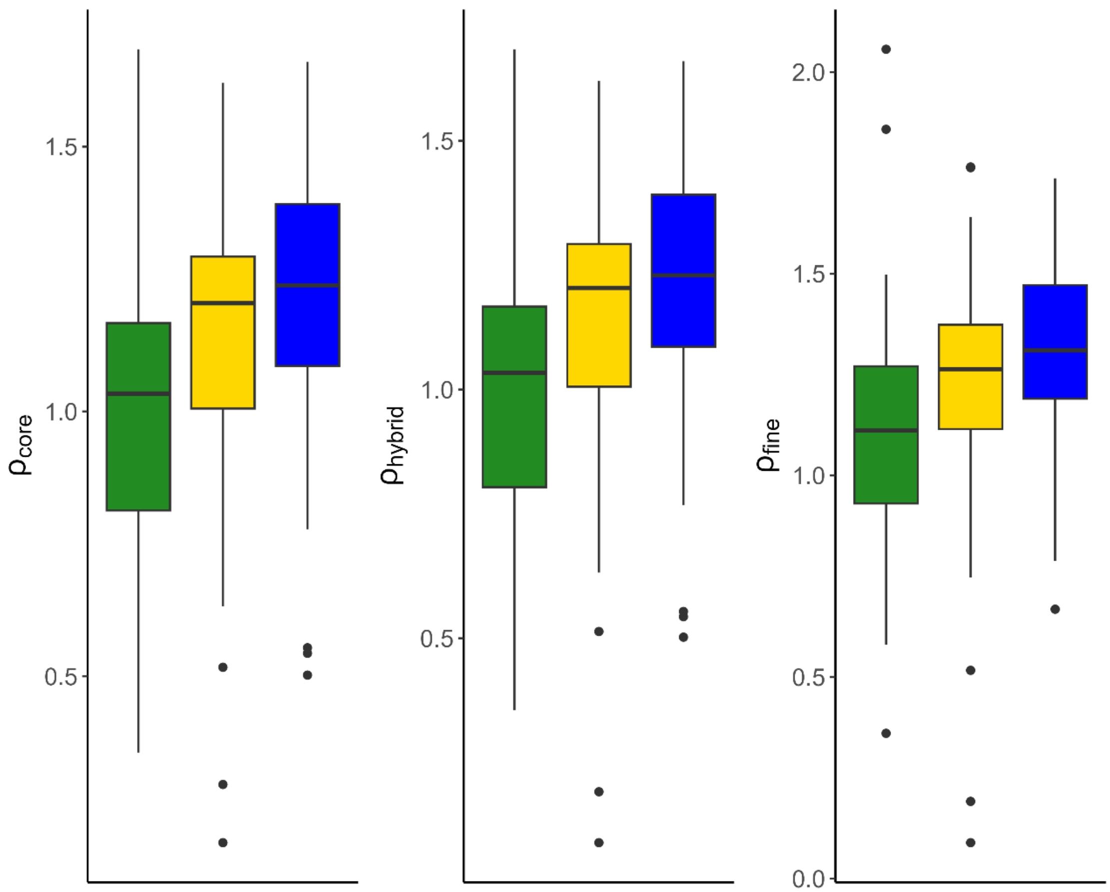
Table A2.3: Sites with historical industrial land-use.

Arthur-Therrien	Baldwin	Félix-Leclerc	Frédéric-Back	Jardin botanique
2-HIGH02	3-HIGH01	5-HIGH01	6-HIGH01	7-HIGH02
2-HIGH04	3-HIGH03	5-HIGH04	6-HIGH03	7-HIGH04
2-HIGH06	3-HIGH05	5-HIGH06	6-LOW01	7-HIGH05
2-LOW01	3-LOW01	5-LOW01	6-LOW03	7-LOW01
2-LOW02	3-LOW02	5-LOW03	6-LOW05	7-LOW03
2-LOW04	3-MED01	5-LOW05	6-MED01	7-LOW04
2-MED04	3-MED02	5-MED01	6-MED02	7-MED02
2-MED06	3-MED05	5-MED03	6-MED03	7-MED04
2-MED07		5-MED04		7-MED05

Lafond	Lalancette	Père-Marquette	Pointe-aux-Prairies	Promenade-Bellerive
8-HIGH01	9-HIGH02	11-HIGH03	12-HIGH01	13-HIGH06
8-HIGH03	9-HIGH04	11-HIGH04	12-HIGH03	13-LOW02
8-HIGH04	9-HIGH05	11-HIGH05	12-HIGH05	13-LOW03
8-LOW02	9-LOW04	11-LOW01	12-LOW02	13-LOW04
8-LOW03	9-MED01	11-LOW03	12-LOW04	13-ME02
8-LOW05	9-MED02	11-LOW4	12-LOW05	13-MED01
8-MED01	9-MED03	11-MED04	12-MED01	13-MED02
8-MED03		11-MED05	12-MED03	13-MED04
8-MED05		11-MED06	12-MED05	

Appendix III: Comparing bulk density calculations

Figure A3.1: Visual comparison of three methods of calculating bulk density. X-axis legend: historically forested sites (in green), historically agricultural sites (in yellow), and historically industrial sites (in blue).



Appendix IV: Historical documents

Figure A4.1: Photograph (circa 1900) showing the Montreal Hunting Club getting ready for the day;s hunt. Founded in 1826, the club is the oldest of its kind, in perpetual existence, in all of North America. This is an example of the recreational hunting parties with hounds, called la *chasse à courre*, that were a prevalent pastime in the forests of Montreal in the 18th – 19th century. Retrieved from la Bibliothèque et Archives nationales du Québec.

Figure A4.2: Two photographs of typical farmers practicing their trade in the early 20th century in Montreal, location unknown. Retrieved from les Archives de Montréal.

Figure A4.3: La Presse photographs of Bomar Dump in Laval circa 1968. Though not found on the Island of Montreal, this former landfill-turned-quarry demonstrates the landfilling practice of law Q-2 article 41, where operators would pile the incoming waste of the day into neat, but compacted, mounds of waste. This suggests that the articles of law Q-2 were based on the common practices of the time as this photograph precedes the enactment of the law. Retrieved from les Archives de Montréal.

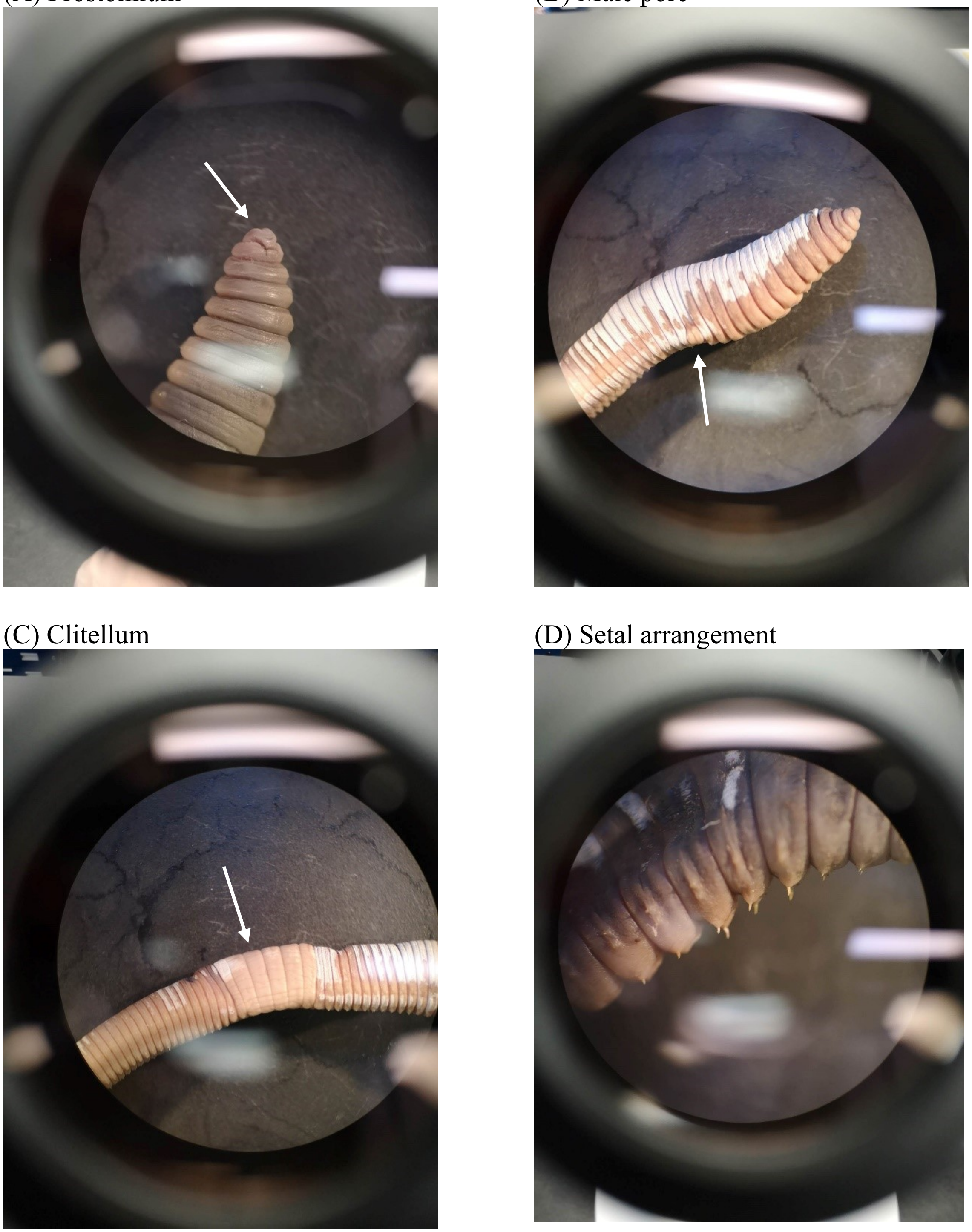
Appendix V: Background levels of heavy metals in Quebec

Figure A5.1: Concentrations of heavy metals across historical land-uses. All sites except for copper levels in previously forested and agricultural sites exceeded background concentrations. Background concentrations taken from Baillargeon-Nadeau (2016) and based upon samples from the *Basses-terres du Saint-Laurent* region in southern Quebec.

Soil variable	Historical land-use			Comparison
	Forested	Agricultural	Industrial	Baillargeon-Nadeau (2016)
Arsenic ($mgkg^{-1}$)	9.75 (± 7.52)	7.54 (± 3.86)	6.81 (± 3.02)	1.7
Chromium ($mgkg^{-1}$)	228.85 (± 154.76)	210.12 (± 64.25)	228.26 (± 64.76)	6 to 40
Copper ($mgkg^{-1}$)	40.73 (± 20.38)	40.12 (± 18.80)	53.79 (± 34.70)	10.8 to 39.6
Nickel ($mgkg^{-1}$)	52.21 (± 29.98)	45.21 (± 9.11)	48.27 (± 14.79)	7 to 33 in sandy soils 23 to 50 in clayey soils
Lead ($mgkg^{-1}$)	76.41 (± 79.64)	60.75 (± 41.84)	114.01 (± 132.77)	4.4 to 19.7
Zinc ($mgkg^{-1}$)	151.11 (± 88.42)	128.89 (± 43.40)	167.70 (± 84.66)	35.7 to 89.8

Appendix VI: Earthworm microscope photos

Figure A6.1: External anatomy used to identify earthworm individuals to the species level.



(A) Prostomium

(B) Male pore

(C) Clitellum

(D) Setal arrangement

Appendix VII: Leaf litter palatabilities

Table A7.1: unique tree and shrub species found in sampling plots and their respective palatabilities on our constructed index (in brackets).

Common Hackberry (NA)	American Basswood (4)	Gray Birch (4)
Honey Locust (NA)	Tatarian Maple (4)	Swamp White Oak (2)
Staghorn Sumac (NA)	Common Buckthorn (4)	Black Maple (4)
Ohio Buckeye (NA)	Eastern Cottonwood (4)	Siberian Elm (4)
American Hophornbeam (NA)	Bitternut Hickory (4)	Black Cherry (4)
Common Honeysuckle (NA)	Freeman Maple (4)	American Mountain Ash (4)
Frosted Hawthorn (NA)	Trembling Aspen (3)	Chokecherry (4)
Common Hawthorn (NA)	Green Ash (4)	Red Pine (1)
Viburnum spp. (NA)	Columnar Aspen (3)	Black Walnut (4)
Crabapple (NA)	White Spruce (1)	Elm (4)
Japanese Lilac (NA)	Austrian Pine (1)	Red Maple (3)
Bur Oak (2)	White Walnut (4)	Tamarack (1)
Cherry (4)	Sugar Maple (3)	Columnar Oak (1)
Littleleaf Linden (4)	White Elm (3)	Turkish Hazelnut (3)
Norway Spruce (1)	Norway Maple (4)	Tapao Shan Spruce (1)
Blue Spruce (1)	Manitoba Maple (4)	
Silver Maple (3)	Red Oak (1)	
White Ash (4)	White Birch (4)	