RECLAIMING LANDFILL TO WOODLAND: ECOSYSTEM SERVICE PROVISION USING TREES EARTHWORMS AND COMPOSTED GREEN WASTE

By

Martin Darcy Lampert

A thesis submitted in partial fulfilment of the requirements for the degree of Doctor of Philosophy at the University of Central Lancashire

January 2023

RESEARCH STUDENT DECLARATION FORM

Type of Award : Doctor of Philosophy

School: Natural Sciences

1. **Concurrent registration for two or more academic awards**

   I declare that while registered as a candidate for the research degree, I have not been a registered candidate or enrolled student for another award of the University or other academic or professional institution

2. **Material submitted for another award**

   I declare that no material contained in the thesis has been used in any other submission for an academic award and is solely my own work

3. **Collaboration**

   I declare this thesis was not part of a collaborative project

4. **Use of a Proof-reader**

   No proof-reading service was used in the compilation of this thesis.

Signature of Candidate ____

Print name: MARTIN LAMPERT

ABSTRACT

Composted green waste (CGW) is a general-purpose organic soil amendment made from plant and food waste. Over the last 25-years, field trials have been conducted on reclaimed land throughout the UK, to understand CGW influence on tree growth and soil development during restoration. These trials found CGW increases tree growth and soil organism activity by supplying organic matter, nutrients, and provides other physical, chemical, and biological benefits. However, this knowledge has been based exclusively on short-term data obtained during the first 2 to 4-years of woodland establishment. To resolve this, the present study revisited historic field experiments to examine CGW impacts on the long-term recovery of woodland ecosystems. Tree growth, soil formation, nutrient cycling, and carbon storage were assessed to evaluate CGW impact on supporting and regulating ecosystem services. Similarly, earthworm populations were also recorded as their activities mediate these ecosystem services. Data revealed one-off application of compost has short acting and longer acting 'legacy effects', that remained detectable at sites reclaimed 5, 10, and 20 years ago. When sufficient quantities of CGW are used the 'compost-effect' is clear and consistent, lowering soil C:N, and increasing nitrogen stocks, soil organic matter content, and soil carbon storage. During the first 5 years of site development, ground-vegetation, earthworms, and CGW drive soil formation, nutrient cycling, and carbon storge. Then, as restoration progresses the influence of trees grows coming to dominate site and soil development. Of the trees species studied, Italian Alder (*Alnus cordata*) and Silver birch (*Betula pendula*) are compatible with CGW. Their fast growth and high-quality litter compliments' the fertile soil conditions compost and earthworms create. Yet, the fertile conditions CGW creates may not be suitable for all trees and restoration contexts as evidenced by CGW's chemical properties and general profile. Indeed, organic amendments, trees, and soil organisms should be conceptualised as complementary integrated parts of a broader 'reclamation system', where components interact with one another and their surrounding ecological conditions. The elements of a reclamation system must be compatible with one another and can be adjusted or replaced depending on site conditions and project goals.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1.0 INTRODUCTION

Transformation of natural ecosystems to mining, quarrying, and landfill provides countless benefits to society, but destroys the living organic and mineral soil horizons which support terrestrial life (MEA, 2005). Soil degradation is a serious threat to society, human health, and economic stability, and undermines the normal function of terrestrial ecosystems (FAO, 2011; Jones et al., 2012; Ali et al., 2013; Gregory et al., 2015; Sutton et al., 2016).

Mining and other forms of industrial land-use remove soil and vegetation from land to access sub-surface minerals and strata, dismantling soil structure, degrading soil carbon, disrupting nutrient cycling, and collapsing soil food-webs (Bradshaw, 1997; Wong and Bradshaw, 2002). The primary objective of land restoration is to re-establish soil physical, chemical, and biological conditions and processes within socially relevant timescales (Bradshaw, 1983; Heneghan et al., 2008). Achieving this requires energy intensive soil reconstruction, with waste minerals from construction and engineering activities forming the bulk of the new soil mineral fraction. These materials are crushed, blended, and layered to produce a soil substrate suitable for plant growth. However, these manufactured mineral substrates generally lack soil organic matter, nutrients, and may have extremely acidic or alkaline pH. Consequently, organic amendments can be added to manufactured mineral substrates to alleviate these issues (Moffat, 1996a; Bending et al., 1999; Haigh, 2000; Butt, 2008).

Numerous organic amendments are now available for use in reclamation, including anaerobic digests, sewage sludge, composted green waste, spent mushroom compost, papermill sludge, and animal slurry and manures. All have varying levels of organic matter, C:N ratio, and macronutrient content (Kilbride, 2014). These materials provide resources which plants and soil organisms utilise for metabolism, generating fundamental ecological processes (Jouquet et al., 2006; Lavelle et al., 2006). These include primary production, soil formation, nutrient cycling, and carbon storage. Without these processes ecosystems cannot function, outlining the importance of re-establishing them on reclaimed land (MEA, 2005).

Earthworms ability to mediate the above processes has been widely recognised, as their feeding and burrowing activity involves mixing and combining organic and mineral substrates. This simple activity promotes nutrient cycling and plant growth and leads to formation of structural aggregates which stabilise carbon within newly formed organo-mineral composites (Blouin et al., 2013; Vidal et al., 2019). Earthworm activity can be crucial to pedogenesis on reclaimed land (Scullion and Malik, 2000; Frouz et al., 2013). Yet despite their important role, the utilization of earthworms in land reclamation to woodland is comparatively low (Butt, 1999). Earthworms also perform poorly when organic matter and other resources are limited, hence organic amendments may improve earthworms ability to promote the recovery of damaged soils by acting as a food-source for earthworms, fueling their soil forming activities.

Composted green waste (CGW) has been shown to be effective in increasing earthworm activity and tree growth (Butt et al., 2004; Ashwood et al., 2018). However, evidence remains confined to a limited number of studies which monitor woodland establishment over short periods, or fail to evaluate how soil biota, organic amendments, and trees interact and combine to influence ecological recovery over longer periods (Foot et al., 2003; Foot and Moffat, 2008; Ashwood et al., 2018). Consequently, the present study focuses on legacy impacts of organic soil amendment (defined as 5+ years) and examines earthworm contributions to multiple ecosystem services simultaneously. This will provide new information on earthworms' all-round contribution to landfill regeneration, and explore which combinations of trees, organic amendments, and soil organisms are effective when used together.

1.1 Aims and objectives

Explore CGW legacy impacts (defined as 5+ years) on provision of ecosystem services related to tree growth, soil formation, earthworm activity, and soil organic carbon fate.

1. Determine CGW long term impact on above-ground tree growth, biomass production, and foliar nutrient content at reclaimed sites;

2. Record physical, chemical, and biological attributes of CGW-amended soils to identify potential links between CGW application rate, CGW incorporation depth, soil formation, and carbon storage;

3. Determine how earthworm community composition, density, and biomass impacts pedogenesis in CGW-amended soils;

4. Outline CGW and earthworms synergistic impact on supporting and regulating ecosystem services

CHAPTER 2.0 LITERATURE REVIEW

2.1 Ecosystem services and earthworms as ecosystem engineers

2.1.1 Ecosystems and ecosystem services

The millennium ecosystem assessment (MEA) is a key text in contemporary ecology that provides environmental managers with a framework for understanding nature. In its opening introduction the MEA (2005) characterizes ecosystems as *"dynamic complexes of plant, animal, microorganism communities, and the non-living environment, interacting as a functional unit".* The (MEA) posits that *"people are integral parts of ecosystems and that a dynamic interaction exists between humans and other parts of ecosystems";* adding *"the actions people take that influence ecosystems result not just from concern about human well-being, but also from considerations of the intrinsic value of species and ecosystems".* In contrast, the MEA (2005) defines 'ecosystem services' as *"the benefits people obtain from ecosystems"*, with Fisher et al. (2009) submitting that *"without human beneficiaries they are not services".* The suggestion is that whilst ecosystems involve all living things, ecosystem services are only for people (Fisher et al., 2009; Dominati et al., 2010; Gómez-Baggethun et al., 2010; La Notte et al., 2017; Small et al., 2017). This is problematic because, if ecosystem services belong exclusively to humans, then environmental managers may value them more than the processes underlying them, resulting in failure to link services and processes together (Dominati et al., 2010; Raymond et al., 2013). Processes are defined by Dominati et al. (2010) as: '*the transformation of inputs into outputs*'. In ecosystems, processes that transform and cycle matter are just as important as the final outputs ecosystems provide, and in many cases are inseparable. Soil organic matter (SOM) typifies this. SOM is not a service, but rather an input and output for multiple ecological cycles including primary production, nutrient cycling, and carbon storage, resulting in SOM being considered a unifying concept for above-below-ground interactions (Fierer et al., 2009; Paul, 2016).

The MEA (2005) provides a framework for understanding the relationships between ecosystem services, and the processes underlying them by separating ecosystem services into four distinct categories. These are: 1) Supporting services – including primary production, soil formation, nutrient cycling, hydrological cycling; 2) Regulating services – including regulation of climatic extremes, flooding, disease, and carbon storage; 3) Provisioning services – i.e. production of material outputs such as food, fresh water, fibres, timber, and fuels; and 4) Cultural services – including recreation, education, aesthetics, and physical, psychological, and spiritual well-being.

Supporting services can be distinguished from other services as 'the processes that make ecosystems work'. Maintaining their integrity ensures ecosystems generate all other services (MEA, 2005). Multiple authors stress their unique nature, re-conceptualising supporting services as bio-physical processes (Dominati et al., 2010; Haines-Young and Potschin, 2010; Oliver et al., 2015; La Notte et al., 2017; Small et al., 2017; Jones et al., 2018; Gann et al., 2019). Bio-physical processes are: 'The architecture of ecosystems, which result from interactions between the abiotic, physical environment, and biotic communities particularly vegetation' (Maes et al., 2016). This echoes the MEA's definition of a multi-participant ecosystem, where living organisms drive processes, create structures, and generate outputs that bind ecosystems together. Consequently, any actions which maintain the "life supporting functions of nature" ultimately sustain "functions for people", making the former a pre-requisite for the latter in environmental management (Noël and O'Connor, 1998). This means ecosystem services 'for people' are not the goal of environmental management. As **FIGURE 2.1** outlines, human derived benefits are simply natural byproducts of organised resilient ecosystems (La Notte et al., 2017; Small et al., 2017). Hence, rather than aiming to provide a specific ecosystem service, the challenge is to develop management approaches and techniques that maintain the bio-physical processes which sustain ecosystem functionality (Dominati et al., 2010; Blouin et al., 2013).

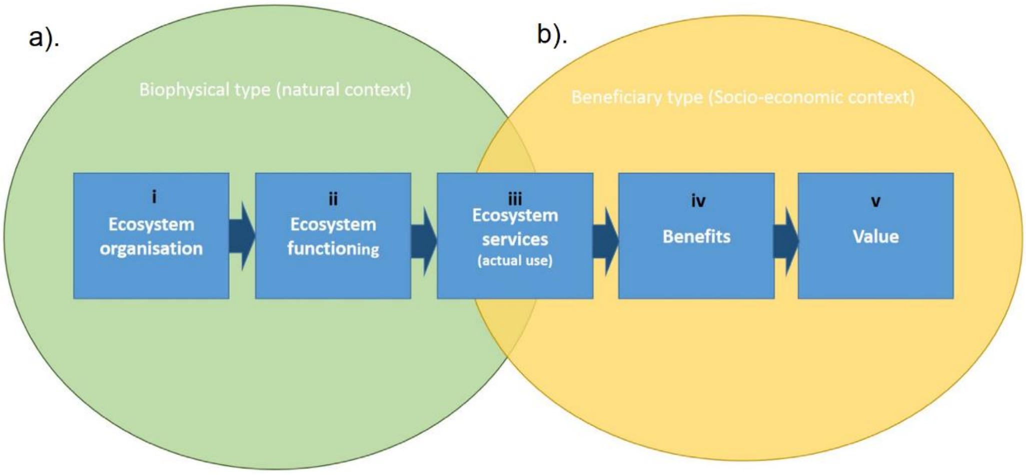


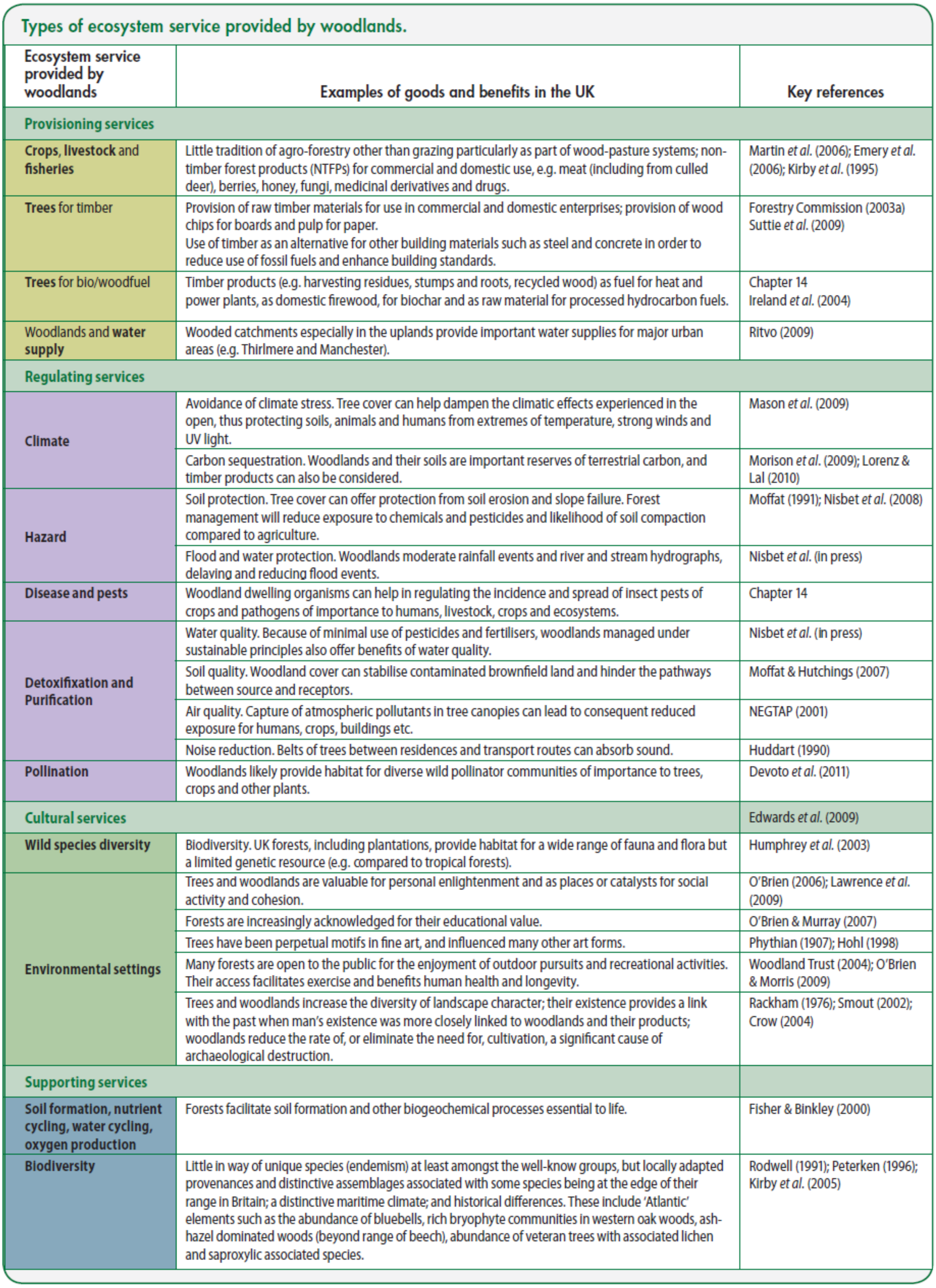
FIGURE 2.1 – *Traditional ecosystem services appraisals generally begin at (**box iii**), the 'ecosystem services' stage, without fully considering the processes which bring these products to bear (**box i** and **box ii**). By emphasizing: **(box i). ORGANISATION** – How ecosystems are organized and work; and **(box ii). FUNCTION** – What the ecosystem does or could do i.e. its ecological integrity and ability to produce; then **(iii). SERVICES** – i.e. how the ecosystem is used or could be used; can be delivered more effectively. Restoration activities that support recovery of underlying processes are a primary concern for ecological restoration practitioners (Small et al., 2017).*

2.1.2 Woodland ecosystem services

The UK Forestry Standard defines woodland as *'land under stands of trees with a canopy cover of at least 20%'* (Quine et al., 2011; Forestry-Commission, 2017). Where tree cover is lacking; shrubs, grasses, herbaceous plants, waterbodies, footpaths, and other landscape elements contribute towards ecosystem services derived from woodlands. Yet trees give woodlands their overarching structure. Tree establishment and growth can also improve the functionality of degraded land, and indicate success in land reclamation to woodland end-use (Bradshaw, 1983; Doick et al., 2009).

Quine et al. (2011) synthesised the ecosystem services provided by UK woodlands, finding woodlands underpin multiple supporting, regulating, and provisioning services **(TABLE 2.1)**. These include soil formation, water cycling, nutrient cycling, organic matter cycling, climate regulation, carbon storage, and production of timber and food (MEA, 2005). To understand how woodland expansion impacts these services, Burton et al. (2018) reviewed 160 articles but found a research bias towards production forestry and carbon storage. The impact of woodland expansion on multiple ecosystem services (MES) had never been documented. Considering this, Burton et al. (2018) recommended future research should focus on provisioning and cultural services from new and naturally regenerating community woodland sites.

However, supporting services were overlooked by Burton et al. (2018), revealing the tendency to perceive ecosystems solely for their value to human beneficiaries. This anthropocentric impulse can limit our understanding of how ecosystems work. Conceptual frameworks like the ecosystem services paradigm are helpful for organising and structuring our thinking but cannot organise ecosystems themselves. Only biophysical processes mediated by living organisms including trees can organise ecosystems. Hence, to deliver the services stakeholders desire, restoration practitioners must create conditions which allow trees and other woodland organisms to thrive.



Types of ecosystem service provided by woodlands.

Ecosystem service provided by woodlands	Examples of goods and benefits in the UK	Key references
Provisioning services		
Crops, livestock and **fisheries**	Little tradition of agro-forestry other than grazing particularly as part of wood-pasture systems; non-timber forest products (NTFPs) for commercial and domestic use, e.g. meat (including from culled deer), berries, honey, fungi, medicinal derivatives and drugs.	Martin *et al.* (2006); Emery *et al.* (2006); Kirby *et al.* (1995)
Trees for timber	Provision of raw timber materials for use in commercial and domestic enterprises; provision of wood chips for boards and pulp for paper. Use of timber as an alternative for other building materials such as steel and concrete in order to reduce use of fossil fuels and enhance building standards.	Forestry Commission (2003a) Suttie *et al.* (2009)
Trees for bio/woodfuel	Timber products (e.g. harvesting residues, stumps and roots, recycled wood) as fuel for heat and power plants, as domestic firewood, for biochar and as raw material for processed hydrocarbon fuels.	Chapter 14 Ireland *et al.* (2004)
Woodlands and **water supply**	Wooded catchments especially in the uplands provide important water supplies for major urban areas (e.g. Thirlmere and Manchester).	Ritvo (2009)
Regulating services		
Climate	Avoidance of climate stress. Tree cover can help dampen the climatic effects experienced in the open, thus protecting soils, animals and humans from extremes of temperature, strong winds and UV light.	Mason *et al.* (2009)
	Carbon sequestration. Woodlands and their soils are important reserves of terrestrial carbon, and timber products can also be considered.	Morison *et al.* (2009); Lorenz & Lal (2010)
Hazard	Soil protection. Tree cover can offer protection from soil erosion and slope failure. Forest management will reduce exposure to chemicals and pesticides and likelihood of soil compaction compared to agriculture.	Moffat (1991); Nisbet *et al.* (2008)
	Flood and water protection. Woodlands moderate rainfall events and river and stream hydrographs, delaying and reducing flood events.	Nisbet *et al.* (in press)
Disease and pests	Woodland dwelling organisms can help in regulating the incidence and spread of insect pests of crops and pathogens of importance to humans, livestock, crops and ecosystems.	Chapter 14
Detoxification and Purification	Water quality. Because of minimal use of pesticides and fertilisers, woodlands managed under sustainable principles also offer benefits of water quality.	Nisbet *et al.* (in press)
	Soil quality. Woodland cover can stabilise contaminated brownfield land and hinder the pathways between source and receptors.	Moffat & Hutchings (2007)
	Air quality. Capture of atmospheric pollutants in tree canopies can lead to consequent reduced exposure for humans, crops, buildings etc.	NEGTAP (2001)
	Noise reduction. Belts of trees between residences and transport routes can absorb sound.	Huddart (1990)
Pollination	Woodlands likely provide habitat for diverse wild pollinator communities of importance to trees, crops and other plants.	Devoto *et al.* (2011)
Cultural services		Edwards *et al.* (2009)
Wild species diversity	Biodiversity. UK forests, including plantations, provide habitat for a wide range of fauna and flora but a limited genetic resource (e.g. compared to tropical forests).	Humphrey *et al.* (2003)
Environmental settings	Trees and woodlands are valuable for personal enlightenment and as places or catalysts for social activity and cohesion.	O'Brien (2006); Lawrence *et al.* (2009)
	Forests are increasingly acknowledged for their educational value.	O'Brien & Murray (2007)
	Trees have been perpetual motifs in fine art, and influenced many other art forms.	Phythian (1907); Hohl (1998)
	Many forests are open to the public for the enjoyment of outdoor pursuits and recreational activities. Their access facilitates exercise and benefits human health and longevity.	Woodland Trust (2004); O'Brien & Morris (2009)
	Trees and woodlands increase the diversity of landscape character; their existence provides a link with the past when man's existence was more closely linked to woodlands and their products; woodlands reduce the rate of, or eliminate the need for, cultivation, a significant cause of archaeological destruction.	Rackham (1976); Smout (2002); Crow (2004)
Supporting services		
Soil formation, nutrient cycling, water cycling, oxygen production	Forests facilitate soil formation and other biogeochemical processes essential to life.	Fisher & Binkley (2000)
Biodiversity	Little in way of unique species (endemism) at least amongst the well-know groups, but locally adapted provenances and distinctive assemblages associated with some species being at the edge of their range in Britain; a distinctive maritime climate; and historical differences. These include 'Atlantic' elements such as the abundance of bluebells, rich bryophyte communities in western oak woods, ash-hazel dominated woods (beyond range of beech), abundance of veteran trees with associated lichen and saproxylic associated species.	Rodwell (1991); Peterken (1996); Kirby *et al.* (2005)

***TABLE 2.1 –** Ecosystem services provided by UK woodlands (Quine et al., 2011).*

2.1.3 Soil-based ecosystem services

UK soils make invaluable contributions towards multiple ecosystem services (MES), with Haygarth and Ritz (2009) attributing eighteen services to soil-based activities. However, collectively the literature attributes six supporting services to soils, namely: primary production, soil formation, water cycling, nutrient cycling, organic matter cycling, and biodiversity (Kibblewhite et al., 2008; Dominati et al., 2010; Quine et al., 2011; Jónsson and Davíðsdóttir, 2016). These overlap with the supporting services attributed to tree growth, demonstrating plant and soil organisms' combined role in generating 'bundles' or 'multiple' ecosystem services simultaneously. Plants and soil organisms generate MES as a general byproduct of their activity and growth, demonstrating how soil processes and outputs are intimately linked **(FIGURE 2.2)** (Kibblewhite et al., 2008; de Vries et al., 2013; Bünemann et al., 2018)**.**

Within parameters set by climate and underlying geology, plant and soil organism interactions are the most dynamic factor shaping soil pedogenesis, exerting impacts observable at scales from microscopic to continental**.** Plants and soil organisms connect above and belowground ecosystems together, transforming the non-living substrate into earths biological engine (Haygarth and Ritz, 2009). Roots, soil-fauna, and various microbes generate soils familiar bio-physical structures, in the shape of aggregate formations, interconnected passageways and pores, and mycorrhizal networks. Lavelle et al. (2006) suggest that soil processes are integrated within these structures, whilst Brussaard et al. (2007) and others assert biogenic structures (i.e. structures generated by living organisms) unify *all* soil concepts (Wardle et al., 2004; Brussaard et al., 2007; Zanella et al., 2011). Indeed, by facilitating movement of air, water, organic matter, and nutrients, biogenic structures generate a multitude of ecosystem services (Lavelle et al., 2006; Brussaard et al., 2007; Zanella et al., 2011).

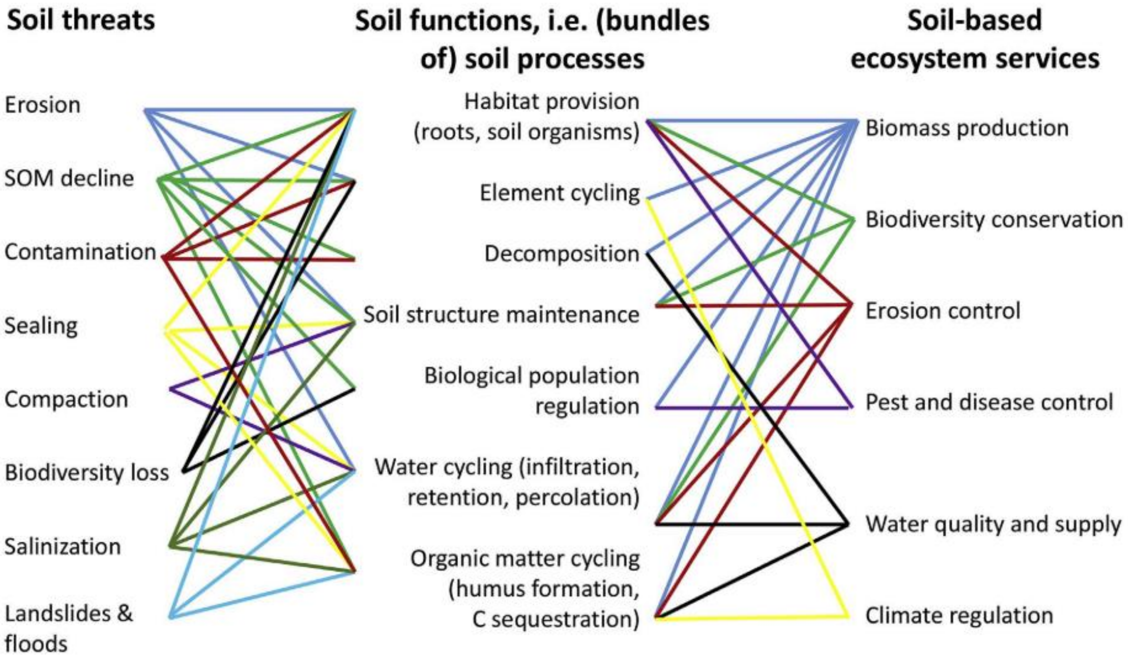


__FIGURE 2.2__ – The link between soil-based functions (Centre) and ecosystem services (Right) are negatively impacted by soil threats (Left). This demonstrates how soil functions, services, and threats are inter-linked, with the latter interrupting the flow of services from the soil system (Bünemann et al., 2018).

Despite their soil defining nature, bio-genic structures and the organisms that create them are rarely considered in ecological restoration interventions (Bradshaw and Hüttl, 2001; Farrell et al., 2020). Awareness and applied knowledge of techniques which promote soil biological development are lacking, whilst inadequate monitoring means natural regenerative processes remain poorly understood (Heneghan et al., 2008; Farrell et al., 2020). Studies in grasslands and degraded arable soils suggest complex food-webs are needed to reinstate soil function (Wagg et al., 2014; Morriën et al., 2017). Yet this takes time, and the poor quality of most reclaimed soils inhibits natural recolonization by trees and soil organisms (Bradshaw, 2000; Stanturf et al., 2014; Farrell et al., 2020). Removal of barriers to natural colonization, and direct introduction of tree species and soil organisms capable of exerting immediate, significant, beneficial impacts on soil processes, are therefore useful in land reclamation to woodland end-use.

2.1.4 Earthworms as ecosystem engineers

Ecosystem engineers have received increasing attention as functional groups of organisms that have a disproportionate impact on ecosystem structure and function (Jones et al., 1994; Jouquet et al., 2006; Brussaard et al., 2007; Blouin et al., 2013). Ecosystem engineers impact their environment by modifying habitat structure and availability of resources in ways the majority of organisms do not (Pulleman et al., 2012).

Earthworms represent the largest animal biomass in many soil ecosystems (Lavelle and Spain, 2001), and in temperate soils ingest up to 15% of organic matter inputs (Blouin et al., 2013). The energy they obtain is expended modifying soil, with earthworm burrowing and bioturbation creating soils macro-structure (Lavelle et al., 2006). Earthworms' intestine like bodies digest and mix different soil elements together, providing a physiological soil formation mechanism (Brown et al., 2000; Briones, 2018). Earthworms' gizzard, gut, and intestines synchronistically grind and massage mineral particles, plant material, and microbial biomass into a homogenous, cement like, nutrient rich, micro-organism infused paste (Brown et al., 2000; Curry and Schmidt, 2007; Liu et al., 2019; Van Groenigen et al., 2019). By passing these materials through their digestive tract, networks of mucus lined burrows are created, alongside nutrient dense aggregates, and new layers of surface soil. Earthworm excretions also stimulate microbial activity, but as these excretions 'set' they harden and stabilize, regulating organic matter decomposition (Brown et al., 2000; Blouin et al., 2013; Briones, 2018). This egestion and re-organization of soil by earthworms is known as bioturbation, and has shaped landscapes on earth for millions of years (Darwin, 1881; Meysman et al., 2006). The simple action of bioturbation supports multiple processes, including soil formation, organic matter cycling, water cycling, and nutrient cycling, earning earthworms the classification 'ecosystem engineer' (Jones et al., 1994; Lavelle et al., 2006; Brussaard et al., 2007; Pulleman et al., 2012; Blouin et al., 2013).

2.1.5 Earthworm ecology

Earthworms are soil invertebrates belonging to the phylum Annelida, (class Clitellata, subclass Oligochaeta). They are found in most habitats worldwide, except arid, frozen and other extreme environments (Briones, 2018). The environmental parameters which influence earthworm populations include temperature, soil moisture, plant species, extent of vegetative cover, soil texture and mineralogy, pH, and organic matter quantity and composition (Birkhofer et al., 2012; Rajapaksha et al., 2013; De Wandeler et al., 2016; Schelfhout et al., 2017; Briones, 2018; De Wandeler et al., 2018; Phillips et al., 2019). In temperate systems earthworm abundance and diversity is greatest in mid-latitude countries (e.g. Germany, Poland, England) where plant cover is extensive and moisture availability high (Rutgers et al., 2016; Phillips et al., 2019). Within temperate climates, temperate grasslands have greater earthworm abundance and diversity than woodlands due to higher root density and greater availability of organic matter (Fragoso and Lavelle, 1992; Edwards and Bohlen, 1996; Rutgers et al., 2016; Phillips et al., 2019). However, earthworms remain important in many temperate woodland systems, being most strongly associated with broadleaf tree species that produce low C:N (<30:1), high calcium litter (>15 mg-g^{-1}) (Reich et al., 2005; De Wandeler et al., 2016; Schelfhout et al., 2017) **(FIGURE 2.3)**.

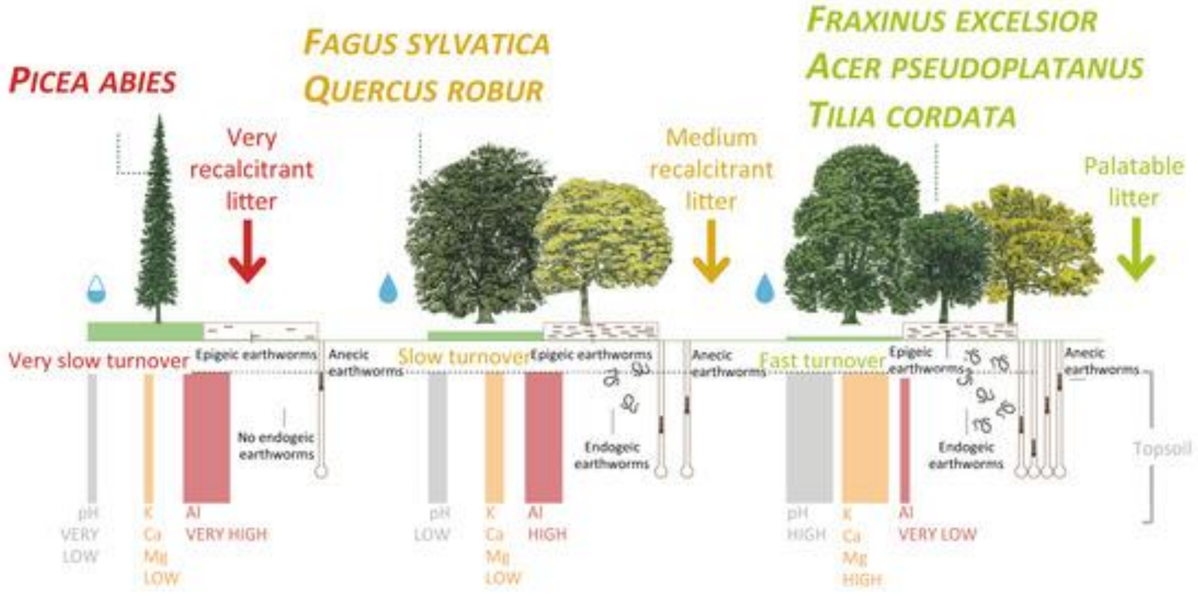


FIGURE 2.3 – *Earthworm community density, diversity, and composition shifts depending on the tree species present. Highly palatable leaf litter, i.e. litter that is relatively high in calcium and nitrogen, and low in lignin, promotes the greatest abundance and diversity of earthworms. The further litter and soil properties deviate from this ideal, earthworm community structure shifts and abundance declines until only residual populations exist (Schelfhout et al., 2017)*

Earthworms are commonly divided into three broad functional groups based on contrasting behaviour and morphology. The first group – epigeic earthworms; feed on surface litter, inhabiting and burrowing the upper most O-horizons. The second group – endogeic earthworms are geophages; and occupy deeper organo-mineral A-horizons where they consume organo-mineral soil. The third group – anecic earthworms, inhabit O, A, and B horizons by constructing long vertical burrows which connect surface and subterranean soils together. They drag surface litter into these burrows to feed on plant matter and SOM (Bouché, 1972; Bouché, 1977; Lavelle, 1988; Doube et al., 1997). Despite these feeding preferences, earthworms are opportunistic and omnivorous. Their diets vary across space and time depending on local availability of food, which includes biomass and detritus of animals, plants, microbes, and other soil fauna (Curry and Schmidt, 2007; Curry et al., 2008; Montecchio et al., 2015).

Various thresholds influence the structure of earthworm communities, impacting soil biophysical development. In European forests, minimum thresholds for earthworm occurrence include: annual mean temperature ranges of 6 – 9 °C, forest floor pH of 5-6, soil C:N ratios <20:1, and plant biomass with C:N and C:P ratios <15:1 and <300:1 respectively (De Wandeler et al., 2016). Where these thresholds are met, earthworm community biomass begins to increase, whilst forest floor litter mass decreases (De Wandeler et al., 2016; Huang et al., 2020). For example, where C:N ratios are below 12:1, anecic and endogeic earthworms accelerate forest floor litter mass loss. This is because anecics incorporate surface litter into the A-horizon, increasing organic matter availability for endogeics, resulting in an organic matter rich mull soil (Frouz et al., 2013; Zanella et al., 2009; De Wandeler et al., 2016). Epigeic earthworms depend on surface litter for habitat and food, however with the O-horizon absent their numbers decline. Conversely, where litter C:N ratios rise above 12:1 decomposition begins to slow, increasing epigeic habitat and abundance resulting in a moder-like topsoil (Zanella et al., 2009; Eijsackers, 2011; Ferlian et al., 2014; De Wandeler et al., 2016; Schelfhout et al., 2017) **(FIGURE 2.4)**. Ultimately, differing feeding behaviours, plus inter-specific dynamics among different functional groups and their environment, provide a biophysical mechanism for altering habitat structure and generating multiple ecosystem services (Liu et al., 2019; Huang et al., 2020).

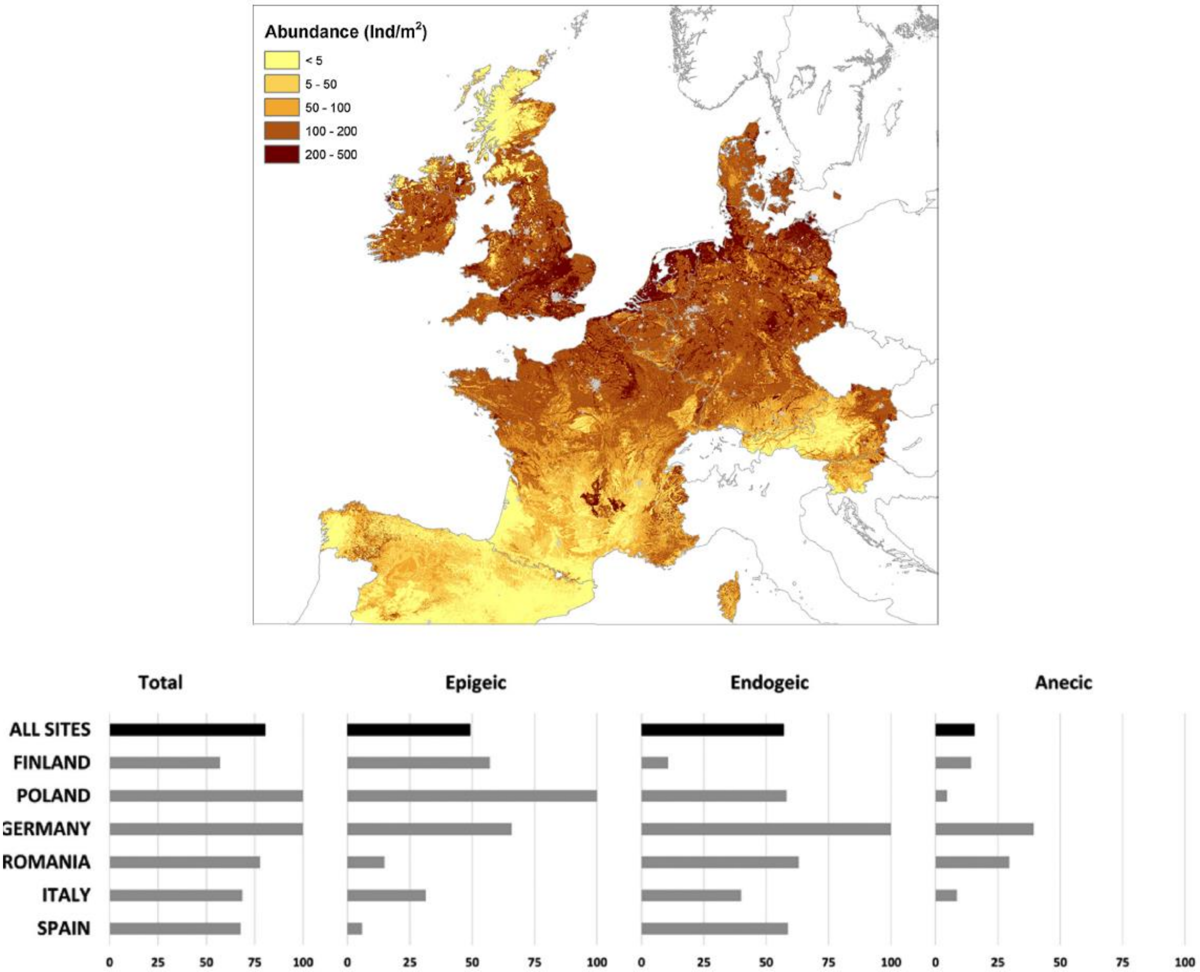


FIGURE 2.4 – Earthworm abundance in different European countries. Map represents earthworm abundance predicted by modelling (Rutgers et al., 2016). Bar-chart represents earthworm abundance based on field data collected from six European countries (De Wandeler et al., 2016). Both datasets show increased abundance in mid-latitude zones such as England, Poland, Germany, and Northern France where plant cover and soil moisture are higher.

2.2 Land reclamation and restoration

2.2.1 Background to restoration ecology

Land degradation is a pervasive and systematic phenomenon, occurring wherever ecosystems are used for human oriented production. Land degradation is defined by Scholes et al. (2018) as: *'the many human-caused processes that drive the decline or loss in biodiversity, ecosystem functions or ecosystem services, in any terrestrial and associated aquatic ecosystems'.* Agriculture, urban and industrial development, mineral extraction, and waste disposal all benefit society. However, as currently practiced these activities are driving the decline and collapse of ecosystem function. By removing plants and organisms from ecosystems overall structural, biological, and material complexity becomes diminished, degrading the productivity and functionality of natural systems. Indeed, at the present rate of land degradation, estimates suggest 90% of Earths land surface could become degraded by 2050 (Scholes et al., 2018).

Ecological restoration is: *'the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed'.* It is one of several methods of recovery on the restoration continuum (Gann et al., 2019) **(FIGURE 2.5).** *Restoration* aims for complete recovery of functions and services to pre-defined native conditions but is rarely achieved in practice (Gann et al., 2019). A historically referenced past habitat usually acts as a 'target' for restoration, providing species assemblages, biodiversity levels, physical conditions, and overall functionality (McDonald et al., 2016; Gann et al., 2019). Yet many studies suggest complete recovery is unlikely especially where degradation is severe (Benayas et al., 2009; Jones et al., 2018). Severe degradation is defined by Handley et al. (1998) as: *'land so damaged by industrial or other development that it is incapable of beneficial use without treatment'.* Mineral extraction and landfill cause severe degradation, removing soils and vegetation from land rendering natural processes incapable of repairing said degradation within decadal timescales (Bradshaw, 2000; Scholes et al., 2018). In such cases, *rehabilitation* becomes a pragmatic alternative for promoting long-term recovery, *providing a* 'first-step' through re-instatement of baseline functions without referring to pre-defined historically referenced habitat conditions (Alexander et al., 2016; Gann et al., 2019) **(FIGURE 2.5)**.

In the UK, rehabilitation is known as *reclamation,* and is synonymous with returning severely degraded land to beneficial use (Bradshaw and Chadwick, 1980; Harris et al., 1996). According to Bradshaw (1997) reclamation implies creation of new states, where structure and function are different from what went previously, allowing use of novel organisms and soil materials to aid recovery. Consequently, supporting services provide a natural starting point for re-establishing old or creating new ecological baselines, with multiple authors suggesting primary production, soil structural formation, water cycling, nutrient cycling, and soil biological activity best characterize ecosystem structure and function (Groot et al., 2013; Alexander et al., 2016; Gann et al., 2019).

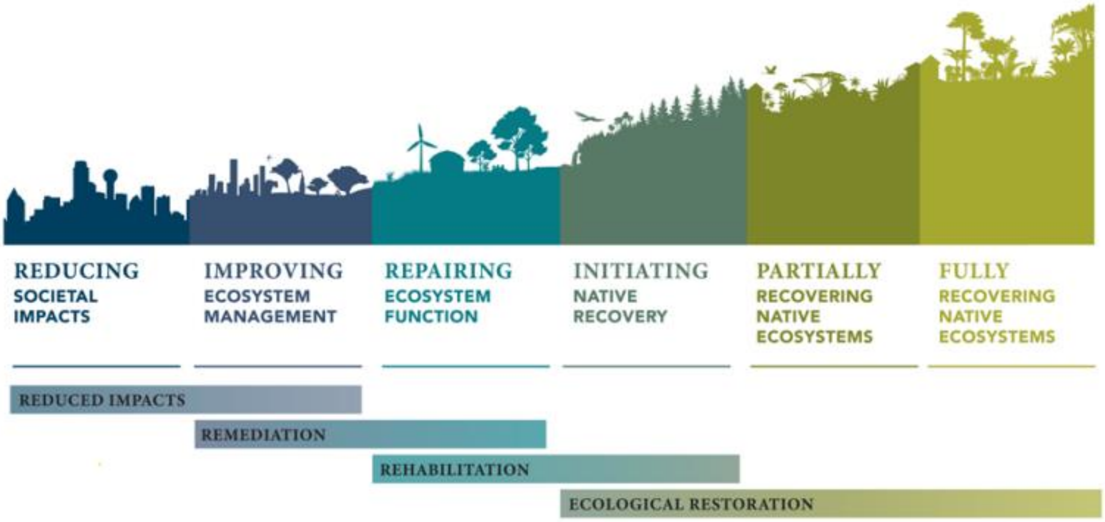


FIGURE 2.5 – 'THE RESTORATION CONTINUUM' *emphasizes continuous improvement. Rehabilitation is an initial step towards long-term recovery of native reference conditions. Rehabilitation is frequently applied to post-industrial land and is also known as reclamation. On post-industrial land, soil and vegetation are often destroyed and completely absent. Consequently soil recovery using recycled soil forming materials, soil organisms, and revegetation with native and non-native plants becomes a key focus in reclamation (Gann et al., 2019).*

2.2.3 The UK reclamation experience

In the UK, mining and landfill have been two of the most pervasive forms of land degradation. Post war advancements in explosives and heavy machinery gave extractive industries the ability to exploit vast tracts of land for minerals (Simpson, 1998; Bradshaw, 2000; HM Dept for Business, 2016). This created a legacy of scarred, de-vegetated landscapes throughout Britain. Voids left from mineral extraction were then repurposed to receive landfill waste (Bradshaw, 1998; Brand et al., 2018).

A post-war boom in mineral extraction was followed by bust, and by the 1970s the British government began taking restoration of post-industrial landscapes seriously. The aim was to stabilize land for safety purposes and provide a general green covering over degraded land (Bradshaw, 1998). The expression "*if it was green restoration it was good*" summarizes early reclamation philosophy (Tomlinson, 1984). Throughout the 1980s and 1990s reclamation philosophy expanded to include ecological and social conditions, which became known as 'the ecological approach'. This involved more complex planting arrangements, and community involvement in scheme design. In the 21st century this is known as 'ecological restoration', with creation of resilient social-ecological systems being the express primary aim (Ward, 1996; Bradshaw, 1998; Handley et al., 1998; Simpson, 1998; Bradshaw, 2000; McDonald et al., 2016; Gann et al., 2019).

Despite philosophical advancements, mining and landfill present ongoing technical challenges for restoration practitioners (Bloodworth et al., 2009; Brand et al., 2018) **(FIGURE 2.6**; **FIGURE 2.7)**. UK professionals must transform severely degraded land into a variety of grassland, heathland, or open mosaic habitats capable of supporting biodiversity and socio-ecological needs. Yet the methods used sometimes counteract this, relying on instant, heavy handed, physico-chemical techniques (Box, 1998; Coppin and Box, 1998; Moffat and Laing, 2003). To deliver the multi-functional ecosystems society needs, biophysical processes mediated by soil organisms must be integrated into reclamation practice (Butt, 2008; Heneghan et al., 2008; Farrell et al., 2020).

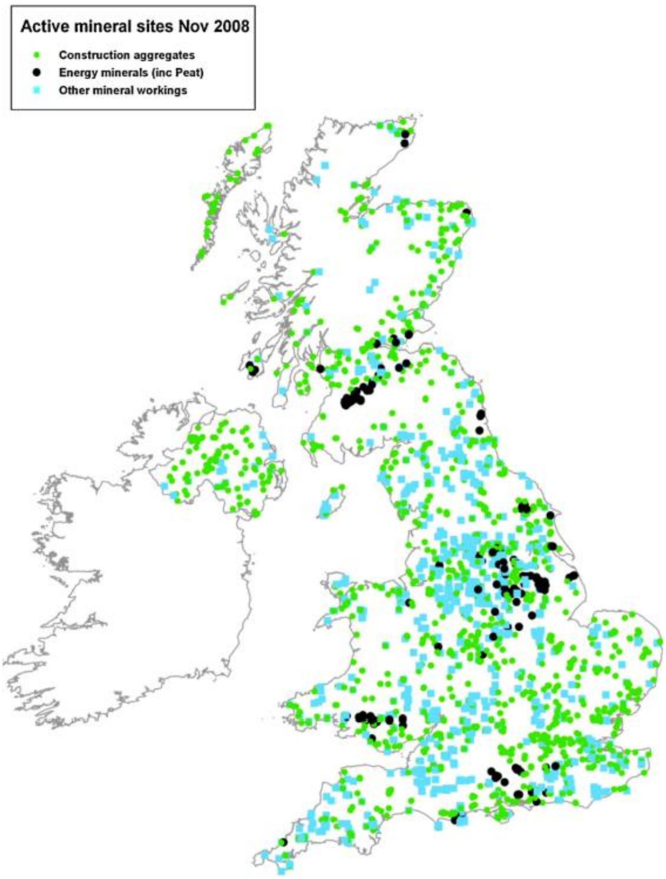


FIGURE 2.6 – Map indicating active mineral extraction sites throughout the UK. As of 2008 approximately 2,100 active mining and quarrying sites were operating in England covering a land area of 130,000 ha (Bloodworth et al., 2009).

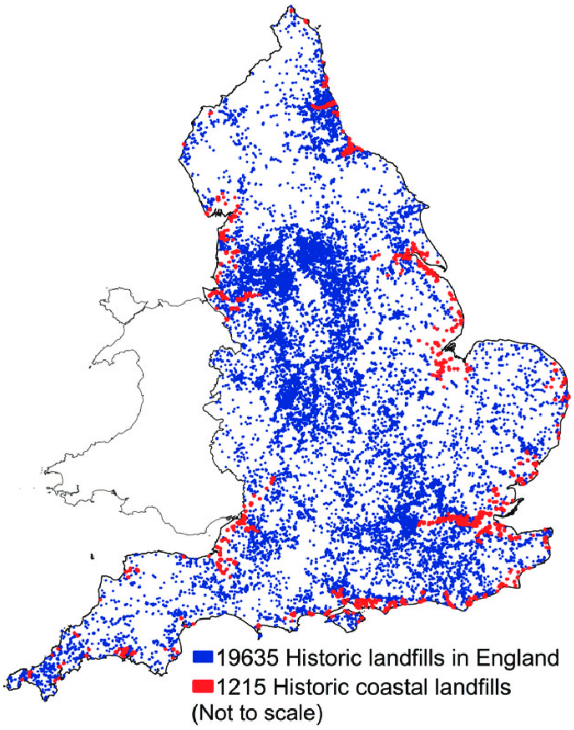


FIGURE 2.7 – *Historic landfills in England (Brand et al., 2018). A total of 19,635 closed landfills currently exist. In addition, 510 are actively receiving waste. Though not to scale, landfill distribution is geographically widespread throughout the UK, with sites focused in and around major urban areas where waste is produced. Landfills present continued risk of pollution that requires careful ongoing management and severely diminishes provision of soil-based ecosystem services.*

2.2.4 The quality of reclaimed soils

To successfully reclaim severely degraded land, the difference between natural and manufactured soils must be understood. Natural soil formation is a long-term process mediated by plant growth and soil organisms which, given sufficient time, will repair degraded land (Bradshaw, 1997; Bending et al., 1999). However, natural soil formation is driven by abiotic physical and chemical weathering processes which operate on timescales of centuries to millennia. These processes include: (1) breakdown of rocks into soil minerals; and (2) formation of distinct mineral horizons. Manufactured soils lack exposure to natural soil formation and weathering processes, hence their physico-chemical and hydrological behaviour differs (Bending et al., 1999). Conversely, biological soil formation works alongside climatic and geological processes, and interactions among plants, micro-organisms, and soil-fauna can form soil in years and decades. These comparatively rapid timescales make biological soil formation attractive for land reclamation. Biological soil formation processes include: (i) bio-genic fragmentation of rocks and minerals; (ii) nutrient transfer from subsoil to topsoil; (iii) carbon and nutrient cycling; (iv) soil organic matter production; (v) soil nitrogen accumulation; and (vi) formation of organo-mineral aggregates and structures (Bradshaw, 1997; Bending et al., 1999).

As soil depth increases, biology's influence on pedogenesis decreases, with un-weathered bedrock found at soils deepest extent. Mining activities target materials below un-weathered bedrock, bringing minerals and surplus geological deposits to the surface. Biologically active soil layers, namely vegetation, organic topsoil, and upper mineral soils, are stripped from land and buried under excavated minerals. The soil profile is essentially inverted disrupting decades, centuries, and millennia of natural soil development (Bradshaw, 1997; Bending et al., 1999). Even where biologically active soil layers are stockpiled and set aside for re-use, long-term storage can lead to compaction, organic matter degradation, nutrient leaching, and food-web collapse (Abdul-Kareem and McRae, 1984; Scullion et al., 1988; Boyer et al., 2011). When mining operations cease leftover voids are often used as landfill, resulting in industrial land-use that persists for decades and destroys soils natural form and function (Harris et al., 1996).

The destruction of naturally formed soils by mining and landfill necessitates the use of industrial bi-products and wastes as inputs materials for re-constructing new soils. These 'soil-forming materials' are generated by mining and industry and are analogous to 'parent material' (Bending et al., 1999). Soil forming materials can arise on-site or from external sources, and include colliery shales, coal spoils, sand and gravel from quarries, excavated material from civil engineering projects, crushed construction waste, and dredgings from waterbodies (Bending et al., 1999). Soil forming materials are often considered inferior to natural soils, having been exposed to far less physical, chemical, and biological pedogenesis. They possess deficiencies arising from mechanical handling and long-term storage, and their inherent chemical composition. Deficiencies include compaction, lack of structure, low organic matter, lack of nutrients, extreme pH, high salinity, and excessive toxicity (Bradshaw, 1983; Bending et al., 1999). Interestingly, many deficiencies inherent to soil forming materials can be linked to a lack of organic matter (Bending et al., 1999).

Techniques for ameliorating these deficiencies have been established, centering on improved mechanical handling, and incorporation of organic amendments (OA). Bending et al. (1999) defines amendments as: *'a material, either inorganic or organic in origin, added to soil forming materials to raise levels of fertility and improve chemical and physical properties, to aid the establishment of a vegetation cover and accelerate soil formation by creating an environment favourable to soil organisms'.* Organic amendments provide a crucial source of soil organic matter (SOM) and can be considered essential ingredients for improving the quality of manufactured soil. Organic amendments are made from organic waste derived from plants, food, timber, paper manufacturing, animal manure, and human sewage. When added to manufactured soils they serve as a SOM substitute, supporting trees and soil biota as they re-establish biological soil formation processes.

2.3 Earthworm impacts on reclaimed soil quality and ecosystem services

2.3.1 Soil formation

Soil formation is a key bio-physical process. The term describes how climate, plants, soil organisms, above-ground fauna, and time, interact with rock minerals and topography of land to create soils with markedly different functions and characteristics (Weil and Brady, 2016). Soil formation depends on the breakdown and transformation of primary mineral and organic materials into more processed and homogenised forms (Bending et al., 1999). The synchronistic feeding, burrowing, and bioturbation activity of earthworms influences this process. Earthworms transport mineral particles from below ground and organic matter from above, combining and grinding these into a nutrient rich organo-mineral paste. This paste is excreted in topsoil as casts and aggregates producing a distinctive type of soil formation (Zanella et al., 2011; Blouin et al., 2013).

Specifically, anecic and endogeic feeding behaviour transfers 90-100% of surface litter into the A-horizon, creating an aggregated organo-mineral surface layer (Zanella et al., 2009; 2011; Blouin et al., 2013; De Wandeler et al., 2016). This tendency is so pronounced that when classifying European humus forms, Zanella et al. (2009); (2011) created specific categories to describe this effect. Blouin et al. (2013) also suggests that where > 50% of the A-horizon, and > 25% of the B-horizon are composed of anecic and endogeic excretions, the term 'vermiform soil' should be applied. Indeed, soils with marginal or absent O-horizons, and organic matter enriched A-horizons are broadly known as mull soils, and are bio-indicators of anecic and endogeic earthworm activity (Paoletti, 1999; Blouin et al., 2013).

The organo-mineral mull soils that anecic and endogeic earthworms produce have been documented on reclaimed land. An example being the Sokolov coal mine in the Czech Republic, where soil formation processes were monitored using a 70-year chrono-sequence of restored mines (Frouz et al., 2001). Epigeic populations rapidly recolonized the site, then, after 20-30 years were joined by numerous endogeic species. Wherever endogeic *Aporrectodea caliginosa, Aporrectodea rosea,* and *Octolasion lacteum* were found, the humus type shifted from moder to mull, demonstrating burrowing earthworms ability to produce 'vermi-form' soils on reclaimed mine wastes (Frouz et al., 2001). However, earthworm compatible plant communities and soil conditions were required for this to take place. With Alder tree species, alkaline pH, and clay soil texture encouraging endogeic recolonization. Where tree species and soil conditions differed, endogeics remained absent and pedogenesis differed strongly, tending towards moder humus forms in earthworms absence (Frouz et al., 2001; 2006; 2013).

Similar 'vermi-form' soils were observed at a restored opencast coal mine in South Wales, although here the effects were less pronounced. At the time of sampling, only 15-years had elapsed since site restoration, hence familiar mull soil forms had less time to develop. Nevertheless, areas inoculated with soil dwelling *Aporrectodea caliginosa, Aporrectodea longa,* and *Lumbricus terrestris,* exhibited greater levels of organo-mineral mixing than an adject site restored 12-years earlier. The adjacent site was colonized passively by *Allolobophora chlorotica* and *Lumbricus rubellus* and thus lacked deeper burrowing earthworms. Organic matter and aggregate stability were also higher in inoculated areas, signaling advanced soil formation and development when burrowing earthworms were actively introduced (Scullion and Malik, 2000; Marashi and Scullion, 2003).

Evidently, where conditions are favourable for burrowing earthworms, they promote a specific type of soil formation. On reclaimed land, earthworm-generated soil formation promotes retention of SOM, carbon, nutrients, and moisture in otherwise poor-quality, under-formed soils. Data from multiple post mining sites suggest ideal conditions for anecic and endogeic earthworms include; a temperate climate, deciduous trees with high quality litter, and a fine soil texture with weakly acidic to alkaline pH (Frouz and Vindušková, 2018). It should be noted that reproduction of these exact conditions is not appropriate in all site conditions. Hence, the soil forms that anecic and endogeic earthworms produce depend on amenable tree species, soils, and climatic conditions.

2.3.2 Soil structure

Soil structure is a physical characteristic that underpins multiple ecological functions. (Bradshaw, 2000). It refers to the spatial arrangement of particles into networks of aggregates, passageways, and pores (Bottinelli et al., 2015). Soil structure is formed by the growth, movement, feeding, and interactions that take place between plants, microbes, and fauna (Wardle et al., 2004; Lavelle et al., 2006; Brussaard et al., 2007; Erktan et al., 2020). As ecosystem engineers, earthworms are important to soil structural formation creating networks of burrows, aggregates, and pores. By building these structures certain earthworm species heavily modify soil and re-orientate the system towards their precise ecological needs (Lavelle et al., 2006; Jouquet et al., 2012). Consequently, multiple authors describe soil structure as an extended phenotype of its resident microbial and macrofaunal community (Jouquet et al., 2006; Lavelle et al., 2006; Crawford et al., 2012).

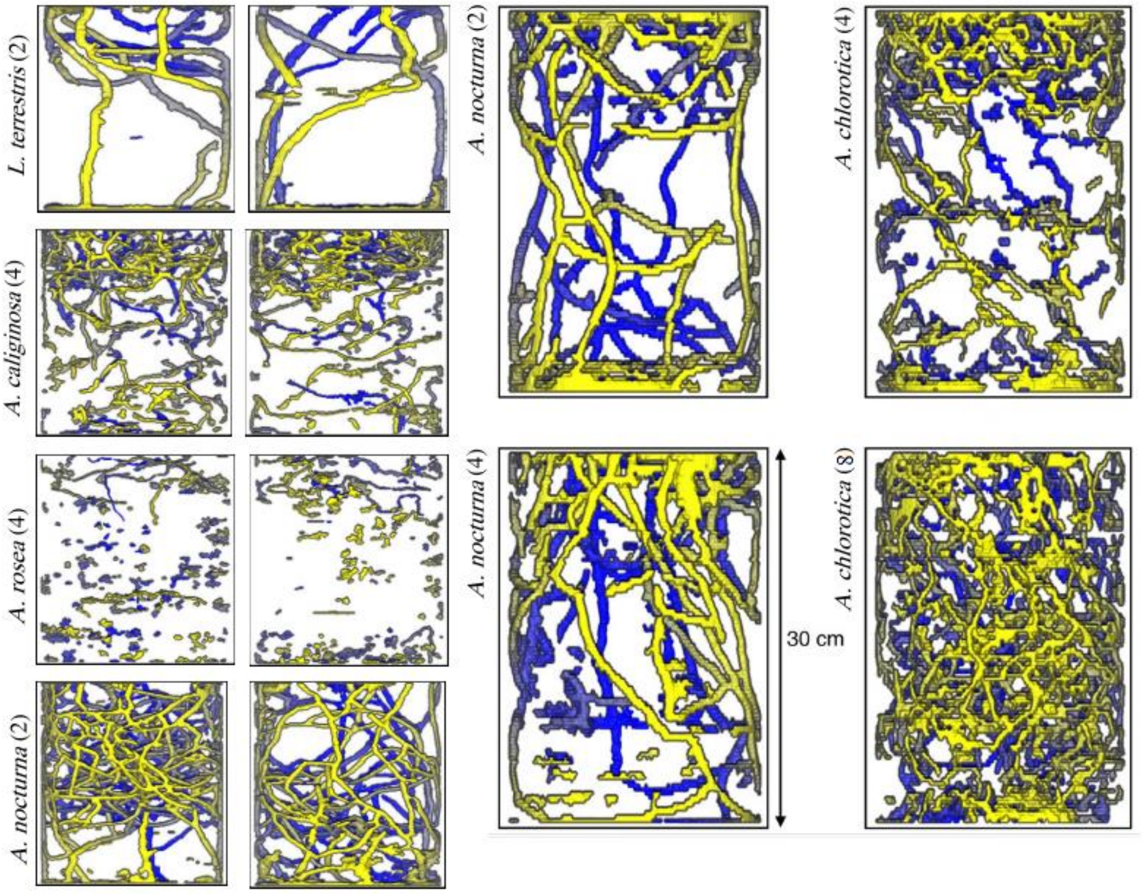


***FIGURE 2.8** – Earthworm burrow structures as revealed by X-Ray tomography scanning of soil cores inoculated with adult earthworms. Soil cores were cylindrical, 35 cm in length, and 16 cm in diameter. The species and number of earthworms in each core is indicated on the left hand side of each scan. Yellow indicates foreground, blue indicated background with transition colours in between, allowing images to be observed in 3D. Clear differences in the burrow structures of different species can be seen (Capowiez et al., 2011; Capowiez et al., 2015)*

The burrow and aggregate networks that anecic and endogeic earthworms construct help determine soils architecture at the centimeter to decimeter scale (Jouquet et al., 2006; Lavelle et al., 2006). Using X-ray tomography these networks can be visualised directly, indicating which species and ecological groups are present and their potential impacts on soil structure and function. For example, vertical burrows of anecic species promote water infiltration due to greater burrow diameter, length, and surface connectivity. Conversely, the horizontal burrows of certain endogeic species slow infiltration, due to their smaller diameter and increased backfilling by endogeic species (Capowiez et al., 2011; Capowiez et al., 2015) **(FIGURE 2.8)**.

Earthworms' biogenic structures also appear to be species-specific, with various anecic burrows differing in terms of form, function, and lifespan. For example, *L. terrestris* is described as epi-anecic, creating a single permanent burrow used for months or years for surface feeding and habitation. In contrast, *A. nocturna* and *A. longa* are endo-anecic, creating multiple temporary vertical and horizontal burrows by ingesting higher quantities of mineral soil (Bottinelli et al., 2020; Bottinelli and Capowiez, 2021). These variances show how single species can embody a gradient of epigeic, endogeic, and anecic behaviors to achieve niche differentiation (Bottinelli et al., 2020). This niche diversity facilitates development of inter-species equilibrium, allowing diverse earthworm communities to behave as functional units with downstream consequences for soil structure and function (Bottinelli et al., 2020; Lang and Russell, 2020; Bottinelli and Capowiez, 2021).

In general, diverse earthworm communities are more likely to benefit soil structure (Blanchart et al., 1999; Jouquet et al., 2006; Blouin et al., 2013). Lab and field-based studies of common UK species suggest competition for resources promotes behavioural flexibility, providing balance between different feeding behaviours, facilitating self-organisation of diverse burrow structures (Lowe and Butt, 2002; Scullion and Malik, 2000; Uvarov, 2009). Indeed, species that demonstrate dynamic feeding and burrowing behaviours may adapt better to poor quality substrates typical of disturbed environments, explaining the frequent success of *A. chlorotica,* *A. caliginosa,* and *A. longa* in manufactured soils (Piearce et al., 2003; Marashi and Scullion, 2003; Butt et al., 2004; Butt, 2008; Butt and Putwain, 2017; Butt and Briones, 2017). These species are some of the UK's most common (NE, 2014), hence their success in reclamation may simply indicate their adaptability to a broad range of soil conditions. Plant community is also a key factor in earthworm community dynamics, and the aforementioned earthworm species frequently favour open grassland habitats. Open grassland is common throughout the UK, dominating agricultural landscapes, and also reclaimed land, where grasses are used to stabilise soils and may encourage proliferation of the most regularly recorded earthworm species (Butt, 1999; Curry et al., 2008; Eisenhauer et al., 2009a; NE, 2014; Roubíčková and Frouz, 2014).

Yet, despite earthworms undoubted ability to drive soil formation, poor soil structure is considered the principal barrier to reclamation success (Moffat and McNeill, 1994; Moffat and Laing, 2003). Reclaimed soils are often compacted and lack a porous structure, hence it is incumbent on reclamation practitioners to overcome this by creating conditions that support plant growth and faunal colonization from a projects outset (Bradshaw, 1987; Heneghan et al., 2008). The main physical properties of interest during soil creation and manufacture are texture, bulk density, compaction, stone content, particle size distribution, soil strength, and soil depth (Moffat and McNeill, 1994; Bending et al., 1999) **(TABLE 2.2)**. Loose tipping is recommended for soil placement, and soil materials should also be graded and layered in order of particle size. Whilst loose tipping creates a loose uncompacted rooting medium, the layering of fine particles over coarse stones ensures structure is not too open thus providing stability for expanding plant roots and slowing water percolation. The newly formed soil profile then retains water and moisture, supporting plant growth and survival (Moffat and McNeill, 1994; Bradshaw, 1997; Bending et al., 1999). When the basic conditions set out in **TABLE 2.2** are met, earthworm communities are more likely to establish, aiding the development and maintenance of a porous soil structure.

In contrast, high bulk density values (>1.7 g cm^{-3}) limit aeration, porosity, moisture holding capacity and water infiltration, preventing plant root growth and earthworm burrowing (Söchtig and Larink, 1992; Buck et al., 2000; Jouquet et al., 2012). In dry weather, compaction creates drought conditions, affecting plant and earthworm survival. Similarly, in wet conditions compact soils become waterlogged and anaerobic, starving roots and aerobic organisms of oxygen (Moffat and McNeill, 1994). Indeed, a survey of 27 sites reclaimed to woodland following mineral extraction or landfill, found 21 sites were suffering from compaction resulting in poor tree growth, and/or complete tree establishment failure, outlining the importance of good soil structure (Moffat and Laing, 2003).

Soil Property	**Minimum Standards**
Soil Thickness	> 1.3 m over engineered / vehicular surfaces
Bulk Density	< 1.5 g cm^{3} to 0.5m depth < 1.7 g cm^{3} to 1.3m depth
Stoniness	< 40% of volume; few stones greater than 100 mm size
pH	4.0 - 8.5
Electrical Conductivity	< 2000 µS cm^{1} (in 1:1 soil water suspension)
Iron pyrite content	< 0.5%
Heavy Metal and Organic Contaminants	Environmental Protection Act 1990 Part IIA Risk based approach - Does contamination pose significant risk of harm to human health, ecological health, or property https://www.gov.uk/guidance/land-affected-by-contamination https://www.gov.uk/guidance/land-contamination-how-to-manage-the-risks

TABLE 2.2 *– Minimum standards for soil and soil-forming materials used in land restoration to forestry (Bending et al., 1999; Moffat, 2001; Hutchings et al., 2006)*

2.3.3 Organic matter and nutrient cycling

Soil organic matter (SOM) is a unique biological material made from the biomass and metabolic outputs of plants and microorganisms (Liang et al., 2017; Angst et al., 2021). It contains a blend of carbohydrates, proteins, lipids, phenol-aromatics, amino sugars, and biomolecules, making it a nutrient rich food-source for plants and soil organisms (Stockmann et al., 2013; Lorenz et al., 2021). As soil organisms produce SOM and interact with it, they generate important soil physical, chemical, and biological processes. These include soil structural formation, nutrient cycling, moisture retention, thermal regulation, chemical buffering, cation exchange, and carbon storage (Wardle et al., 2004; Brussaard et al., 2007; Jackson et al., 2017; Briones, 2018). Consequently, SOM is recognised as a key soil property and indicator of forest productivity (Grigal and Vance, 2000; Sayer, 2006).

Traditionally, SOM has been envisioned as humus, a fully decomposed and stabilized organic material which accumulates gradually via decomposition leading to increased SOM content (Berg, 2000; Lehmann and Kleber, 2015). However, contemporary views consider SOM as a continuum of organic materials, from fresh litter to metabolized 'humus', fluctuating between different states of metabolic utilization and decomposition. Rather than reaching a fully decomposed and 'stabilised' endpoint, organic matter decomposition is temporarily halted when organic materials become physico-chemically incorporated within an organo-mineral substrate (Stockmann et al., 2013; Lehmann and Kleber, 2015). For example, through their feeding and metabolic activity earthworms and microbes produce casts and necromass which becomes intimately mixed into the soil mineral fraction. At a later date, this newly formed organo-mineral substrate is then re-degraded as the same organisms that created this material seek out organic matter, outlining how organic materials flux through multiple cycles of production and decomposition (Ferlian et al., 2014; Basler et al., 2015; Vidal et al., 2017; Cui et al., 2020).

Indeed, many of SOM’s beneficial properties arise through continuous fragmentation, mineralisation, and turnover of new and previously stabilized organic material (Lehmann and Kleber, 2015). This performs two key functions: 1). Increases SOM reactivity, which 2). promotes SOM adsorption to mineral surfaces and incorporation inside organo-mineral complexes (Schmidt et al., 2011; Stockmann et al., 2013; Lehmann and Kleber, 2015). This is termed ‘stabilisation’ (Angst et al., 2021), and occurs when plant litter, root inputs, microbial cells, exoenzymes, and other biomaterials are physically bound inside mineral aggregates, or chemically attached to mineral surfaces (Lutzow et al., 2006; Schmidt et al., 2011; Stockmann et al., 2013; Lehmann and Kleber, 2015). OM stabilisation is extremely beneficial for reclaimed soils as it promotes formation of aggregated structures in a generally un-structured soil environment, enabling the manufactured soil to retain moisture, carbon, and nutrients (Scullion and Malik, 2000; Frouz et al., 2013; Józefowska et al., 2017; Frouz and Vindušková, 2018).

Earthworm feeding and bioturbation provides a key mechanism for organic matter stabilisation. The passage of litter, organic, and mineral particles through the earthworm gut fuses organic and mineral soil fractions together, stabilising and concentrating SOM and nutrients inside biogenic structures (Blouin et al., 2013; Van Groenigen et al., 2019). This effect has been demonstrated in reclaimed soils at a laboratory scale (Pey et al., 2013; Deeb et al., 2017). Indeed, earthworm bioturbation continuously drives formation, breakdown, and regeneration of organo-mineral casts and aggregates, mediating SOM turnover and cycling of nutrients (Blouin et al., 2013; Ferlian et al., 2014; Van Groenigen et al., 2019). For example, a meta-analysis of 81 articles by Van Groenigen et al. (2019) found that compared to bulk soil, earthworm casts contain significantly more organic carbon, total and plant available N and P, and have greater cation exchange capacity, base saturation, and higher pH. Thus, as earthworms mix organic and mineral fractions together, they transform plant derived SOM into more stable, nutrient enriched, bioavailable forms (Van Groenigen et al., 2019; Vidal et al., 2019).

Yet, earthworms cannot transform any and all plant material into stabilized SOM. In fact, litter properties control the initial stages of decomposition, with certain litter restricting or preventing earthworm mediated decomposition (Vidal et al., 2019). Indeed, multiple studies suggest earthworms require litter with specific characteristics. These include low C:N ratio (<30:1), low (%) lignin (<20 %), and high Ca content (>15 mg-g^{-1}) (Rajapaksha et al., 2013; Frouz et al., 2013; Schelfhout et al., 2017; Angst et al., 2021; Lorenz et al., 2021). Litter with these properties facilitates decomposition via earthworm mediated channels, influencing the composition, distribution, and turnover of SOM (Wardle et al., 2004; Vesterdal et al., 2008; Zanella et al., 2011; Vesterdal et al., 2013; Schelfhout et al., 2017; De Wandeler et al., 2018; Phillips et al., 2019; Steidinger et al., 2019; Wiesmeier et al., 2019; Angst et al., 2021; Lorenz et al., 2021).

In post-mining soils, SOM has been shown to adopt properties consistent with the plants and soil organisms which formulated it (Frouz et al., 2013; Vindušková and Frouz, 2013; Józefowska et al., 2017; Frouz and Vindušková, 2018; Lorenz et al., 2021). This means even in reclaimed soils, SOM can possess physical, chemical, and biological properties which favour the growth and survival of organisms originally responsible for its synthesis (Aponte et al., 2013). If correct, reclamation practitioners can use this knowledge to select trees, organic amendments, and soil organisms that complement one another. Knowing the ecological contexts in which earthworms succeed can provide an indication as to which tree species and organic amendments complement earthworms and why. In turn, the reclamation contexts to which earthworms are most suited could be outlined, including the functions and services an earthworm dominant system might generate.

Concerning this, Wardle et al. (2004) outlined two broad scenarios responsible for fast or slow turnover of SOM. Broadly, fast turnover systems are dominated by bacteria and earthworms, and contain fast growing plants, fertile soils, and leakier nutrient cycles. Conversely, slow turnover systems are dominated by fungi and enchytraeid worms, and contain slow growing plants, infertile soils, and tightly regulated nutrient cycles. Several additional studies suggest something akin to this may indeed be true. For example, high earthworm abundance coincides with fast turnover systems where continuous input of high quality litter (i.e. low C:N ratio, low lignin to N ratio, high Ca content), plus rapid turnover of SOM and nutrients, and bacterially dominated microbial communities exists (Hättenschwiler and Gasser, 2005; Reich et al., 2005; Reich, 2014; Józefowska et al., 2017; Schelfhout et al., 2017; Semchenko et al., 2018; Angst et al., 2021; Lorenz et al., 2021). Many grassland ecosystems and certain deciduous trees appear adapted to these fast turnover systems, however their ability to achieve this in manufactured soils remains undocumented.

2.3.4 Carbon storage

Earthworm's impact elemental cycling through their burrowing and bioturbation activity. As such, their impacts on soil carbon storage and organic matter decomposition have attracted increasing attention (Blouin et al., 2013). Several authors argue earthworms have a negative impact, reducing soil carbon storage by promoting organic matter decomposition and microbial respiration. For example, in a meta-analysis of 36 studies, Lubbers et al. (2013) found earthworms increase CO_2 emissions by 33%. This takes place when earthworms feed on surface organic matter or organic particles bound within aggregates. Zhang et al. (2013) acknowledged these negative impacts but argued earthworm bioturbation simultaneously stabilises carbon in even greater quantities, effectively spending carbon to store carbon, producing net storage overall.

In response, Lubbers et al. (2017) established an indoor mesocosm to test whether earthworm activity increased or decreased carbon stabilisation through organic matter decomposition. The study placed strict controls on carbon inputs and measured all outputs for 750-days. The study concluded earthworms do indeed protect and stabilise carbon, however unlike Zhang et al. (2013) they argued earthworms tilt the balance towards carbon losses and emissions. Yet this conclusion was inevitable as plants were intentionally excluded from mesocosms by Lubbers et al. (2017), preventing further inputs from entering soils. All soils depend on plants to fix new carbon and transfer it into the food-web. With no inputs there can only be outputs and the balance tips in favour of carbon losses and emissions (Blouin et al., 2013).

More recently, a global meta-analysis of 69 separate studies by Huang et al. (2020), found the carbon balance depends on earthworm community structure and density. When certain functional groups are present at high densities decomposition is increased. Specifically, the co-occurrence all three functional groups together, or just endogeics alongside epigeics or anecics, increases soil carbon and litter mass loss through interactive community effects. Many studies featured in this meta-analysis observed carbon dynamics for less than one year, but those that lasted longer than this demonstrated carbon losses overall. Undoubtedly, earthworms appear to alter carbon cycling and turnover, by steering plant and microbial community composition towards one that is compatible with their needs (Craven et al., 2017; Ferlian et al., 2018; Liu et al., 2019). Yet, over long-term periods this may lead to stable carbon cycling, where compatible vegetation and microbes cycle sufficient carbon inputs to sustain earthworms metabolic needs and create a carbon surplus (Liu et al., 2019). Overall, these studies give insight into the complexities of earthworm activity and their influences on the soil carbon balances. However, further research into the long-term effects of earthworm activity on soil carbon dynamics is required (Blouin et al., 2013).

2.3.5 Soil food-web

Lavelle et al. (2006) describes how soil organisms occupying different spatial scales work together as an integrated system. Micro-organisms inhabit and build soils microstructure, binding microscopic soil particles together using biofilms and hyphae. Earthworms facilitate this process, operating at the centimeter to decimeter scale, where their movement, foraging, and excretions create soils macrostructure and provide freshly available material for microbial processing and nutrient exchange with plant roots.

Earthworms appear to have specific effects on soil food-webs, steering soil ecosystems to better suit their needs. They do this by incorporating leaf litter into soil, and ingesting soil material infused with plant and microbial biomass. These actions simultaneously remodel the pore and aggregate structure of soil, and the soil microbiome, altering the cycling of vital elements (Lavelle et al., 2006; Erktan et al., 2020). Earthworm remodeling of soil structure is a major trophic interaction involving selective feeding on a range of plant materials, seeds, and microbes, and has a cascade of effects on plant communities and organisms throughout the soil food-web (Butenschoen et al., 2007; Eisenhauer et al., 2009a; Eisenhauer, 2010; Stromberger et al., 2012; Ferlian et al., 2018).

Earthworm effects on food-webs can depend on the earthworm species present, and the overall structure of the earthworm community. For example, in woodland systems anecic earthworms are more abundant under trees with calcium rich litter including *Acer pseudoplatanus, Fraxinus excelsior,* and *Tilia cordata* (Schelfhout et al., 2017). Anecic earthworms like *Lumbricus terrestris* deposit the materials they acquire around the inner walls of their constructed burrow systems, creating distinct microbial communities around their burrows (Stromberger et al., 2012). In grasslands, the same earthworm species have been shown to interact positively with arbuscular mycorrhizal fungi (AMF), whilst the endogeic species *Aporrectodea caliginosa* had no interaction with AMF at all. In a separate study another endogeic earthworm *Octolasion tyrtaeum*, was found to be strongly antagonistic towards litter-decomposing fungi, grazing on these functional groups, disrupting their hyphae, limiting litter-decomposing fungi effects on decomposition. Thus, earthworms' impact can be profound however different outcomes can occur depending on the ecological context.

Recently, earthworms have been shown to co-ordinate the activities of multiple functional soil biota groups. Earthworms simultaneously increase some groups whilst decreasing others, shifting microbial food-webs towards bacterially dominated communities (Józefowska et al., 2017; Hines, 2019; Liu et al., 2019; Medina-Sauza et al., 2019). In particular, anecic and endogeic earthworms encourage bacteria involved in nitrogen cycling, leading to increased soil nitrogen availability (Blouin et al., 2013; Medina-Sauza et al., 2019). Plants that prefer nitrogen-rich soils therefore have a competitive advantage when soil dwelling earthworms are present. In addition, because SOM tends to have a similar molecular and chemical profile to the organisms that created it, systems dominated by bacteria and nitrogen rich plants return easily digestible OM back to soil, creating positive feedback favouring earthworm metabolism (Angst et al., 2021; Lorenz et al., 2021).

These examples of earthworm interactions with plants and microorganisms throughout the food-web suggest earthworms might thrive in fast turnover soil systems theorized by Wardle et al. (2004) and Reich (2014). Earthworms are known to consume large quantities of organic matter, and increase soil respiration, and in woodland systems earthworms have been shown to prefer high quality organic inputs (Hättenschwiler and Gasser, 2005; Rajapaksha et al., 2013; Schelfhout et al., 2017). From this perspective, the idea that anecic and endogeic earthworm activity aligns with fast turnover principles makes sense. Indeed, these earthworms may promote decomposition of plant tissues resulting in production of SOM and microbial biomass with low C:N ratio (<20:1), low C:P ratio (<300:1), and high Ca content (>1.0 mg-g^{-1}). In turn, this may create feedbacks which favour other 'fast turnover' compatible organisms. Understanding whether such linkages exist in reclaimed ecosystems could improve the design and implementation of ecological restoration schemes, enabling practitioners to assemble plants, trees, and composts that are complementary.

2.4 Woodland establishment on reclaimed land

2.4.1 The role of Community Forests in land reclamation to woodland end use

Between 1988 and 1994, over 30% of land reclaimed in England was converted to forestry and amenity end-use, signaling the growing importance of trees for repairing degraded land (Simpson, 1998). In an effort to expand UK woodland cover by planting on reclaimed land, the Forestry Commission created the Community Forest Programme (CFP) in 1989 (Mackay and Hesketh, 1998; Moffat, 2001). The CFP is a long-term woodland creation project dedicating 500,000 ha of land at the urban fringe to formation of a nationwide urban forest network. Significant quantities of degraded land are located within the CFP boundary, forming a common association with woodland creation using reclaimed land (Perry and Handley, 2000; Moffat, 2001). As an example, at Thames Chase Forest, 70% of land planned for woodland was formerly used for mining and landfill. At present, the CFP is set to expand further under a 25-year plan to create a 'Northern Forest' stretching coast to coast from Liverpool to Hull (NFM, 2017)**(FIGURE 2.9).** Reclaimed land is set to play a key role, however past failings must be understood to ensure the success of future restoration (Moffat and Laing, 2003).

A key criticism of land reclamation to woodland is that planting schemes are unnecessarily complex - containing an over-abundance of tree species that are incompatible with each other and the site (Goodman, 1998; Moffat and Laing, 2003). This arises from the CFP's guiding principle, which requires practitioners to develop woodlands that perform multiple functions, and simultaneously repair degraded land (Mackay and Hesketh, 1998). According to Ward (1996) multipurpose forestry's essential roles are: 1) timber production; 2) habitat creation/wildlife conservation; 3) landscape quality; and 4) recreation for people. Recent CFP publications encourage an even greater number of potential functions and services, mounting increasing pressure on land reclaimed to woodland end-use (NFM, 2017; Nolan, 2017).

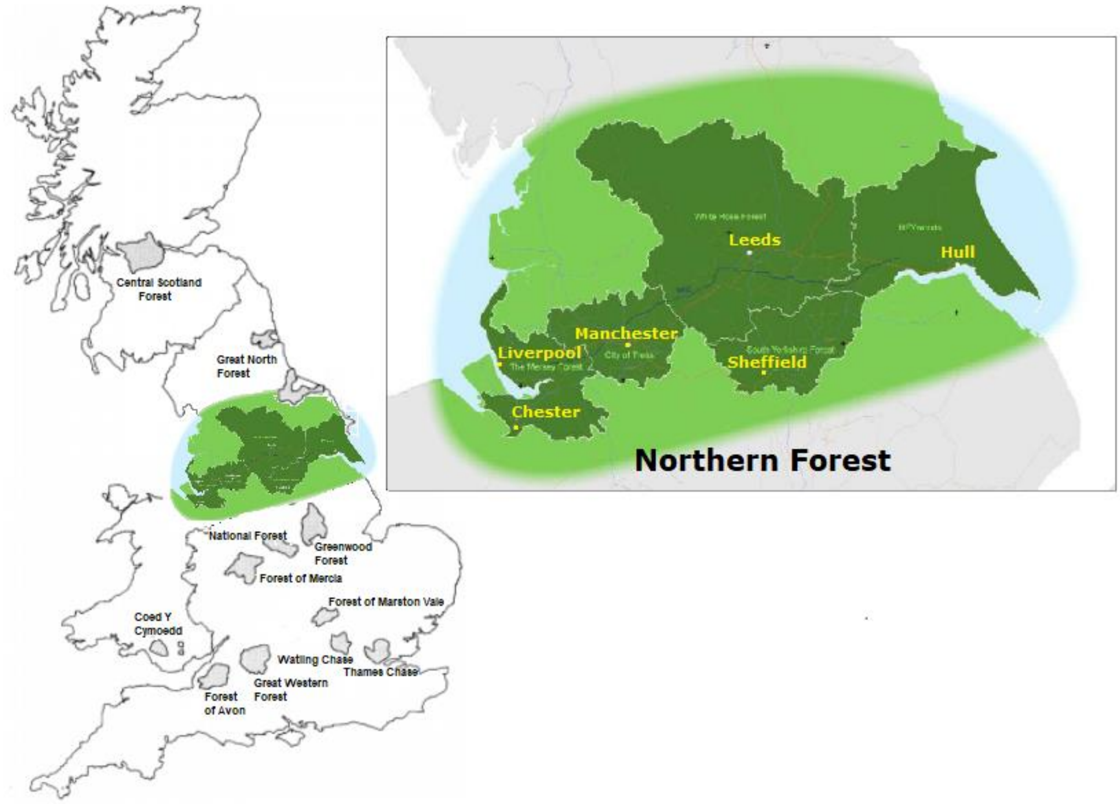
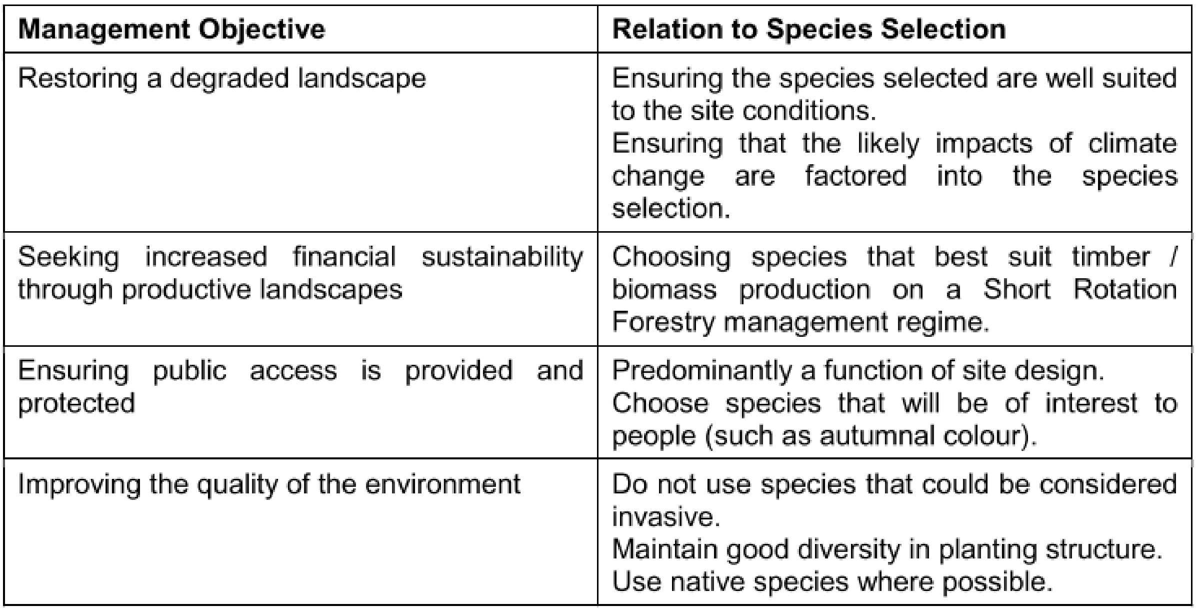


__FIGURE 2.9 –__ The Community Forest network of Britain, and the newly proposed 'Northern Forest' (Moffat, 2001; Nolan, 2017).

To create multi-functional forestry on reclaimed land, practitioners adopted a 'maximum species richness' approach to woodland design. This involves planting as many different tree and shrub species as possible, to increase the number of roles woodland performs (Kendle, 1996; Ward, 1996). The 'maximum species richness' approach also insures projects against tree losses, guaranteeing some species survive in the event of widespread failure (Dickinson et al., 2004). Yet, 'maximum species richness' can over-complicate planting designs, especially on small to moderately sized sites (<20 ha) as demonstrated by a recent project at Thames Chase Community Forest. The management objectives and species selection criteria used for this project link directly to multipurpose forestry's four original roles **(TABLE 2.3)**, resulting in 26 different tree and shrub species being selected for a modest 17 ha site (FC, 2010). Use of complex planting arrangements in challenging site conditions may be unsuitable for many newly created sites. The approach can shift practitioner focus away from restoring ecosystem functions, and increases risk of poor tree and soil performance (Moffat and Laing, 2003).



Management Objective	Relation to Species Selection
Restoring a degraded landscape	Ensuring the species selected are well suited to the site conditions. Ensuring that the likely impacts of climate change are factored into the species selection.
Seeking increased financial sustainability through productive landscapes	Choosing species that best suit timber / biomass production on a Short Rotation Forestry management regime.
Ensuring public access is provided and protected	Predominantly a function of site design. Choose species that will be of interest to people (such as autumnal colour).
Improving the quality of the environment	Do not use species that could be considered invasive. Maintain good diversity in planting structure. Use native species where possible.

TABLE 2.3 – *Management objectives and species selection criteria used for a recent land reclamation to woodland project at Thames Chase Community Forest. These link directly to the CFPs four multipurpose forestry roles, which are: 1). timber production, 2). habitat creation/wildlife conservation, 3). landscape quality, and 4). recreation for people. Emphasizing an over-abundance of roles risks convoluting tree planting designs, with 26 tree and shrub species selected for this site (FC, 2010).*

When the Community Forest Programme began, The National Urban Forestry Unit (NUFU) was created to co-ordinate the agencies involved in its creation. The NUFU championed a strategic ‘joined-up’ approach among agencies, emphasising technical excellence backed by research. Throughout the 1990s the NUFU adopted a minimalist strategy, advocating ‘*bold simple planting designs……’* that should *‘use only a limited number of tree species, typically 4-8 but often fewer’* (Goodman, 1998). The aim was to produce coherent designs, using a ‘minimum species richness’ approach. However, stands with too few species may be unable to perform some of the multipurpose forestry roles. Indeed, present research suggests species richness is not the fundamental factor for planting success (Aerts and Honnay, 2011; Pichancourt et al., 2014). Instead, reclamation practitioners should ensure stands possess *functional diversity*, i.e., a range of *complementary* functional traits, because a species ability to complement neighbouring trees in mixture is more important than increasing or decreasing species richness for its own sake (Aerts and Honnay, 2011; Morin et al., 2011; Pichancourt et al., 2014).

In reclamation, interactions between species are rarely considered, despite their demonstrable influence on ecological processes and functions. For example, in naturally formed soils Verheyen et al. (2016) documented how above ground carbon storage for a given tree species could be improved or inhibited depending on the species it was mixed with. Stanturf et al. (2014) described numerous configurations for mixed species planting, capable of influencing interactions between species, whilst Sercu et al. (2019) demonstrated how different configurations can influence soil development. Hättenschwiler and Gasser (2005) and Jacob et al. (2010) showed leaf litter decomposition is faster when tree species and fauna are compatible, impacting rates of soil formation and nutrient turnover. Toïgo et al. (2015) found certain mixtures of species can over yield when growing in unproductive highlands. Yet, when the same mixtures were planted in productive lowlands, stand performance declined suggesting certain mixtures perform better in specific environments. Despite this, a similar cohort of species have been planted in intimate mixture throughout numerous UK-based reclamation projects. Despite this, the ecological consequences of this widespread practice has rarely been investigated.

To guide future CFP planting on degraded land, Pichancourt et al. (2014) put forward three broad strategies, outlining that under certain site conditions multipurpose forestry becomes unsuitable. The strategies are: 1) promote natural regeneration; 2) use '*framework functional diversity'* planting; or 3) use '*maximum functional diversity*' planting. In strategy 1) extreme heat, drought, and/or exposure limit the pool of available species. Eventually, too few species can tolerate site conditions, rendering multipurpose forestry impossible. Natural regeneration is therefore the primary method for recruiting trees. Strategy 2) is comparable to NUFUs '*minimum species richness'* and involves planting a 'framework' of fast-growing trees. These form a protective overstory, creating suitable conditions for a wider range of shade tolerant species in their understory. Strategy 3) is comparable to CFP's *'maximum species richness'*, with multiple species planted at the same time to promote greater survival. However, unlike conventional CFP planting, *complementary* species are planted based on contrasting *functional traits*. In summary, when site conditions deteriorate species choices become increasingly restricted, with strategy 3), 2), then 1) employed as site conditions continue to decline. And wherever trees are actively planted, functional diversity should be used to encourage complementary planting designs **(FIGURE 2.10)**.

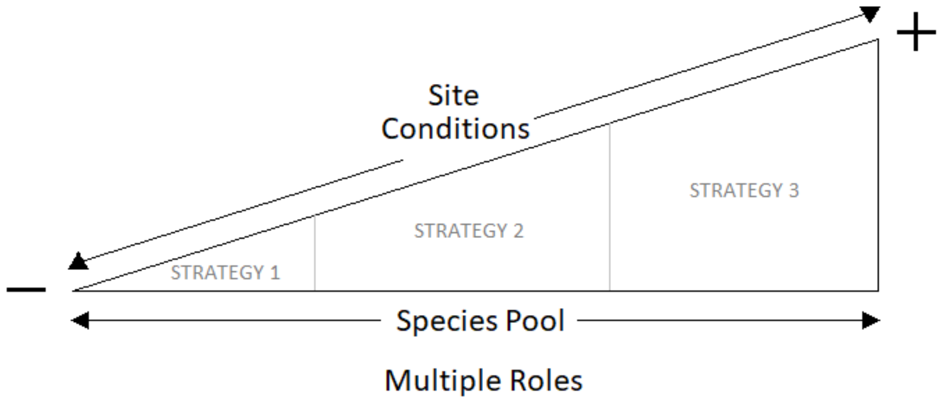


***FIGURE 2.10 –** Multipurpose forestry depends on multiple species, but site conditions dictate which species will grow. As site conditions improve, the species pool and number of potential functions grows **(+)**. When site conditions decline, the species pool and number of potential functions contracts **(-)**. Strategy 1 equals fewer species and functions. Strategy 3 equals more species and functions.*

These findings highlight that getting '*the right mixture, in the right place*' can be crucial to success or failure for multifunctional forestry in reclamation. What constitutes 'the right mixture' depends on species suitability to site conditions, and whether interspecies functional traits are broadly complementary. Functional traits which determine species suitability for mixture include contrasting growth rates, opposing requirements for light and shade, complementary foliar chemistry, and similarities and differences in speed of resource acquisition and nutrient turnover (Piotto, 2008; Rajapaksha et al., 2013; Reich, 2014; Oldfield et al., 2015). In addition, height at maturity, seed mass, leaf mass, and wood density are useful morphological characteristics which indicate species spatial requirements and demand for water and sunlight (Pichancourt et al., 2014; Stanturf et al., 2014). Examining tree performance in mixed stands at different reclamation projects, could improve future planting designs, improve tree performance, and widen the pool of species used in land reclamation.

2.5 Aims and objectives

The literature review demonstrated that a single, moderate application of composted green waste (CGW) promotes growth of certain tree species and can improve soil structure, moisture retention, nutrient status, and other physical, chemical, and biological soil parameters (WRAP, 2009; WRAP, 2010; WRAP, 2011b). However, to date, existing research has focused on CGW short-term impacts over periods of 0 to 4 years, covering the initial stages of woodland establishment only. To extend knowledge beyond this initial phase the present study will:

Explore CGW legacy impacts (defined as 5+ years) on provision of ecosystem services related to tree growth, soil formation, earthworm activity, and soil organic carbon fate.

1. Determine CGW long term impact on above-ground tree growth, biomass production, and foliar nutrient content at reclaimed sites;

2. Record physical, chemical, and biological attributes of CGW-amended soils to identify potential links between CGW application rate, CGW incorporation depth, soil formation, and carbon storage;

3. Determine how earthworm community composition, density, and biomass impacts pedogenesis in CGW-amended soils;

4. Outline CGW and earthworms synergistic impact on supporting and regulating ecosystem services

CHAPTER 3. SITE IDENTIFICATION AND SELECTION

3.1 Introduction

In order to identify a range of sites capable of demonstrating composted green wastes (CGW) legacy impact on tree growth, soil development, carbon storage, and earthworm activity, a process of selecting and evaluating previously reclaimed sites was undertaken. The following chapter summarises this process, which led to identification of 70 sites amended 5 or more years ago. Several were chosen for further study and the reasons why are formally explained.

3.2 Identification of suitable reclamation projects

To understand the available site resource, a desktop survey was conducted. This survey focused on journal articles and technical papers produced by the Forestry Commission (FC), Waste & Resources Action Programme (WRAP), Newlands Regeneration Programme, and the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER). These organisations have been consistently involved in UK land regeneration, and have published case studies, technical guidance, and formal research detailing multiple reclamation projects where organic amendments (OA) were utilised. Consequently, their publications became a core resource, providing details on multiple reclamation sites suitable for this project. When searching these documents, three criteria, presented as the questions below, were used to establish site suitability for further study: 1) Was restoration completed five years ago or more? 2) Did trees form a significant part of site vegetation? 3) Were organic amendments applied? If the answers to all these questions was “yes”, the site was added to a list of projects that might be investigated further. If OA had not been applied, the site was added, but as a control project to compare amended with unamended sites.

The process of identifying potential study sites was iterative and involved combining information from published documents, internet resources, and satellite imagery. It began by taking site names and other information from the core resources listed in **APPENDIX X**. To ensure all available information was captured for each site, a search engine query was performed by entering the site name into Google and Bing. This query returned multiple online resources, including planning applications, newspaper articles, and websites devoted to nature-based recreation. Search engines also provided geographical information on each project including addresses, postcodes, and nearby towns enabling sites to be geo-located and visually inspected using satellite imagery. Google Maps, Bing Maps, and UK Grid Reference Finder provided the necessary satellite imagery. This was important for confirming the continued existence of a site, the extent of tree cover, and resulted in the discovery of previously established field experiments still standing at numerous sites. All available information was entered into a spreadsheet using the categories provided by **TABLE 3.4.**

CATEGORY	DESCRIPTION / EXAMPLES
Site Name	Documented site name
UK Geographical Region	North East England; Central Scotland
UK Grid Reference	NT 00300 63100
Former Land Use	Landfill; Colliery; Quarry
Restored Land Use	Grassland; Woodland; Golf Course
Size (ha)	Hectares
Reclamation Start	Year reclamation began
Reclamation End	Year reclamation ended
Time Since Restoration	Time elapsed since completion
Tree Species	Tree species planted
Soil Texture	Sand; Silt; Clay; Loam
Soil Cultivation Technique	Loose Tipping; Ripping; Spreading
Organic Amendment Type	CGW, Sewage Sludge; Paper Crumble
OA Application Method	Surface Speading; Fully Incorporated
OA Incorporation Depth (mm)	300; 500; 750; 1000
OA Quantity Applied (t/ha)	50, 100, 250, 500, 1000
Fauna Added to Soil (Y/N)	Were soils innoculated with earthworms
Experiment On-site (Y/N)	Does a formal experiment still exist on-site
References	Documents/websites info sourced from

TABLE 3.4 – Categories used to collect and summarise information on 70 potential study sites

Once site information was tabulated and summarised into a larger 'matrix' of sites (**APPENDIX X**), it was analysed using the 'filter' and 'pivot table' functions in Excel. Projects were grouped according to shared similarities, including site age, former land-use, and organic amendment applied. **TABLE 3.5** summarises the results of the desktop survey, and outlines which characteristics were most common among the 70 strong sample of sites. 'Landfill' followed by 'Colliery' were the most common former land-uses, whilst 'Composted Green Waste' followed by 'Sewage Waste' were the most common organic amendments applied. Understanding this meant a series of field surveys could be planned with the aim of locating a smaller number of complementary sites; namely, those reclaimed using similar amendments and soil manufacturing techniques, but with differing site conditions and time elapsed since restoration. It was hoped this blend of similarities and differences would reveal OA temporal impacts on tree growth and soil development, under differing but comparable site conditions. The following section summarises findings of the field survey and explains why the final five sites were ultimately chosen.

Reclamation Site Summary									
	Total number of sites	Different OA used	Avg. years since restoration	No. sites with OA	No. sites without OA	No. sites with multiple OA	Sites containing field experiments		
	70	9	14	49	18	18	14		
Organic Amendment Summary									
OA type	Composted Green Waste	On-site soil re-use	Sewage wastes	No amendment	Paper Crumble	Anaerobic Digestate	Imported topsoil	Woodchip	Other
No. times OA used	31↑	17	17	13	10	4	4	3 ↓	4
Avg. years since application	9	11	13	25 ↑	9	5 ↓	15	7	12
Fauna added	2	1	1		1	1			
Former Land Use Summary									
Land use type	Landfill	Colliery	Quarry	Industrial	Spoil tip	Steelworks	**KEY**		
No. sites	26↑	25	8	4	3 ↓	4	Highest Overall Value ↑		
Avg. years since reclamation	15	16 ↑	13	14	6 ↓	9	Lowest Overall Value ↓		
x CGW applied	13↑	10	2	1↓	3	3	Category Selected for study		
x Fauna added	4 ↑	1				1↓			
Existing field experiment	10↑	2	1		1 ↓	1↓			

TABLE 3.5 – Summary of reclamation projects identified during the search for potential sites. A total of 70 projects were found. Those highlighted in RED were the most frequently occurring types of sites under the categories 'land-use' and 'organic amendment'.

3.3 Rationale for selection of the final study sites

To decide which sites should be chosen for in depth study, field surveys were conducted at nineteen reclamation projects **(FIGURE 3.1, FIGURE 3.2, FIGURE 3.3)**. The purpose was to confirm details that desktop surveys could not, including tree species planted, accessibility for fieldwork, patterns of public usage, and geography of surrounding land. The nineteen projects were chosen on the basis they possessed characteristics desired for further study. Thus, all were former landfills or collieries treated with sewage sludge or composed green waste and received woodland planting. Sites were located in different UK regions, however it remained unclear which would be included in the final study. One line of thinking was that sewage sludge and CGW may impact tree growth and soil development in different ways. Hence, comparing highly similar sites treated with different organic amendments might reveal which amendment was most effective.

FIGURE 3.1 – Reclaimed sites visited in North-West England. ***ELEVEN*** *were screened as part of the site selection process.* ***YELLOW*** *pins are former landfills,* ***BLACK*** *pins are former colliery sites,* ***RED*** *pins are nearby cities. The Eleventh site (Cross Lane) is not illustrated but was adjacent to Bidston Moss. No North-West sites were chosen for in-depth study.*

FIGURE 3.2 – *Reclaimed sites visited in Central Scotland.* ***FIVE*** *sites were screened as part of the selection process.* ***YELLOW*** *pins are former landfills,* ***BLACK*** *pins are former colliery and shale disposal sites,* ***RED*** *pins are nearby cities.*

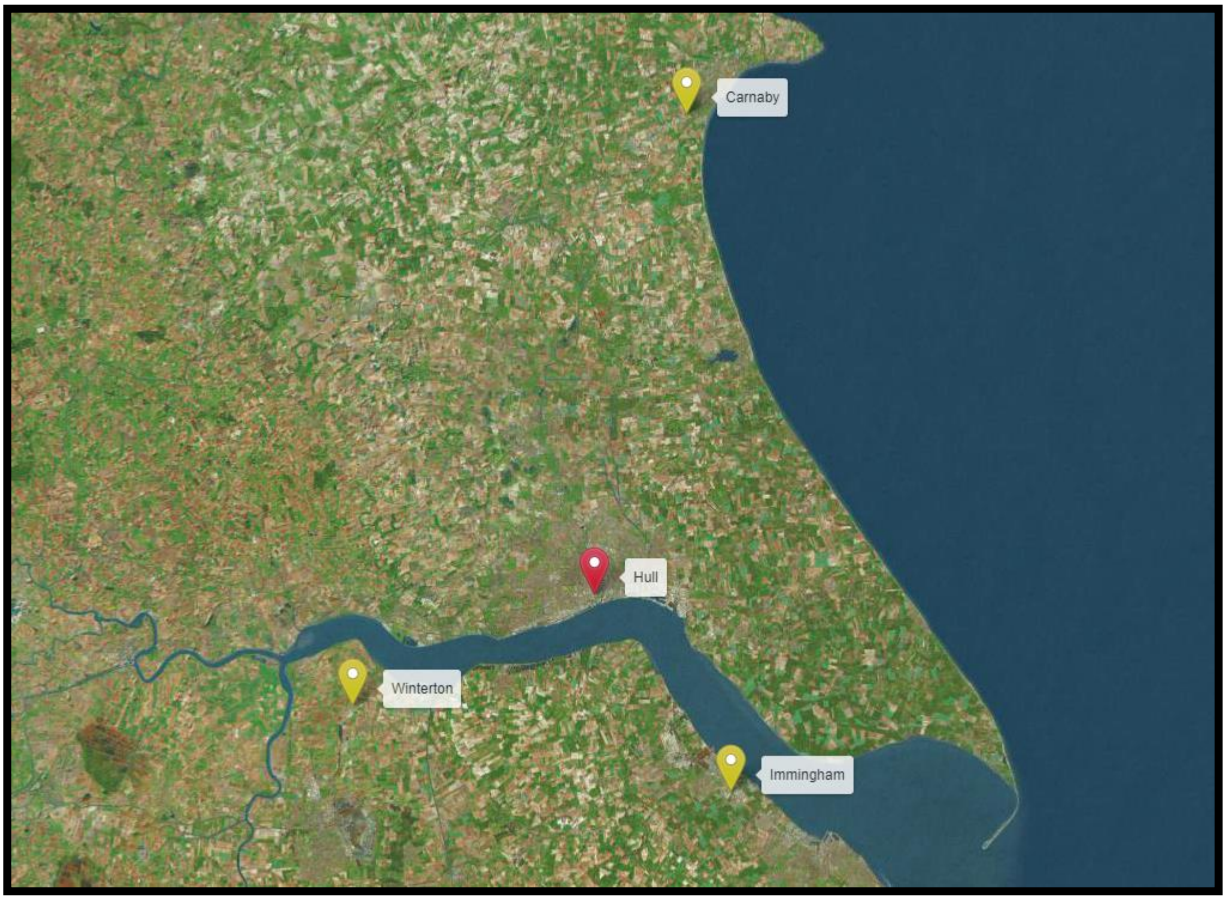


FIGURE 3.3 – *Reclaimed sites visited in North-East England.* ***THREE*** *different sites were screened as part of the site selection process.* ***YELLOW*** *pins are former landfills,* ***RED*** *pins are nearby cities.*

Unfortunately field surveys confirmed the available site resource could not support a comparison of amendments. This came to light as sites in North-West England were surveyed to locate woodland blocks with and without a variety of organic amendments, so that growth in each could be contrasted and compared **(FIGURE 3.1)**. However, in many cases exact locations of amended areas could not be determined using field surveys or documentation. Further, in many cases slightly different tree mixtures had been planted at different times, making comparison between blocks and sites difficult. Consequently, attention shifted to sites with pre-existing field experiments, as these had been formally documented and contained pre-defined controls and treatments. Accordingly additional surveys were conducted in Central Scotland and North-East England at sites with pre-existing field experiments **(FIGURE 3.2, FIGURE 3.3).** It was felt formal experiments would facilitate more rigorous study and reveal CGW impact under well-defined and documented conditions. Thus, using a combination of published research, archived literature from Forest Research, and field visits, details on several historic experiments were re-constructed. The use of existing field experiments meant the location, timing, quantity, and depth of compost application was known and could be more easily compared with unamended controls.

After visiting eight sites, the existence of multiple previously established experiments was confirmed. Subsequently, five were chosen for in-depth study. These five shared complimentary attributes, creating a coherent group of study sites. For example, all were established on landfill, exposing each experiment to similar soil manufacturing and reclamation techniques. All were amended with CGW only, maintaining consistency and comparability between locations. Tree species were also similar, and where they differed, analogues from the same genus, or species with similar functional roles had been planted instead.

In contrast, several differences broadened the research scope. For example, experiments were in different UK regions, enabling CGW performance under contrasting site conditions to be observed. Soil biological treatments varied, with earthworms actively introduced at one site but not considered in experimental designs elsewhere. Importantly, experiments were established several years apart, bringing the impact of time on site development into the research. The final five study sites are summarized in **FIGURE 3.4** and **TABLE 3.6**, with detailed descriptions provided in their dedicated chapters.

FIGURE 3.4 – Geographical location and names of the final FIVE study sites. One was in South-East England, two in North-East England, and two in Central Scotland under the Greenoakhill pin.

Experiment Overview					
Site Name	**Ingrebourne Hill**	**Greenoakhill North**	**Greenoakhill South**	**Winterton**	**Immingham**
Location	Rainham (Essex)	Glasgow (Scotland)	Glasgow (Scotland)	Scunthorpe (Humberside)	Scunthorpe (Humberside)
Grid Reference	TQ 52574 83209	NS 66622 62760	NS 66412 62135	SE 91431 20056	TA 20310 14086
Former Land-use	Landfill	Landfill	Landfill	Landfill	Landfill
Current Land-use	Community Woodland	Active Landfill	Community Woodland	Active Landfill	Active Landfill
Experiment Established	2013	2011	2011	1998	1998
Year Sampled	2018	2019	2019	2018/19	2018/19
Age at Sampling (yrs)	5	8	8	20	20
Tree Species	Italian alder	Silver birch	Silver birch	Italian alder	Italian alder
	Norway maple			Sycamore	Sycamore
Soil Type	Sandy Clay Loam	Sandy Silt	Clay Loam	Sandy Loam	Heavy Clay
Exposure	Very Exposed	Very Exposed	Moderate	Moderate	Very Exposed
Annual Rainfall (mm)	550 - 560	1100 - 1250	1100 - 1250	600 - 700	600 - 700
Annual Sunshine (hours)	1600 - 1650	1200 - 1300	1200 - 1300	1500 - 1550	1500 - 1550
Avg. Min - Max Temp (C)	6 - 15	6 - 13	6 - 13	6.5 - 14	6.5 - 14

TABLE 3.6 – *Overview of the chosen field experiments. Age, location, soil type, site conditions, time of sampling are shown.*

CHAPTER 4. GENERAL MATERIALS AND METHODS

4.1 Overview and rationale

This chapter describes the techniques used to collect and analyse data from the five UK study sites and explains the rationale underpinning the methodology. Each study site contained a field experiment designed to investigate CGW impacts on tree growth and soil quality. The ecosystem services framework was used to conceptualise CGW potential impacts and where these might be detected, helping structure on-site sampling. The sampling strategy was conceptualised around three supporting and one regulating ecosystem service. These were, primary production, soil formation, nutrient cycling, and carbon storage as the regulating service. Each field measurement taken corresponded to one or more ecosystem service, thus acting as a proxy for these **(TABLE 4.1)**. To maintain consistency when measuring, collecting, processing, and analysing data, the exact same measurements were taken from every plot at each study site. Thus, if a field experiment contained 40 plots, plot scale sampling **(FIGURE 4.1)** was conducted 40 times, ensuring data collected from different sites remained comparable.

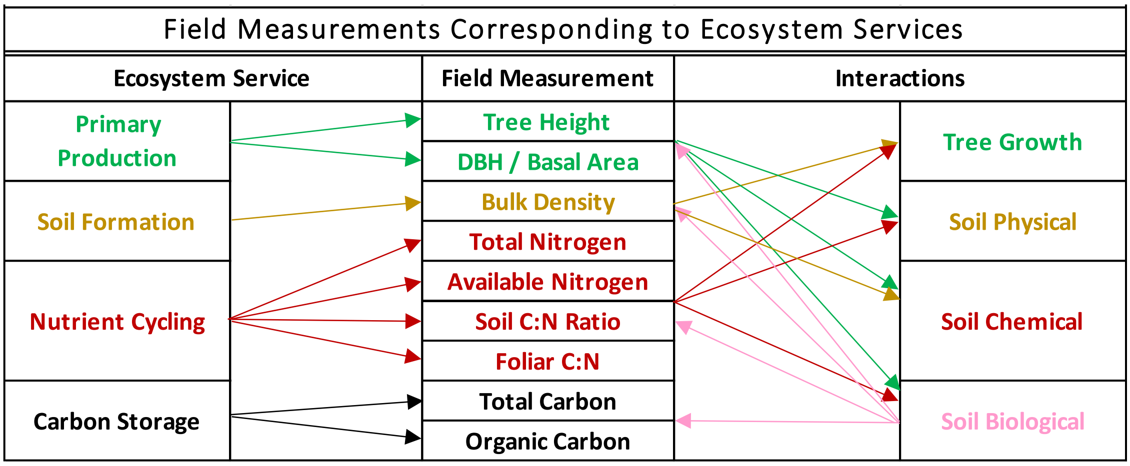


__TABLE 4.1 –__ Ecosystem services and their corresponding field measurements. Measurements taken (central column) were considered to represent specific services (left-hand column). The right-hand column shows the relationships and interactions between different ecological parameters. For example, 'Tree Height' and 'DBH / Basal Area' depend on interactions with soil physical, chemical, and biological parameters. Conversely, soil biology influences all field measurements and associated ecosystem services.

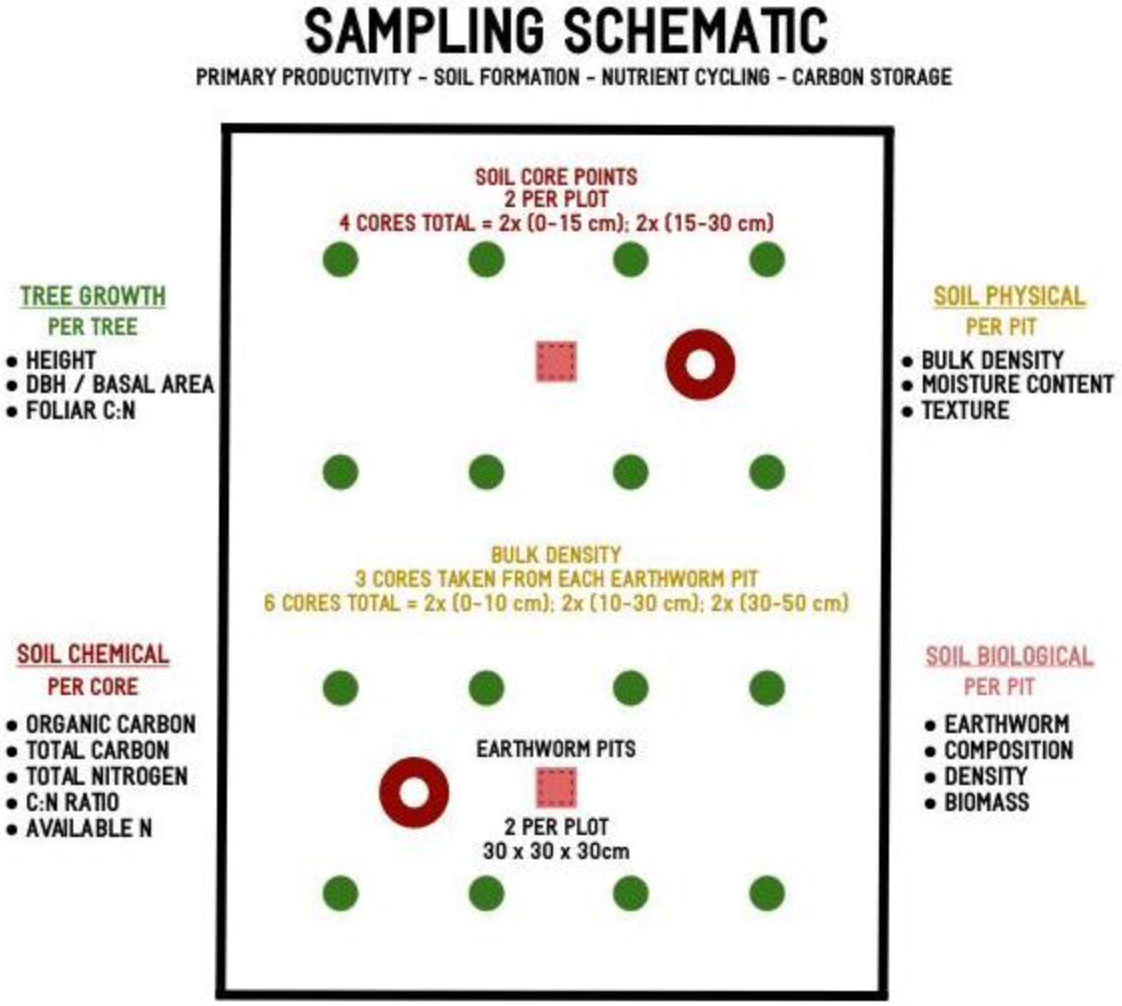


FIGURE 4.1 – *Schematic illustrating plot-scale sampling strategy. The same measurements were taken from every plot in every field experiment studied, generating a range of tree, soil, and earthworm related data. In addition to earthworm sampling, earthworm pits were used to collect soil bulk density samples.*

In line with the aims and objectives of this thesis **(1.1 Aims and objectives)**, the overall hypothesis put forward was that CGW would have a synergistic impact on bio-physical processes, positively influencing tree growth, earthworm activity, nutrient availability, and carbon storage. In essence, trees and earthworms would provide a bio-physical mechanism, through which CGW impacts on site development could be observed. The following sections detail the techniques used to collect, process, and analyse sample material and data collected from the five field experiments.

4.2 Tree sampling and analysis

Tree growth was a key variable of interest for this research project. In reclaimed soils, tree growth and survival are often poor due to compaction, inadequate moisture retention, nutrient deficiencies, extreme pH, contamination, and absence of soil organisms. Organic amendments can help ameliorate this, potentially improving tree performance (Moffat, 1996b; Moffat and Laing, 2003; Kilbride, 2014). To identify whether CGW application had indeed resulted in improved tree growth; tree survival, height, stem diameter, and foliar C:N were assessed.

4.2.1 Tree growth measurement

All measurements of tree height were completed in accordance with Forest Research Standard Operational Procedure (SOP) 0232 for “Determining tree height assessment points”. SOP 0232 specifies the total height of a standing tree (in most cases) is the vertical distance from the base of the tree to the uppermost point. Regardless of whether a tree was leaning heavily or straight, measurement was taken from ground level and traced vertically until it was level with the trees uppermost point. In the case of smaller, younger trees, if the leader was not vertical or the stem not straight, the tree was carefully straightened to provide more accurate measurement.

Indeed, because Ingrebourne Hill contained small young trees aged 5 years at the time of measurement, techniques used to obtain tree height and stem diameter were different from the other sites. Tree height at Ingrebourne Hill was obtained using a 5 m telescopic measuring rod. This required two people, one holding the telescopic rod against the tree, whilst the other observed and recorded the height reading. In addition, given the small stem size of trees at Ingrebourne Hill, calipers were used to measure stem diameter. Two caliper readings were taken from the ‘ground-line’, defined as 2 cm above the soil surface on the main stem (Menes and Mohammed, 1995). These were taken at right angles to each other, to account for asymmetrical stem growth with mean values reported.

At the other four sites **(TABLE 3.6)** height was measured using a Haglof Vertex IV digital hypsometer with ultrasonic transponder, during winter when foliage was minimal. Readings were taken from a single observation point (minimum of 5 m from tree), wherein the entire tree and transponder remained visible. The transponder was attached to the trees main stem at 1.3 m, with this height logged in the hypsometer to ensure height from ground was known. The hypsometer was aimed at the transponder to calculate distance, then at the trees uppermost tip to calculate height. Ground slope from the point of observation was calculated automatically. By bringing all these measurements together, the hypsometer accurately calculates height. Next, stem diameter was recorded using a circumference tape manufactured by Richter. The tape was wrapped around the stem at a height of 1.3 m, and the reading recorded. Circumference (C) was later converted to diameter (d) using the formula $d = C / \pi$.

4.2.2 Foliar sampling

To assess CGW impacts on nutrient cycling and primary production, foliar samples were collected from all five sites. Sampling was conducted between July and August when nutrient concentrations were steady. To create a bulk sample for each experimental plot, foliage was cut from the outer crown of four different trees, from each of the four cardinal directions. This ensured foliage collected was representative and had been exposed to full sunlight. Because tree height varied significantly between sites, different equipment was used to cut and collect leaves. For smaller trees, leaves were cut directly from branches using hand-held secateurs. For larger trees, small branches were sawn off with a 7.7 m Silky Hayate Telescopic Saw, then foliage was cut randomly from downed branches. A minimum of 100 leaves minus petioles were collected from each plot. Leaves were placed into numbered plastic zip-lock bags and stored in cool boxes, before being transported to Alice Holt and UCLan laboratories for refrigeration.

4.2.3 Foliar processing

Given the volume of leaves generated from each site, foliage was refrigerated at 4°C for a maximum of 14 days whilst processing was conducted. To prepare foliar material for instrumental analysis leaves were dried, homogenised, and ground. Firstly, to remove all moisture leaves were placed in aluminium foil trays and oven dried at 70°C for 48 hours. Once dry, leaves were bulked to create a single plot-representative sample. Bulking and homogenisation was achieved by placing dry leaves from a given plot into a single plastic zip lock-bag and breaking them into smaller pieces by hand. Pieces were then blended in a Magimix Compact 3100 Blender and sieved to obtain leaf particles of 1 – 2 mm size. A sub-sample was taken and ground to very fine powder using a Changsha Tianchuang SXQM planetary ball mill. Leaves were then ready for instrumental analysis.

4.2.4 Foliar carbon and nitrogen analysis

Foliar samples were analysed for % CHN and C:N ratio using a Thermo-Scientific CHNS Organic Elemental Analyser. Samples were mass determined by weighing 2-3 mg of processed material into aluminium foil capsules. These were weighed using a Mettler Toledo XP6 microbalance, sealed using forceps, weight logged, then loaded into the CHNS analyser. The instrument was calibrated using 2,5-Bisthiophene (5-tert-butyl-2-benzo-oxazol-2-yl) (BBOT) standard prior to analysis. Samples were pyrolyzed at a temperature of 950 °C and directed through a pre-packed copper column using helium carrier gas (flow rate 140 ml/min), then into a GC column where C H N and S were separated and detected using a Thermal Conductivity Meter (TCD). Results were presented as C:N ratio.

4.3 Soil sampling and analysis

To establish whether CGW had impacted soil formation, nutrient cycling, and carbon storage, multiple soil-based variables were assessed. These were: soil bulk density, total carbon, total nitrogen, available nitrogen, and organic matter content **(TABLE 4.1)**. The techniques used to collect, process, and analyse soil physical and chemical samples differed, and are described separately in the following sections.

4.3.1 Soil physical sampling

First, bulk density (BD) was the main physical variable analysed but also provided soil moisture content. Depending on the site, BD sampling was completed in Spring (April/May) or Autumn (September/October) alongside earthworm sampling to reduce digging on-site. A hammer and 100 cm^3 cylindrical steel core were used to extract BD cores (Robertson et al., 1999). Two 30 x 30 x 40 cm^3 pits were excavated in each plot. Cores were hammered into the side and base of each pit with a mallet and dug out with a knife **(FIGURE 4.1, FIGURE 4.2)**. Three sampling depths were assessed (0 - 10, 10 - 30, and 30 - 50 cm) generating two replicates for each depth (e.g. 2 pits dug, 3 cores from each = 6 cores per plot). BD cores were placed in plastic zip lock bags and stored in a cool dark place. They were transported off-site within 5 days of extraction for refrigeration, processing, and analysis.

4.3.2 Bulk density processing and analysis

BD cores were oven dried to establish bulk density, moisture content, and total porosity. Each 100 cm^3 soil core was placed into a pre-weighed aluminium foil tray and weighed. Similarly, stones >2 cm were removed and weighed, providing individual weights for the tray, stones, and soil. The sample was then oven dried at 105°C for 24 hours and weighed again. BD was the soil mass remaining after drying. Moisture content was the mass lost through drying. Total porosity for all samples was calculated using an assumed particle density. The following equations were used:

$$\underline{\text{Bulk Density } (g/cm^3) = W / V}$$

where W = oven dry soil weight (g)

V = volume of core in cm^3

$$\underline{\text{Soil Moisture } (\%) = (W - D) / (W) \times 100}$$

where W = soil wet weight (g)

D = soil dry weight (g)

$$\underline{\text{Total Porosity } (\%) = [1 - (\text{bulk density} / \text{particle density})] \times 100}$$

particle density assumed as = 2.65 g/cm³

FIGURE 4.2 – *Example of pit used for bulk density (BD) and earthworm sampling. A 30 x 30 x 30 cm volume of soil was excavated, and hand sorted for earthworms. Following this, a further 10 cm depth of soil material was excavated for BD sampling. The deepest BD core (30-50 cm) was extracted from the base of the pit.*

4.3.3 Soil chemical sampling

The second sampling method involved acquisition of soils for chemical analysis, so that soil nitrogen availability and carbon storage could be determined. Soil cores were extracted from 0 - 15 and 15 - 30 cm depth using a Dutch auger fitted with 15 cm attachment. To ensure data was representative of plot-level, soil was taken from two different sampling points in each plot. These were positioned at opposite ends of the plot, but away from boundaries to reduce edge effects. This generated two replicates per plot for each sampling depth (i.e., 4 cores per plot) **(FIGURE 4.1)**. Cores for chemical analyses were sealed in zip locked bags and stored in a cool dark place. They were returned to the laboratory on the same day as extraction for refrigeration and processing to reduce microbial transformation of NO_3^- and NH_4^+ (Robertson et al., 1999).

4.3.4 Soil chemical processing and analysis

Samples were refrigerated at 4°C to keep them field moist whilst awaiting analysis. Both field moist and dry homogenised soils were needed to complete the required chemical analyses. Field moist sub-samples were taken immediately upon return to the lab so that available nitrogen could be assayed. Once complete, soils were dried and homogenised to prepare for the remaining analyses of total carbon, total nitrogen, and organic matter content. Drying was completed by placing soils in aluminium trays and air drying at room temperature for 7-days. This method was adopted to protect organic matter from degradation prior to loss on ignition. Soils were turned halfway through to ensure samples were fully dry. Large stones were removed by hand then dry soils were crushed in a motorised blade grinder which sieved out particles >2 mm in size. At this stage replicate samples were bulked and sieved to 0.5 mm, then poured into plastic zip lock bags and mixed to ensure complete homogenisation.

4.3.4.1 Available Nitrogen

Fresh soils were needed to measure inorganic available nitrogen, with a 1 M KCL-extraction, adapted from Robertson et al. (1999), used to extract NH_4^+ and NO_3^-. Briefly, 10 g fresh soil was added to 50 ml KCL solution in a 120 ml polyethylene screw cap bottle. These were shaken for 2 minutes forcing soil bound nitrogen ions into the KCL. Samples were left to settle overnight, re-shaken for 1 minute the following day, then left to settle for 45 minutes. A syringe with a 0.45 µm attached filter was used to extract 15 ml clear, settled solution into a 15 ml sample vial. These were refrigerated and transferred to Rothamsted Research laboratory for analysis using a Skalar San^{++} anion colourimetric continuous flow analyser. Residual stone content (g) was calculated by washing sediment through a 2 mm sieve. Stones were weighed so available-N data could be adjusted for stone content. BD values were used to calculate available-N present per volume of soil (kg-ha^{-1}).

4.3.4.2 Loss on ignition

Loss on ignition (LOI) was used to estimate soil organic matter content (%), following a method adapted from Rowell (1994). Using a 3 decimal place balance, 2 g soil was weighed into a pre-weighed crucible so that masses for crucible and soil were known separately. Crucibles were placed in a furnace and heated to 550 °C for 3 hours (Hoogsteen et al., 2015). Once complete, samples were placed in a desiccator to cool before post-combustion soil mass was determined. Where site soils contained large quantities of clay or were reported as having high pH a correction factor was applied to all LOI values (divided by 2). This mitigated overestimation of results caused by residual moisture in clay or calcium rich soils.

4.3.4.3 Carbon and nitrogen analysis

For soil carbon and nitrogen analysis, the same process as section 4.2.4 Foliar carbon and nitrogen analysis was followed, with the CHNS analyser presenting results as C:N ratios and percentage carbon and nitrogen. BD values were used to convert percentage C and N to mass per known volume of soil. Results were then reported on a t-ha^{-1} basis.

4.4 Earthworm sampling and analysis

Earthworms were sampled to ascertain whether treatment with CGW had altered earthworm density and community structure. Likewise, processes mediated by earthworms including primary production, nutrient cycling, and carbon storage may also have been affected by CGW addition. Earthworm sampling followed the methods described by Butt and Grigoropoulou (2010). Depending on the site, sampling was conducted in either Spring (April/May) or Autumn (September/October), when temperatures were above 6 – 9 °C and soils were relatively moist. Two earthworm pits were excavated in each plot to ensure a representative community of worms was collected. For each sample, a 0.1 m^2 quadrat was placed on the soil surface. Approximately 30 cm of soil beneath the quadrat was removed with a spade, placed on a polyethylene sheet and hand-sorted for earthworms. Collected earthworms were placed in pre-numbered plastic bottles and preserved in 4% formaldehyde solution, then stored on-site out of direct sunlight in sealed plastic insulated cool boxes. Earthworms were transported to the laboratory for identification within one week of extraction.

4.4.1 Earthworm identification and analysis

All adult worms were identified to species level following the key of Sims and Gerard (1999). Juvenile worms were identified to genus level only. As a result, juvenile numbers were not reported as neither species nor ecological group could be determined. Identification was completed using a stereo microscope with adjustable zoom (X10 – X30 magnification), an external lamp, and nonserrated metal forceps with rounded tip for handling earthworms. Earthworms were rinsed and placed in a water-filled Petri dish to keep them moist whilst being identified. Once complete, specimens were dried on absorbent paper to remove excess moisture, then weighed to determine biomass. Because plot level populations were assessed from two sampling points per plot, individuals were pooled together creating a single plot representative sample. Earthworms were separated into ecotype then analyses of numbers and biomasses per treatment were undertaken. Data was reported as ecotype to determine the effects of earthworm behaviour on tree growth and soil functioning.

4.5 Statistical Analysis

Statistical analysis was performed using two different open-source software packages. These were 'JASP' (Version 0.9.0.1) statistical software, followed by 'R – script' (Version 4.0.2) and its associated 'R – studio' user interface (Version 1.3.959). JASP was used for: one-way analysis of variance (ANOVA), repeated measures ANOVA (RMANOVA), and regression, whilst 'R' was used for principal component analysis. The statistical techniques used in JASP required data normality to be assessed in various ways. Firstly, histograms and Q-Q plots were used to graphically / visually evaluate whether data was normally distributed. Next, numerical values for skewness and kurtosis were considered. If either was above 1.96, data was deemed to deviate significantly from normality. Finally, Levene's test was used to evaluate statistical variance between groups. Where Levene's test was significant ($p < 0.05$), variance was deemed to be unequal violating the assumption of normality (Goss-Sampson, 2018). Where data was not normally distributed and log-distribution could not correct it, non-parametric alternatives were applied. Kruskal-Wallis non-parametric test followed by Dunn's post-hoc replaced one-way ANOVA. Non-parametric Friedman's test followed by Connover's post-hoc replaced one-way RMANOVA. (Goss-Sampson, 2018). And finally, to reduce type-1 error bonferroni corrections were applied to all significant results.

With compost, tree growth, earthworms, and other factors simultaneously driving site development, it was important to know which had the greatest influence on restoration processes. Thus, using the same data, principal component analysis (PCA) was performed in ‘R’ to reveal which variables were chiefly responsible for the effects revealed in JASP. PCA observes data in three-dimensional space by overlaying multiple linear equations (vectors) over the dataset. This allows PCA to explore which of the factors measured are driving the variance observed. Once dominant variables are identified and ranked by order of influence, PCA re-expresses all data around the dominant variables. Results are illustrated using a biplot which provides a visual representation of site dynamics. PCA is completely non-parametric, making no assumptions about normality or how data was recorded. However, PCA functions on the major assumption that all variables are linear. This simplifies the data by eliminating all secondary correlations, restricting the number of relationships and connections present. PCA is also a descriptive technique, meaning it explains only the data to which it is applied, and cannot be used to make inferences about a larger population (Shlens, 2014; Jolliffe and Cadima, 2016).

CHAPTER 5. ORGANIC AMENDMENTS IN THAMES CHASE COMMUNITY FOREST: Composted green wastes impact on tree growth, soil quality, and carbon storage at a former landfill in Essex 5 years after application

5.1 Ingrebourne Hill – Background

Ingrebourne Hill Community Woodland is a 54-ha area of land in Rainham, Essex, UK (Nat Grid Ref: TQ 52574 83209). The site forms part of the larger Thames Chase Community Forest (TCCF), a 104 km² area of countryside along the Thames Estuary in the London Essex borders **(FIGURE 5.1)**. From the beginning of the 20th Century, the natural landscape in TCCF has been impacted extensively by urban sprawl, industrial development, mining, and landfill. Since 1990, a concerted effort has been underway to restore natural landscapes throughout Thames Chase, with the Forestry Commission playing a key role. Thames Beat are the FC department responsible for managing TCCF. Along with Forest Research and commercial partners, TCCF target is to convert all areas of community forest affected by landfill, (namely 16% of the landscape) back to woodland end-use (Thames-Chase-Trust, 2014).

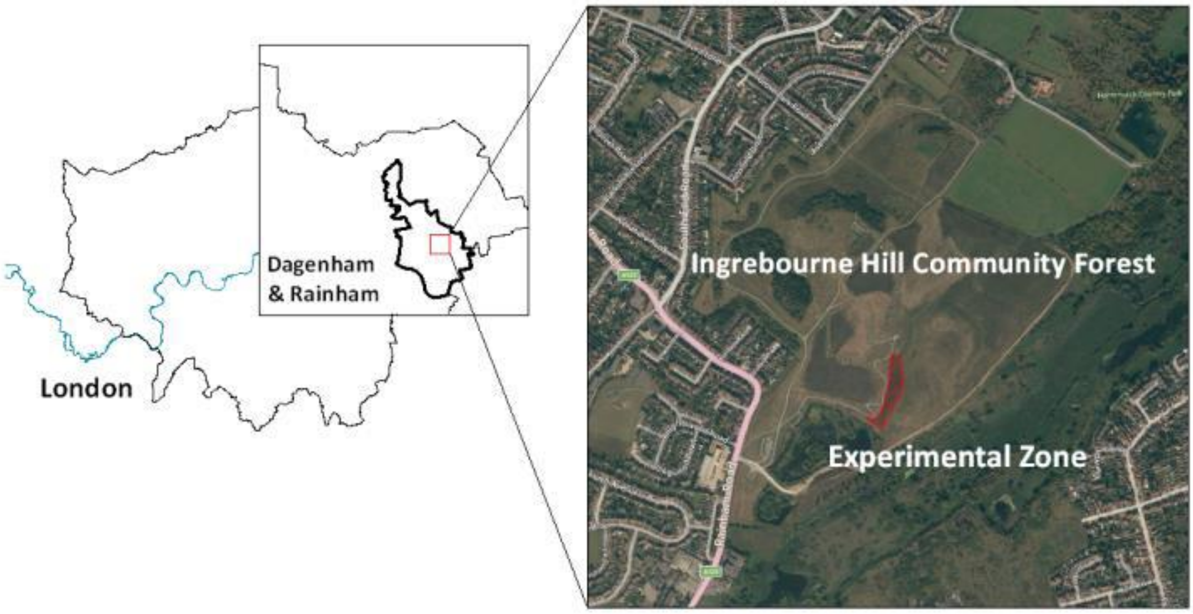


FIGURE 5.1 *– Geographical location of Ingrebourne Hill community woodland (Ashwood, 2016). The site is approximately 16-miles from central London. The field experiment highlighted in RED was established in 2013.*

5.2 Site Conditions

Annual climate data from 1981 – 2010 shows the area received an average of 1,600 to 1,650 hours of sunshine per year, a mean daily maximum temperature of 15 °C, and rainfall of 500 – 550 mm per annum (Met-Office, 2011c). This places Ingrebourne Hill in one of the warmest, driest, sunniest parts of the UK. In 2013 an experiment was established here at the crest of a hill on open ground. There are no natural barriers, landforms, or shelter belts protecting the experiment, leaving it exposed to prevailing wind and weather conditions **(FIGURE 5.2)**.

FIGURE 5.2 – *Experiment located at the crest of Ingrebourne Hill. Its unshaded and unprotected position allows sun and prevailing winds to evaporate plant and soil moisture. Given Norway maple (Acer platanoides) growth and survival was poor, the trees visible in the picture are mostly Italian alder (Alnus cordata). Nevertheless, to the far right a block of Norway maple is visible.*

5.3 Soil Materials

Soils at Ingrebourne Hill are described as a sandy clay loam with high stone content and are the product of industrial land-use and soil manufacturing (Heaven and Richardson, 2007). Following excavations for gravel in the 1940s and 1950s, the site functioned as a landfill for inert and putrescible waste (Thames-Chase-Trust, 2014). Between 1995 and 2007, the site was reclaimed in various stages, the aim being to re-instate ecological processes and improve opportunities for recreation through creation of open woodland habitat. A new soil was manufactured for Ingrebourne Hill by first laying a 0.8 – 1.8 m depth engineered clay cap over inert landfill material. Construction rubble screened to 0.5 – 0.8 mm was loose-tipped over the cap creating a plant growth and rooting medium (Heaven and Richardson, 2007). This manufactured medium contains fragments of brick, concrete, porcelain, plastic, glass, and metals including steel re-bar. Blocks of mixed species woodland were planted into this, however near complete tree mortality ensued.

Subsequent surveys in 2011 identified soils to be of poor physical and chemical quality. Regarding physical quality, heavy compaction created drought conditions at the soil surface restricting tree rooting to 0.5 – 0.6 m depth (Doick and Willoughby, 2011). Concerning chemical quality, soil organic matter (SOM) averaged 4.0%, the lowest quantity recommended for establishing trees on regenerated land (Foot and Sinnett, 2014). Moreover, 11 of 24 soil samples tested at or below the minimum recommended 0.2% total nitrogen per dry weight of soil. As a result, soil C:N ratios were high averaging 29:1. Qualitative soil biological observations were also made and earthworms were found in just 1 of 24 soil pits indicating soil biological quality was poor (Doick and Willoughby, 2011; Ashwood, 2014; Harris et al., 2014; Kilbride, 2014). Professionals involved in reclaiming Thames Chase also had low awareness of available soil improvement techniques further limiting project success (Ashwood et al., 2014). To improve manufactured soil quality, Thames Beat and Forest Research renewed their long-standing interest in organic amendment (OA) research.

5.4 Description of field experiment

To investigate whether composted green waste (CGW) could improve tree establishment, growth, and soil quality, Ashwood et al. (2018) established an experiment at Ingrebourne Hill in April 2013. Its purpose was to investigate CGW's impact on tree growth, soil development, and earthworm activity, with Thames Beat funding the experiment as a means of demonstrating best practice reclamation to partner organisations. After monitoring site development for 30-month period from April 2013 to October 2015, studies concluded interactions between earthworms and compost significantly increased soil organic carbon, plant nutrients, and tree growth (Ashwood et al., 2017; Ashwood et al., 2018). Yet, because the experimental duration was relatively short, peer review recommended CGW long-term impacts should be investigated further. The present project was a response to this, re-visiting Ingrebourne Hill to observe tree growth and soil developmental processes five years post establishment.

Before proceeding it was important to understand technical details about the experiment given its design was influenced by Ashwood et al. (2018) findings, and those generated by the current research. **FIGURE 5.3** shows the experimental layout which consisted of 5 blocks, each containing the 4 treatments set out randomly in a split plot design. The four treatments 'CONTROL', 'EARTHWORMS ONLY', 'COMPOST ONLY', and 'COMPOST PLUS EARTHWORMS' were divided in two, half being planted with a monoculture of Alnus cordata (Italian alder), the other half with Acer platanoides (Norway maple) **(FIGURE 5.4, FIGURE 5.5)**. This design resulted in a total of 40 plots, each separated by a 2 m buffer to limit edge effects. Trees planted were one-year-old root trainer seedlings with 21 planted in each plot. To relieve compaction, all plots were cultivated to 0.5 m depth by hydraulic excavator prior to planting. This involved digging out existing soils, mixing CGW into the excavated material, then loose tipping the entire substrate back into the void. Soil improver grade PAS-100 CGW screened to 0 – 25 mm particle size was incorporated into the manufactured substrate at a rate of 80 t-ha^{-1} to 500 mm depth (Moffat, 2006; Ashwood et al., 2018).

Nutrient	Nutrients kg / t (Fresh Weight)	Nutrients (kg) in 80 tonnes of CGW
Nitrogen	6.20	496
Phosphate	2.13	170
Potash	4.65	372
Magnesium	2.10	168
Sulphur	3.05	244

***TABLE 5.1** – Summary nutrient analysis of Viridor Waste Management 0-25 mm PAS 100 Composted Green Waste. This product was applied to the Ingrebourne Hill experiment using a rate of 80 t-ha^{-1} incorporated to 500 mm depth (Ashwood et al., 2018).*

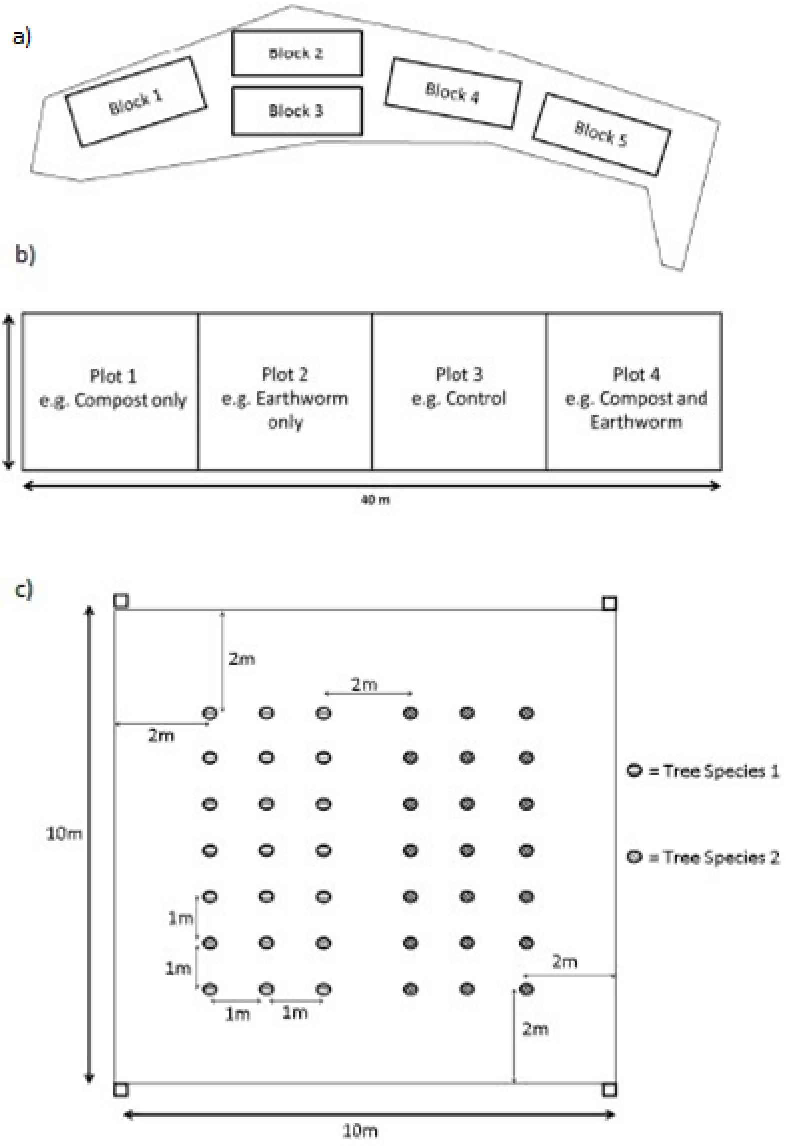


***FIGURE 5.3 –** Experimental layout at Ingrebourne Hill. **a)** shows the block layout. **b)** shows arrangement of treatments in a block. **c)** shows tree planting layout, with every plot containing a monoculture of each tree species studied (Ashwood, 2016).*

Baseline surveys of existing earthworm populations in 2012 revealed abundant *Allolobophora chlorotica* were present, alongside low numbers of *Aporrectodea longa*. Consequently, 4,200 *A. longa* earthworms were collected from surrounding land to boost the residual *A. longa* population. Every tree in an 'earthworm treatment' was inoculated with five *A. longa* earthworms. Inoculation was performed by digging a 5 cm hole at the base of each tree, adding five *A. longa* to the hole, the replacing the soil and soaking with fresh water. To prevent lateral movement of earthworms between plots, physical barriers extending 0.5 m below-ground and 0.2 m above were installed around plot perimeters (Ashwood et al., 2017).

FIGURE 5.4 *– Photo of experimental plots taken in May 2018. Norway maple are in the immediate right-hand foreground with Italian alder behind. The end of 2013 the original study found Norway maple survival was low.*

FIGURE 5.5 *– View of Central London from Ingrebourne Hill experiment. Norway maple in the right foreground. Italian alder at the left.*

5.5 Selection rationale

Considering site conditions and technical arrangements described, Ingrebourne Hill was selected based on the following. Firstly, reclamation of TCCF is ongoing, hence revisiting the experiment to obtain long-term data could inform future restoration planned at TCCF. Secondly site age was important, the experiment at Ingrebourne Hill was 5-years old when sampled, making it the youngest experiment studied in the current research. Only three years had elapsed since Ashwood et al. (2018) concluded their previous study, however this gap was judged sufficient as the site was older than those appearing in published research (Foot et al., 2003; Beesley, 2012; WRAP, 2012; Järvis et al., 2016; Somerville et al., 2018). Thirdly, Ingrebourne Hill's warm dry climate provided a unique opportunity to study CGW performance under near semi-arid conditions. This contrasted with other sites which tended to be cooler and wetter. Moving on, Ingrebourne Hill was the only site where earthworms formed part of the experimental design. This unique feature meant pre-existing soil biological data was available offering insight into earthworms' impact on tree growth and soil pedogenesis over time. Finally, CGW used at Ingrebourne was PAS100 certified which was not the case for all experimental sites, allowing performance of standardised CGW products to be observed. Together, these attributes made Ingrebourne Hill an ideal site to fulfill the aims and objectives of the current research which are reiterated as follows:

5.5.1 Aims and objectives

Explore CGW legacy impacts (defined as 5+ years) on provision of ecosystem services related to tree growth, soil formation, earthworm activity, and soil organic carbon fate.

1. Determine CGW long term impact on above-ground tree growth, biomass production, and foliar nutrient content at reclaimed sites;

2. Record physical, chemical, and biological attributes of CGW-amended soils to identify potential links between CGW application rate, CGW incorporation depth, soil formation, and carbon storage;

3. Determine how earthworm community composition, density, and biomass impacts pedogenesis in CGW-amended soils;

4. Outline CGW and earthworms synergistic impact on supporting and regulating ecosystem services

5.6 Materials and Methods

The following summarises the sampling and analysis techniques used to investigate interactions between compost application, tree growth, soil chemistry, and earthworm activity at Ingrebourne Hill. The same techniques were used at all sites to ensure data generated at different locations could be readily compared and discussed. A more detailed description of the techniques used to evaluate the study sites is provided in **Chapter 4 – GENERAL MATERIALS AND METHODS**. For information on the statistical techniques used to generate site data please refer to section **'4.5 Statistical Analysis'.**

At Ingrebourne Hill, tree growth and foliar chemistry were measured to understand CGW impacts on primary production and nutrient cycling from 24 – 60 months. Tree growth was measured in April-May 2018 using a 5 m telescopic measuring rod to record height, and callipers for recording stem diameter. Foliar collections were completed in July 2018 between mid-summer and late summer, ensuring leaf nutrient concentrations were steady. Foliage was taken from four different trees per plot, then bulked to create a plot representative sample. Foliage was oven-dried, homogenised, then sieved, to obtain smaller particles 1 – 2 mm in size. A sub-sample was taken and ground to powder using a planetary ball mill, ensuring foliage was ready for carbon and nitrogen analysis.

To evaluate the impact of compost on soil development, nutrient cycling, and carbon storage, soil cores were taken from each plot in May 2018. A Dutch auger fitted with 15 cm attachment was used to extract duplicate cores from 0 – 15 and 15 – 30 cm depth. Cores were taken from two different locations, prepared for analysis, then bulked to create a plot-representative sample for each depth. Soil organic matter content, total carbon, total nitrogen, C:N ratio, and available nitrates were all assessed. Available-N was determined by taking 10 g of fresh soil, performing a KCL-extraction, then sending soil extracts to Rothamstead Research laboratory for colorimeter analyses. For the remaining soil analyses cores were air dried, crushed in a motorised blade grinder, and sieved to <2 mm to remove stones and other large particles. Replicate cores were then bulked and sieved again to 0.5 mm, creating a fine homogenised soil. Using this material, organic matter content was determined via loss on ignition, whilst total carbon, total nitrogen, and C:N ratio for soil and foliar samples were all determined using a Thermo-Scientific CHNS Organic Elemental Analyser.

Earthworm population density and community structure was measured to explore how interactions among compost, earthworms, and tree species mediate ecosystem service provision. Sampling was conducted in April-May 2018 following the methods described by Butt and Grigoropoulou (2010). In each plot, two earthworm pits were excavated to obtain a plot representative earthworm sample. A 0.1 m^2 quadrat was placed on the soil surface, then 30 x 30 x 30 cm^3 of soil was excavated from beneath this and hand-screened for earthworms. Specimens were placed in plastic bottles and preserved in 4% formaldehyde solution prior to identification. All adult worms were identified to species level following the key of Sims and Gerard (1999). Visual identification was completed using a stereo microscope with adjustable zoom (X10 – X30 magnification). Specimens were dried on absorbent paper then weighed to determine biomass. Population densities found in each 0.1 m^2 pit were scaled up and reported as average number of species / eco-group per m^2.

5.7 Results

5.7.1 Impact of compost addition on tree species survival, growth, and foliar nutrition

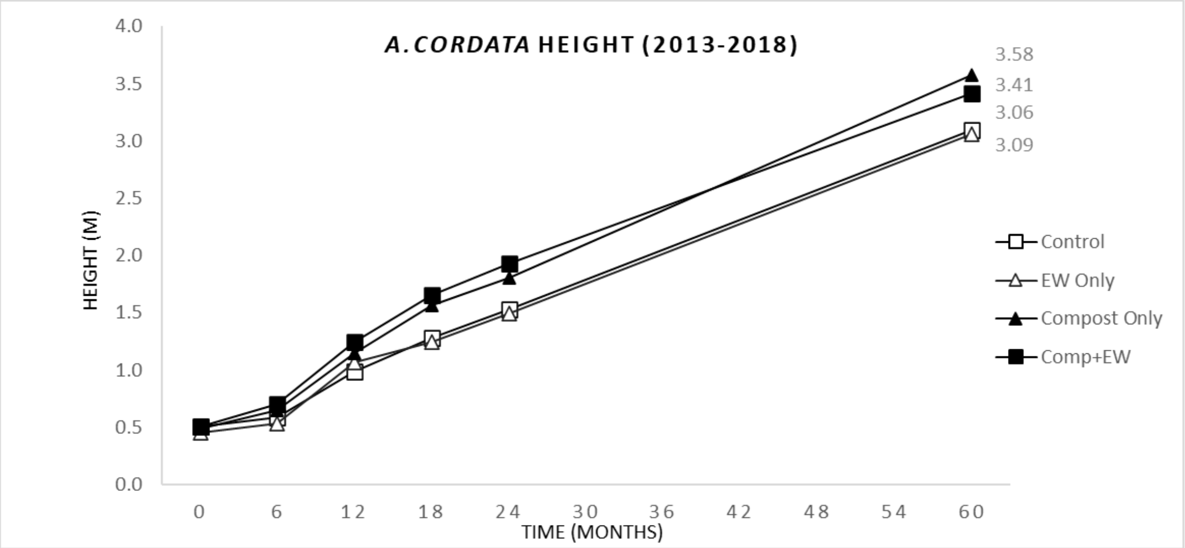
Regarding tree survival, in the experiments first year of 2013, Norway maple mortality was high. Consequently, all dead and missing trees were replaced at this time. In the 2018 survey, all surviving trees were assessed including the replacement beat ups. Losses experienced in 2013 had not been repeated, resulting in considerably higher survival at 60 months **(TABLE 5.2)**. Treatment had no statistically significant impact on survival of either tree species. However, Italian alder demonstrated high survival rates (>92%), with the highest survival under compost-only and compost plus earthworm treatments (99.3% and 98.1% respectively). Norway maple survival was lower than alder, with the compost-only treatment lowest overall (77.1%). Across all remaining treatments Norway maple survival was comparable (>84%). However, at 60 months, treatment had no statistically significant impact on survival of either species.

	Italian Alder Survival (%)			Norway Maple Survival (%)		
Treatment	12 months	24 months	*60 months	12 months	24 months	*60 months
Control	93.3 ± 1.9	90.5 ± 2.1	92.4 ± 2.4	63.8 ± 8.3	45.7 ± 11.3	89.5 ± 2.3
Earthworms	90.5 ± 1.5	88.6 ± 2.4	94.3 ± 2.3	57.1 ± 8.1	54.3 ± 10.3	84.8 ± 5.9
Compost	98.1 ± 1.2	93.3 ± 1.2	99.0 ± 1.0	51.4 ± 9.8	41.9 ± 10.4	77.1 ± 7.3
Comp + EW	99.1 ± 1.0	95.2 ± 2.6	98.1 ± 1.2	59.1 ± 6.8	45.7 ± 6.5	89.5 ± 3.8

TABLE 5.2 *– Mean tree survival (%) after 12, 24, and 60 months ± SE (n = 5). Survival at 12 and 24-months was recorded by Ashwood et al. (2018). Survival at 60-months was recorded by the present study. The 60-month observation is marked with an asterisk as replacement trees planted at 6-months in 2013 are included in these figures.*

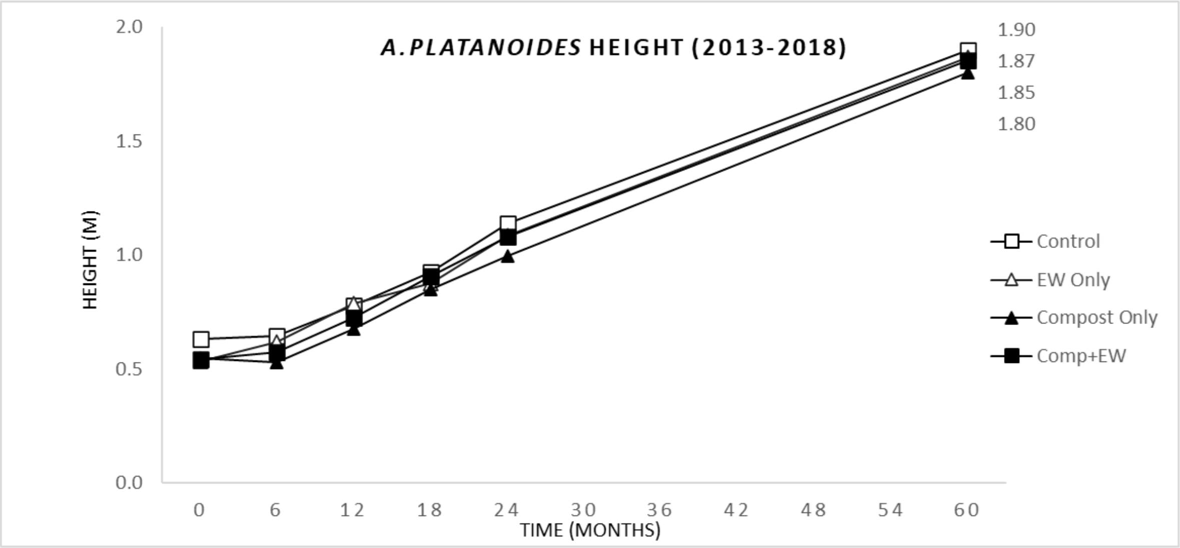
Regarding tree height, at the beginning of the experiment in 2013, no significant difference in mean height was found between treatments. However, at 60 months a treatment effect was detected, with Italian alder growing significantly taller under compost-only and compost plus earthworm treatments (Kruskal-Wallis non-parametric ANOVA, H (7) = 414.6, p <0.001). Indeed, alder trees in compost-based treatments grew 32 – 52 cm^{-1} higher than alder in non-compost treatments. Time series data shows the growth rate of Italian alder increased between 6 and 12 months across all treatments. However, in compost-based treatments the increased rate of growth continued to 18 months. No further increase was observed between 24 and 60 months in any treatment **(FIGURE 5.6).** Overall, Italian alder height outperformed N. maple across all treatments. In contrast, time series data for N. maple shows similar height growth rates across all treatments, increasing at 6 months then falling to lower levels between 24 and 60 months **(FIGURE 5.7)**. Thus, neither compost nor earthworms had a significant impact on the height of N. maple.

Regarding stem diameter at 60 months, Italian alder was significantly greater under compost-only and compost plus earthworm treatments compared with earthworms-only (Kruskal-Wallis non-parametric ANOVA, H (7) = 418.0, p <0.001). Time series data shows Italian alder stem diameter increased between 6 and 18 months across all treatments, with compost amended plots displaying particularly strong growth during from 0 – 6 months. At 60-months, Italian alder stem diameter was 5 – 8 cm wider in compost amended plots than those in earthworms-only **(FIGURE 5.8)**. Italian alder stem diameter was also greater than N. maple throughout the experiment. The experimental treatments had no impact on Norway maple stem diameter **(FIGURE 5.9)**.



Italian Alder - Growth Rate - Height (m)								
Treatment	CONTROL		EARTHWORMS		COMPOST		COMPOST + EARTHWORMS	
Time (months)	Increase (m)	G.Rate (%)	Increase (m)	G.Rate (%)	Increase (m)	G.Rate (%)	Increase (m)	G.Rate (%)
0-6	0.08	16.1	0.08	17.8	0.16	33.4	0.20	39.9
6-12	**0.40**	**68.0**	**0.53**	**99.5**	**0.50**	**77.5**	**0.54**	**76.4**
12-18	0.29	29.8	0.17	16.3	**0.41**	**36.0**	**0.42**	**33.5**
18-24	0.25	19.2	0.25	20.1	0.24	15.5	0.28	16.7
***24-60**	0.26	17.0	0.26	17.4	0.30	16.4	0.25	12.8
Average	**0.26**	**19.8**	**0.26**	**21.0**	**0.31**	**22.1**	**0.29**	**21.1**

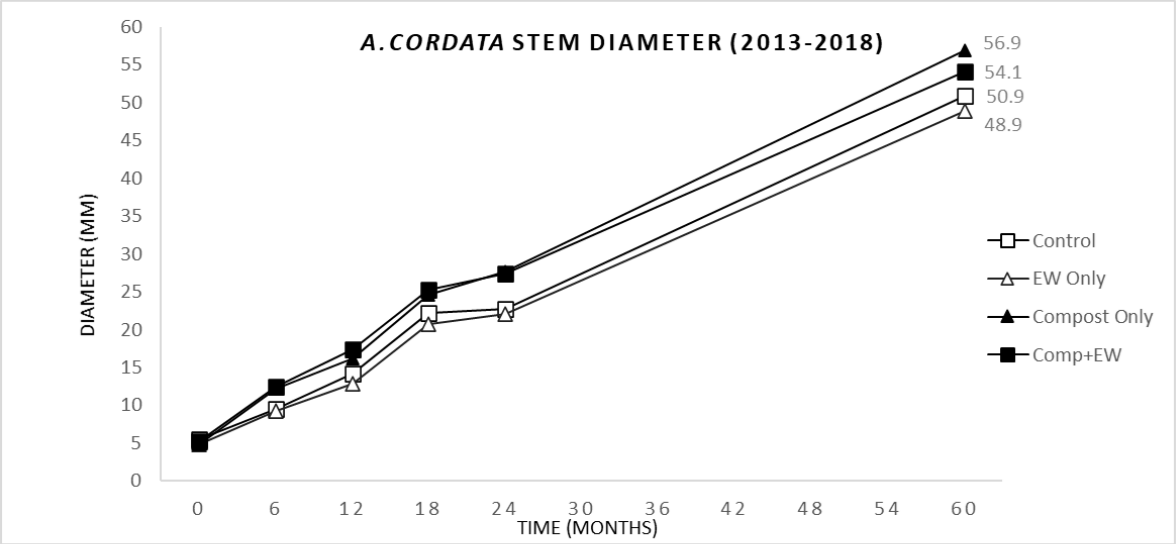
***FIGURE 5.6** – Increase in height (m) and growth rate (%) for Italian alder (A. cordata) from 0 – 60 months. Survival from 0-24 months was recorded by Ashwood et al. (2018). Survival at 60-months was recorded by the present study. GREEN indicates above average increase. Blue bars show growth rate only. An asterisk is placed next to 24 – 60 months as the 'total height increase' and 'average growth rate' are given for this extended 3-year period.*



Norway Maple - Growth Rate - Height (m)

Treatment	CONTROL		EARTHWORMS		COMPOST		COMPOST + EARTHWORMS	
Time (months)	**Increase (m)**	**G.Rate (%)**	**Increase (m)**	**G.Rate (%)**	**Increase (m)**	**G.Rate (%)**	**Increase (m)**	**G.Rate (%)**
0-6	0.01	2.4	0.08	15.7	-0.02	-3.0	0.03	6.0
6-12	0.13	20.5	**0.17**	**27.0**	**0.15**	**27.9**	**0.15**	**25.8**
12-18	**0.15**	**19.1**	0.09	11.3	**0.17**	**25.6**	**0.18**	**25.5**
18-24	**0.21**	**23.0**	**0.21**	**24.1**	0.14	17.1	**0.17**	**19.0**
***24-60**	0.13	11.1	0.13	12.0	0.13	13.5	0.13	11.9
Average	**0.13**	**11.7**	**0.13**	**13.3**	**0.13**	**12.7**	**0.13**	**13.1**

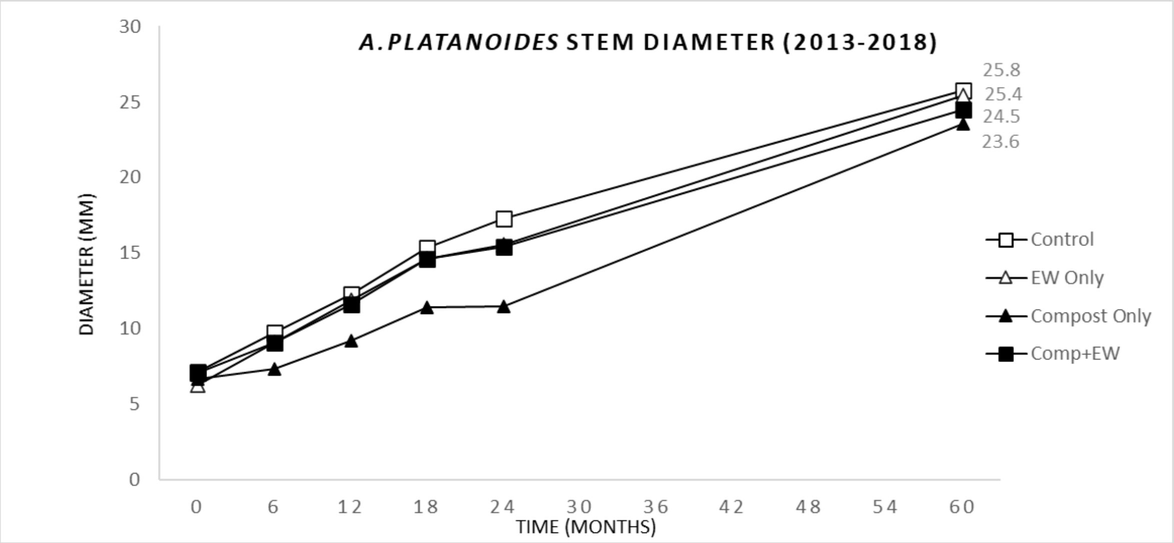
FIGURE 5.7 – Height increase (m) and growth rate (%) for Norway maple (A. platanoides) from 0 – 60 months. Survival from 0-24 months was recorded by Ashwood et al. (2018). Survival at 60-months was recorded by the present study. GREEN indicates above average increase in height and growth rate. Blue bars show growth rate only. An asterisk is placed next to 24 – 60 months as 'total height increase', and 'average growth rate' are given for this extended 3-year period.



Italian Alder - Growth Rate - Stem Diameter (mm)

Treatment	CONTROL		EARTHWORMS		COMPOST		COMPOST + EARTHWORMS	
Time (months)	**Increase (mm)**	**G.Rate (%)**	**Increase (mm)**	**G.Rate (%)**	**Increase (mm)**	**G.Rate (%)**	**Increase (mm)**	**G.Rate (%)**
0-6	**4.0**	71.3	**4.4**	91.82	**7.3**	**151.1**	**7.2**	**138.7**
6-12	4.7	49.3	3.7	39.67	4.1	33.8	5.0	40.3
12-18	**7.9**	56.0	**7.9**	61.09	**8.4**	52.0	**7.9**	45.4
18-24	0.6	2.8	1.3	6.47	3.0	12.1	2.1	8.3
***24-60**	4.7	20.6	4.5	20.27	4.9	17.7	4.4	16.2
Average	**4.5**	**24.8**	**4.4**	**26.1**	**5.2**	**28.0**	**4.9**	**26.4**

FIGURE 5.8 – Stem diameter increase (mm) and growth rate (%) for Italian alder (A. cordata) from 0 – 60 months. Survival from 0-24 months was recorded by Ashwood et al. (2018). Survival at 60-months was recorded by the present study. GREEN indicates above average increase in height and growth rate. Blue bars show growth rate only. An asterisk is placed next to 24 – 60 months as the 'total diameter increase', and 'average diameter growth rate' are given for this extended 3-year period.



Norway Maple - Growth Rate - Stem Diameter (mm)								
Treatment	CONTROL		EARTHWORMS		COMPOST		COMPOST + EARTHWORMS	
Time (months)	Increase (mm)	G.Rate (%)	Increase (mm)	G.Rate (%)	Increase (mm)	G.Rate (%)	Increase (mm)	G.Rate (%)
0-6	**2.6**	**36.9**	**2.8**	**44.45**	0.6	9.7	**2.1**	**29.1**
6-12	**2.5**	**25.8**	**2.8**	**30.64**	**1.9**	**26.0**	**2.5**	**27.8**
12-18	**3.1**	**25.1**	**2.7**	**23.03**	**2.2**	**23.6**	**3.0**	**25.8**
18-24	1.9	12.6	1.0	6.58	0.1	0.4	0.8	5.3
***24-60**	1.4	8.2	1.6	10.59	2.0	17.6	1.5	9.9
Average	**1.9**	**13.7**	**1.9**	**15.0**	**1.7**	**13.4**	**1.7**	**13.3**

FIGURE 5.9 *– Stem diameter increase (mm) and growth rate (%) for Norway maple (A. platanoides) from 0 – 60 months. Survival from 0-24 months was recorded by Ashwood et al. (2018). Survival at 60-months was recorded by the present study. GREEN indicates above average increase in height and growth rate. Blue bars show growth rate only. An asterisk is placed next to 24 – 60 months as the 'total diameter increase', and 'average diameter growth rate' are given for this extended 3-year period.*

Regarding foliar C:N ratio, no significant treatment effect was found at 60 months. In contrast, a significant tree species effect was identified, with Italian alder foliar C:N significantly lower than Norway maple (Kruskal-Wallis non-parametric ANOVA, H (7) = 32.43, p <0.001). Further, according to values published in Foot and Moffat (2008), Norway maple (%) foliar-N was borderline deficient with compost and deficient without compost **(FIGURE 5.10)**. This shows alder did not have a strong effect on soil C:N or foliar nitrogen in Norway maple trees. Surprisingly, despite Norway maples poor performance soil C:N ratio under Norway maple was significantly lower than Italian alder control and earthworms-only (Kruskal-Wallis non-parametric ANOVA, H (7) = 27.83, p <0.001) **(FIGURE 5.11)**. This suggests 'tree species effects' were not responsible for lowering soil C:N at Ingrebourne Hill.

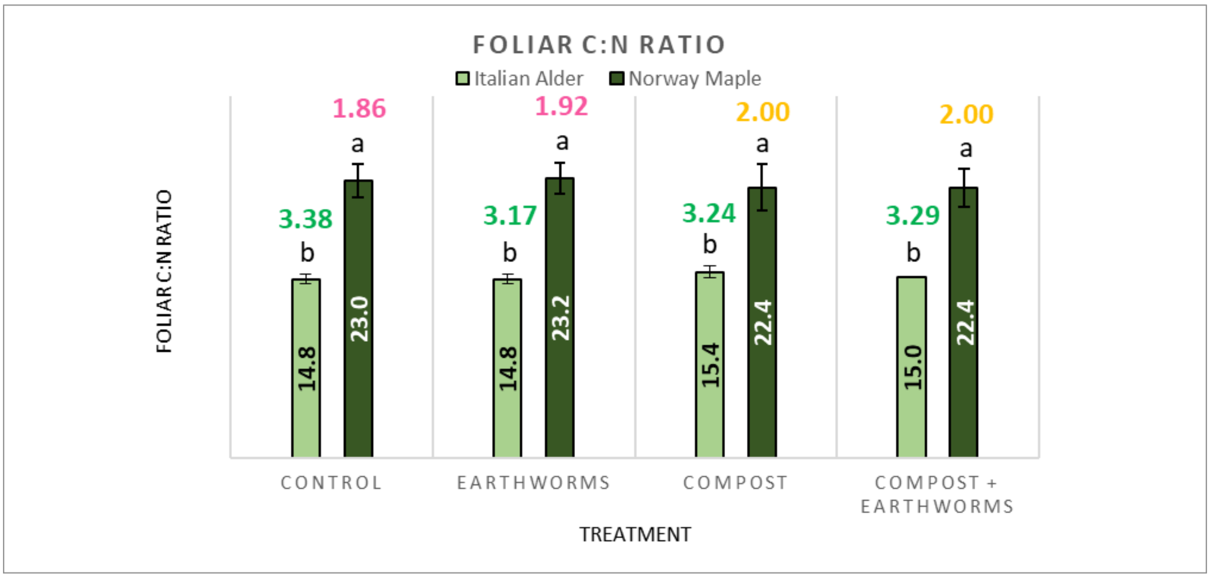


FIGURE 5.10 *– Foliar C:N ratio of Italian alder versus Norway maple. Kruskal-Wallis non-parametric test with Dunn's post hoc bonferroni correction showed Italian alder foliar C:N was significantly lower than Norway maple. Values above bars represent (%) foliar-N.* ***GREEN =*** *above optimum (>2.3 % for Maple; >2.8% for Alder).* ***YELLOW =*** *borderline.* ***PINK =*** *below optimum (<2.0 % for Maple; <2.5% for ALD). Norway maple (%) foliar-N was borderline deficient with compost, and deficient without.*

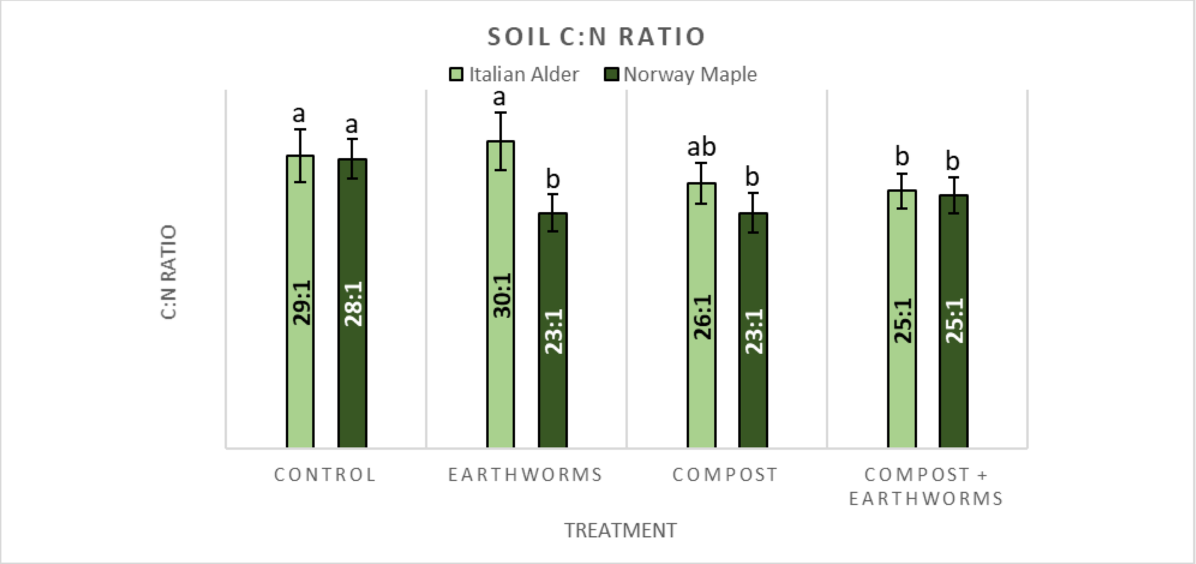
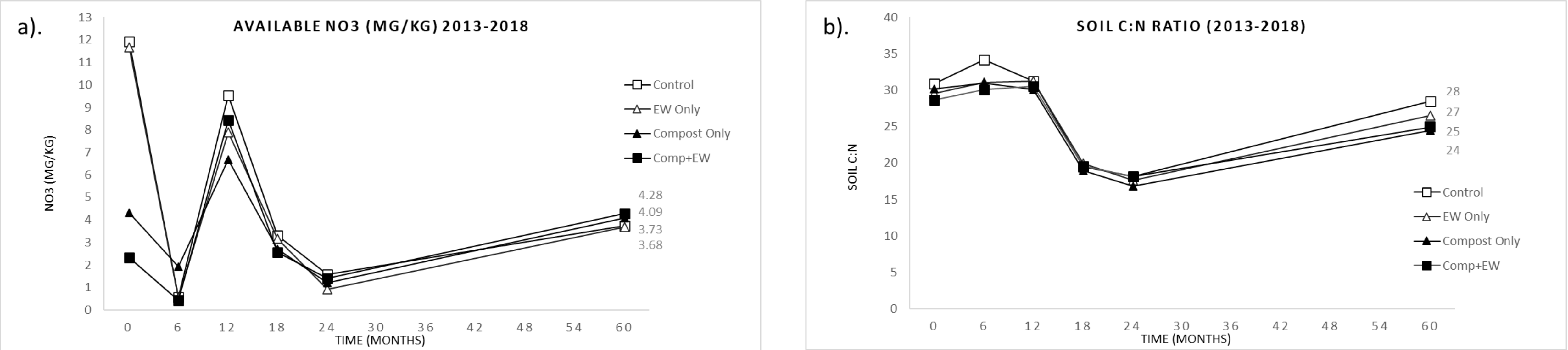


FIGURE 5.11 *– Soil C:N ratio under Italian alder and Norway maple. A Kruskal-Wallis non-parametric test followed by Dunn's post hoc with bonferroni correction showed soil C:N was significantly lower in most compost-based treatments and N. maple earthworms-only plots.*

5.7.2 Impact of compost addition on soil nitrogen, organic matter, and carbon storage

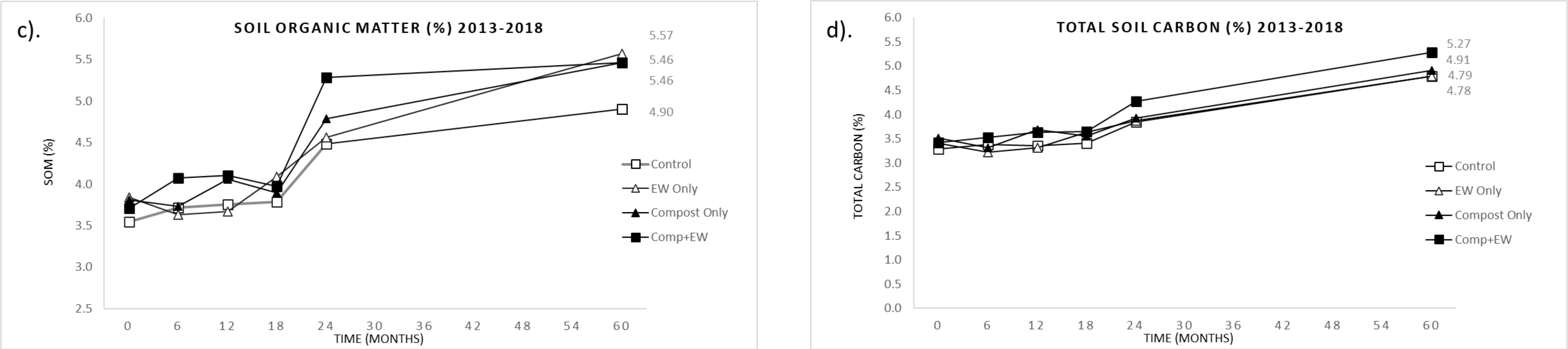
At 60 months, significant differences in soil chemistry were found between the 0-15 and 15-30 cm sampling depths. Across the entire experiment, (%) soil organic matter and total nitrogen were significantly higher, whilst soil C:N ratio was significantly lower at 0-15 cm sampling depth (<u>SOM</u>: one way ANOVA, F (1,78) = 8.284, $p<0.005$); (<u>Total-N</u>: one way ANOVA, F (1,78) = 7.135, $p<0.009$); (<u>Soil C:N ratio</u>: one way ANOVA, F (1,78) = 4.430, $p<0.04$). However, despite these significant differences, experimental treatments had no impact on the depth distribution of soil organic matter, soil carbon, or soil nitrogen. Thus, differences between sampling depths are likely due to topsoil development from a range of plant inputs.

FIGURE 5.12 and **FIGURE 5.13** show how soil chemical variables changed over time across the entire experiment regardless of tree species. **TABLE 5.3** summarises when changes observed were statistically significant. The rate of change observed for each soil chemical variable was greatest during the following time periods and occurred following this sequence: a). available nitrogen increasing at 6 – 12 months; b). C:N ratio decreasing at 12 – 18 months; c). (%) soil organic matter, (%) total carbon, and (%) total nitrogen increasing at 18 – 24 months. In **FIGURE 5.12 (a)**, available NO_3^- and NH_4^+ peak significantly at 12 months, decline at 24 months, then increase gradually on approach to the 60-month observation. **FIGURE 5.12 (b),** shows soil C:N ratios decreasing significantly at 18 months, falling to their lowest recorded values at 24 months, then returning to higher ratios of 24:1 and above at 60 months. Moving on, (%) soil organic matter, (%) total carbon, and (%) total nitrogen experienced their largest growth at 24 months as shown in **FIGURE 5.12 (c), (d);** and **FIGURE 5.13** respectively. However either side of this time point they remain relatively stable. In summary, by examining changes in soil chemistry over time across the entire experiment without separating data according to tree species, all treatments followed a similar pattern of change over time. However, the compost-only and compost-plus earthworm treatments experience a greater intensity of change between 12 and 24 months.



a).	NO3 (mg/kg)			NH4 (mg/kg)		
Time (Months)	**Quantity**	**Change +/-**	**G.Rate**	**Quantity**	**Change +/-**	**G.Rate**
0	9.8			3.7		
0-6	2.7	-7.1	-72.9	3.4	-0.3	-8.2
6-12	**15.5**	**12.8**	**482.9**	**9.0**	**5.6**	**163.1**
12-18	4.3	-11.2	-72.5	4.7	-4.3	-48.1
18-24	1.3	-2.9	-69.3	3.2	-1.5	-31.5
***24-60**	3.9	2.6	33.7	11.3	8.1	42.0
Average		**-0.6**	**-8.7**		**0.8**	**11.7**

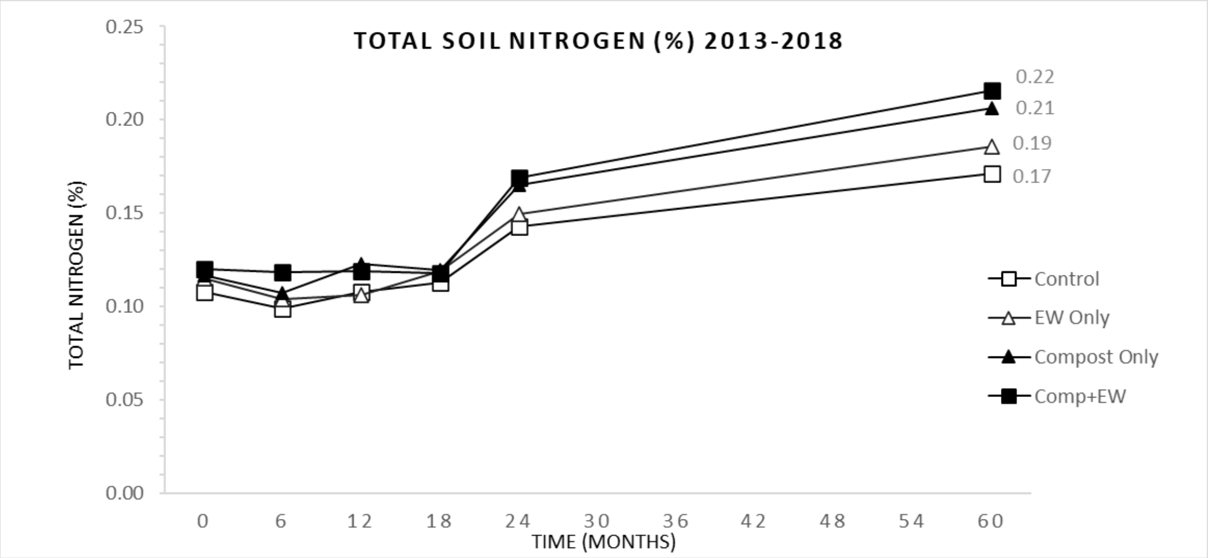
b).	C:N Ratio		
Time (Months)	**Quantity**	**Change +/-**	**G.Rate**
0	29.8		
0-6	31.6	1.8	5.9
6-12	30.7	-0.8	-2.6
12-18	**19.5**	**-11.3**	**-36.7**
18-24	17.7	-1.8	-9.1
***24-60**	26.1	8.4	7.9
Average		**-0.4**	**-1.32**



c).	SOM (%)		
Time (Months)	**Quantity**	**Change +/-**	**G.Rate**
0	3.72		
0-6	3.79	0.06	1.7
6-12	3.90	0.11	2.9
12-18	3.94	0.04	1.0
18-24	**4.78**	**0.84**	**21.4**
***24-60**	5.38	0.60	2.1
Average		**0.17**	**3.75**

d).	TOTAL C (%)		
Time (Months)	**Quantity**	**Change +/-**	**G.Rate**
0	3.40		
0-6	3.36	-0.04	-1.3
6-12	3.50	0.13	4.0
12-18	3.56	0.06	1.8
18-24	**3.98**	**0.42**	**11.7**
***24-60**	4.94	0.96	4.0
Average		**0.15**	**3.79**

***FIGURE 5.12** – Line graphs illustrating changes in soil chemical variables over time. Survival from 0-24 months was recorded by Ashwood et al. (2018). Survival at 60-months was recorded by the present study. Put together, these figures indicate an overall treatment effect may be apparent. These effects emerge in the following sequence a). available nitrogen increases at 6 – 12 months; b). C:N ratio decrease at 12 – 18 months; c). (%) soil organic matter and (%) total carbon increase at 18 – 24 months. At 60 months, the differences that emerged earlier on persist, as indicated by final values for each treatment which show differences between compost-based treatments and controls.*



TOTAL N (%)			
Time (Months)	**Quantity**	**Change +/-**	**G.Rate**
0	0.115		
0-6	0.107	-0.008	-6.79
6-12	0.114	0.007	6.49
12-18	0.117	0.003	2.89
18-24	**0.157**	**0.039**	**33.53**
***24-60**	0.195	0.038	4.05
Average		**0.008**	**5.42**

__FIGURE 5.13__ – Line graph and table illustrating changes in soil Total-N (%) over time. Survival from 0-24 months was recorded by Ashwood et al. (2018). Survival at 60-months was recorded by the present study. A treatment effect emerges at 24 months and remains present at 60 months. The effect arises alongside concurrent changes in SOM (%) and Total-C (%). Final values at 60 months indicate total-N is higher in compost-based treatments compared with control and earthworm-only.

Repeated Measures ANOVA - Treatment Effect on Soil Chemistry Over Time										
Variable	**Treatment**	**Test**	**F - statistic**	**df**	**p-value**	**Effect size - ω2**	**post-hoc test (bonferroni correction)**			**Overall change (+/-) 0-60 months**
							18 - 24 months	**24 - 60 months**	**18 - 60 months**	
SOM	Control	RMANOVA	27.57	(1.2, 4.8)	0.003	0.674	X	X	↑	+
	Earthworms	RMANOVA	17.17	(1.2, 4.8)	0.009	0.701		↑		
	Compost	RMANOVA	63.34	(2, 8)	0.001	0.898	↑			
	Comp + EW	RMANOVA	53.80	(2, 8)	0.001	0.648		X		
Total Carbon (%)	Control	RMANOVA	69.52	(2, 8)	0.001	0.745	↑	↑	↑	+
	Earthworms	RMANOVA	17.94	(2, 8)	0.001	0.544	X			
	Compost	RMANOVA	29.78	(2, 8)	0.001	0.781				
	Comp + EW	RMANOVA	55.84	(1.4, 5.6)	0.001	0.696	↑			
Total Nitrogen (%)	Control	RMANOVA	24.83	(2, 8)	0.001	0.727	↑	X	↑	+
	Earthworms	RMANOVA	23.13	(1.1, 4.3)	0.007	0.740	X	↑		
	Compost	RMANOVA	37.36	(1.1, 4.5)	0.002	0.806	↑	X		
	Comp + EW	RMANOVA	45.44	(1.3, 5.0)	0.001	0.763		↑		
C:N Ratio	Control	F - RMANOVA	8.40	2	0.015	-	X	↑	↑	-
	Earthworms	F - RMANOVA	10.00	2	0.007	-	↓			
	Compost	F - RMANOVA	10.00	2	0.007	-				
	Comp + EW	RMANOVA	30.30	(2, 8)	0.001	0.766	X			
NO3 (mg/kg)	Control	RMANOVA	3.01	(2, 8)	0.134	-	X	X	X	-
	Earthworms	RMANOVA	12.41	(2, 8)	0.004	0.614		↑		
	Compost	F - RMANOVA	7.60	2	0.02	-	↑			
	Comp + EW	RMANOVA	11.24	(2, 8)	0.005	0.608				
NH4 (mg/kg)	Control	RMANOVA	10.15	(2, 8)	0.006	0.477	X	↑	X	+
	Earthworms	F - RMANOVA	7.60	2	0.022	-			↑	
	Compost	RMANOVA	7.18	(1.2, 5.0)	0.042	0.464			X	
	Comp + EW	RMANOVA	69.59	(1.1, 4.5)	0.001	0.915	↓		↑	

__TABLE 5.3__ – Treatment effect on soil chemical variables over time. GREEN arrows & shading indicate a statistically significant increase over a given timeframe. RED arrows & shading indicate a significant decrease. 'X' indicates no significant change. YELLOW shading indicates the overall non statistical trend (+) increase (-) decrease from 0 – 60 months.

Analysing data at the tree-species level, changes observed over time result in a range of tree-species / treatment interactions at 60-months. Under both tree species, the compost-only and compost-plus earthworm treatments contained more (%) soil organic matter, (%) total nitrogen, available NO_3^- (mg kg^{-1}), and lower soil C:N ratio than controls. In certain cases the difference was significant: **FIGURE 5.14** – soil organic matter: Kruskal-Wallis non-parametric ANOVA, H (7) = 26.98, p <0.001; **FIGURE 5.16** – total nitrogen: Kruskal-Wallis non-parametric ANOVA, H (7) = 18.97, p <0.008; **FIGURE 5.17** – available NO_3^- : Kruskal-Wallis non-parametric ANOVA, H (7) = 15.55, p <0.03; **FIGURE 5.11** – soil C:N ratio: Kruskal-Wallis non-parametric ANOVA, H (7) = 27.83, p <0.001). The compost plus earthworm treatments also contained the highest quantities of total carbon (%) for both tree species, however in this instance the difference between treatments was not statistically significant **(FIGURE 5.15)**. The only exception to a pattern of increased SOM, nitrogen, and carbon associated with compost-based treatments was Norway maple earthworms-only. This particular treatment had soil C:N ratio, and (%) total soil nitrogen similar to the compost-based treatments which requires explanation as no compost was added **(FIGURE 5.11, FIGURE 5.16)**.

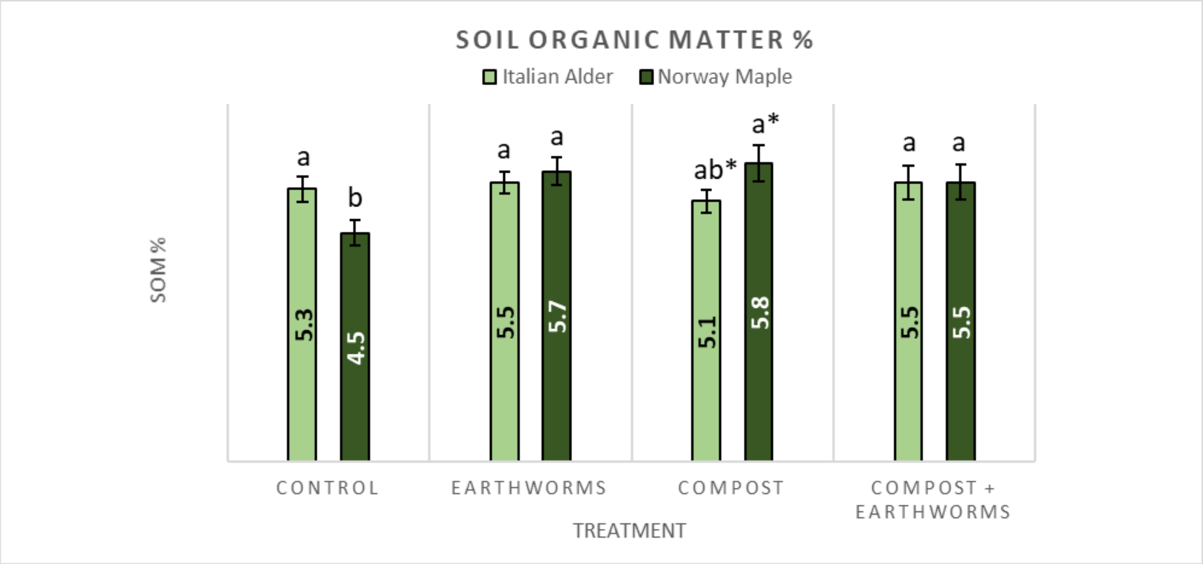


FIGURE 5.14 – *Impact of tree species and treatment on soil organic matter content. A difference in letters denotes a significant difference. Kruskal Wallis non-parametric ANOVA [H (7) = 26.98, p <0.001], followed by Dunn's post-hoc test with bonferroni correction (p < 0.05) revealed all treatments except alder-compost were significantly higher than maple-control. Both compost-only treatments are also marked by an asterisk because N. maple-compost, the highest of all, was the only treatment to contain significantly more SOM than alder-compost, the second lowest of all.*

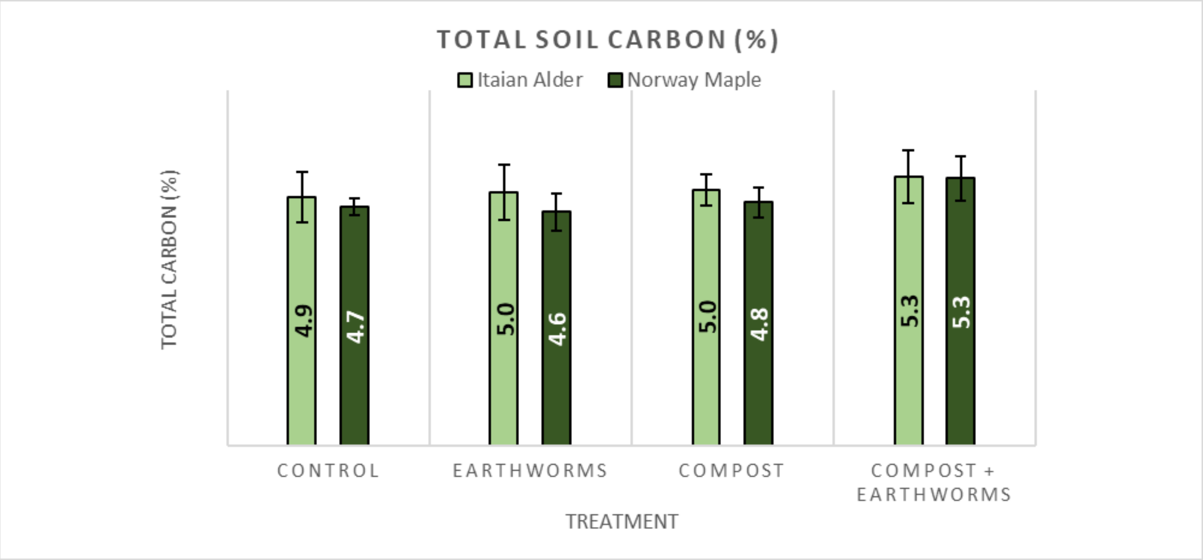


FIGURE 5.15 – *Impact of tree species and treatment on total soil carbon (%). No significant difference was found between treatments. Kruskal Wallis non-parametric ANOVA [H (7) = 9.606, p <0.212.*

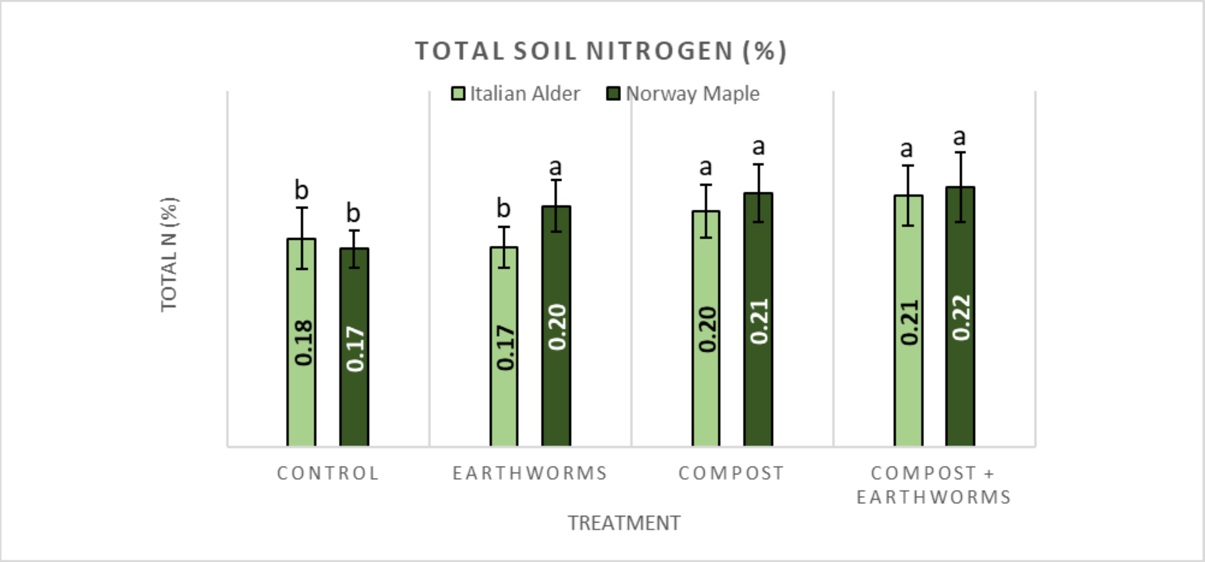


FIGURE 5.16 – *Impact of tree species and treatment on total soil nitrogen (%). Different letters denote significant differences between treatments. Kruskal Wallis non-parametric ANOVA [H (7) = 18.97, p <0.008], followed by Dunn's post-hoc test (p <0.05) revealed all compost-based treatments plus Norway maple earthworms-only were sig. higher than controls. Further, total soil nitrogen was slightly higher under Norway maple in all treatments.*

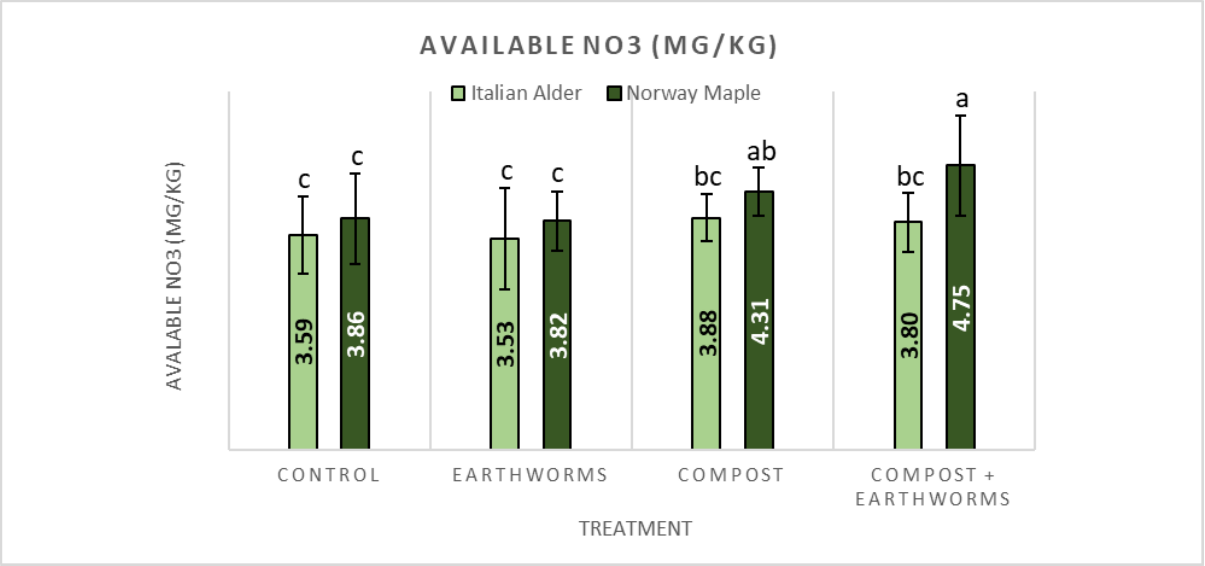


FIGURE 5.17 – *Impact of tree species and treatment on available NO3 (mg/kg). Different letters denote significant differences. Kruskal Wallis non-parametric ANOVA [H (7) = 15.55, p <0.03], followed by Dunn's post-hoc test (p <0.05) revealed compost-based treatments under Norway maple contained sig. more available NO3 than controls.*

5.7.3 Impact of tree species and compost addition on earthworm populations

At 60 months, seven earthworm species were found within the experimental site. Most were present at low mean levels except *Allolobophora chlorotica* and *Aporrectodea longa,* the latter being inoculated in 2013. Average number of earthworms per species were: *A. chlorotica* (48 m^{-2}), *A. longa* (9 m^{-2}), *Lumbricus castaneus* (4 m^{-2}), *Lumbricus terrestris* (3 m^{-2}), *Aporrectodea caliginosa* (2 m^{-2}), *Lumbricus festivus* (1.43 m^{-2}), and *Lumbricus rubellus* (0.29 m^{-2}). **FIGURE 5.18** separates earthworm densities according to tree species. High numbers of the endogeic *A. chlorotica* were found under both Italian alder and Norway maple. As a result, significantly more endogeic earthworms were recorded under both tree species compared with anecic and epigeic populations (Kruskal-Wallis non-parametric ANOVA, H (5) = 88.73, p <0.001). However, although endogeic densities were higher under Norway maple (66 m^{-2}) compared with Italian alder (33 m^{-2}), Dunn's post-hoc test with bonferroni correction revealed the difference was not significant (p <1.000) because A. chlorotica variance between sampling locations was high. For example, in block one alone, an estimated 1060 and 1770 m^{-2} were found in N. maple control and compost-only plots, disproportionately increasing average endogeic density. This may be explained by herding behaviour documented in other species, rather than any tree species or treatment effects.

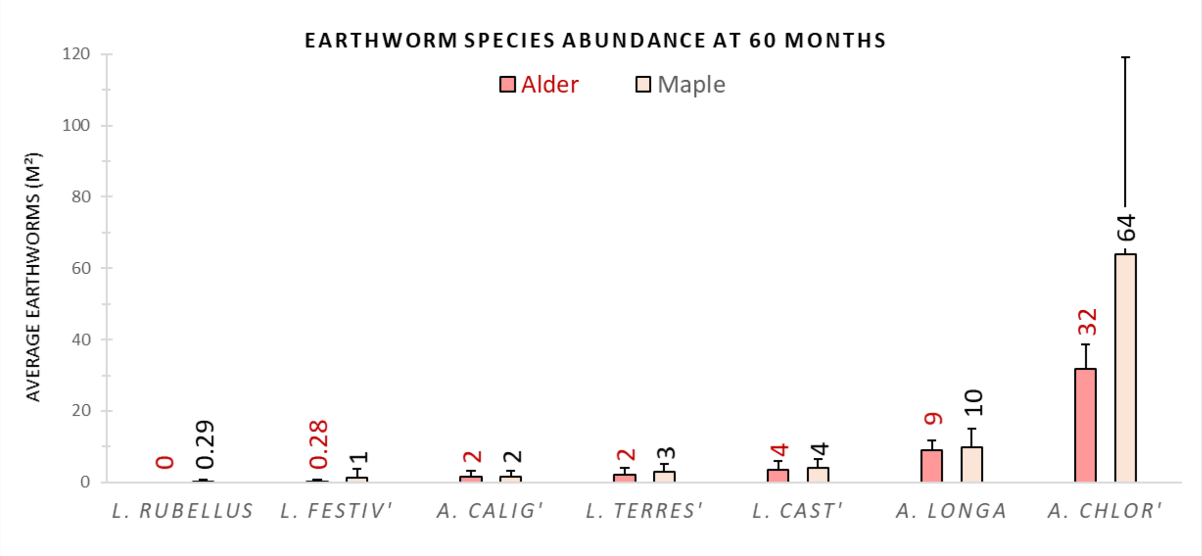
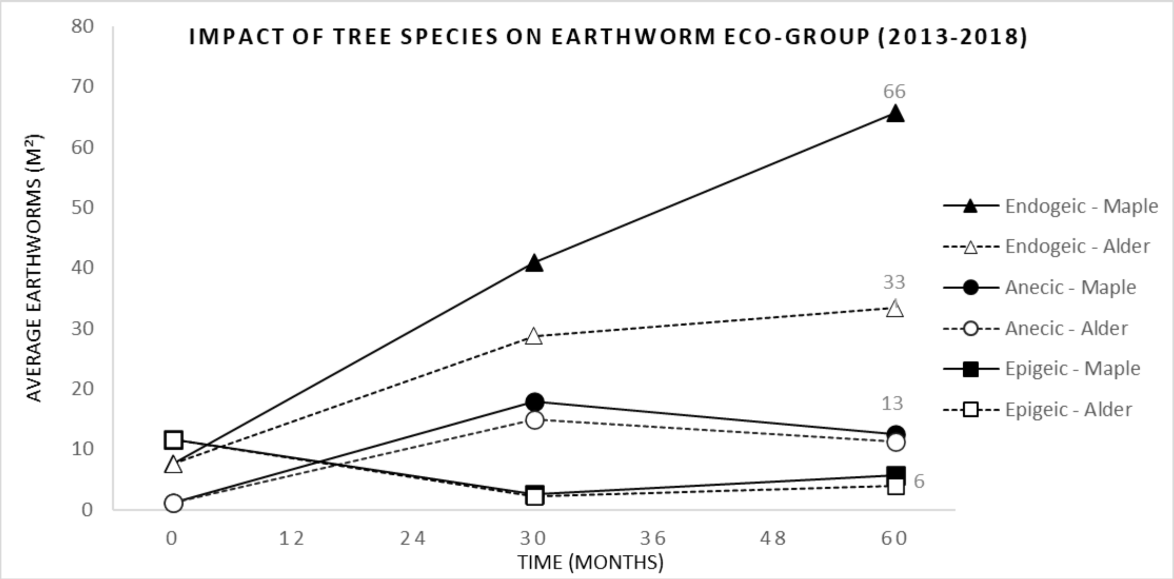


FIGURE 5.18 *– Earthworm abundance (m^{-2}) under different tree species at Ingrebourne Hill at 60-months. Sampling was conducted in May 2018. Similar earthworm abundance for both tree species indicates tree species had limited impact on earthworm populations. Further, differences apparent between individual earthworm species were not significant.*

FIGURE 5.19 illustrates changes in earthworm abundance over time (0 – 60 months) and indicates when these changes were significant via a table of repeated measures ANOVAs. In general, similar trends in earthworm density were observed under both tree species. Overall, anecic and endogeic abundance increased, whilst epigeic abundance decreased from 0 – 60 months. However, repeated measures ANOVA results attached to **FIGURE 5.19** show earthworm density changed significantly from 0 – 30 months but not from 30 – 60 months. This suggests a greater intensity of change early on, followed by slower change or gradual decline as time progressed.

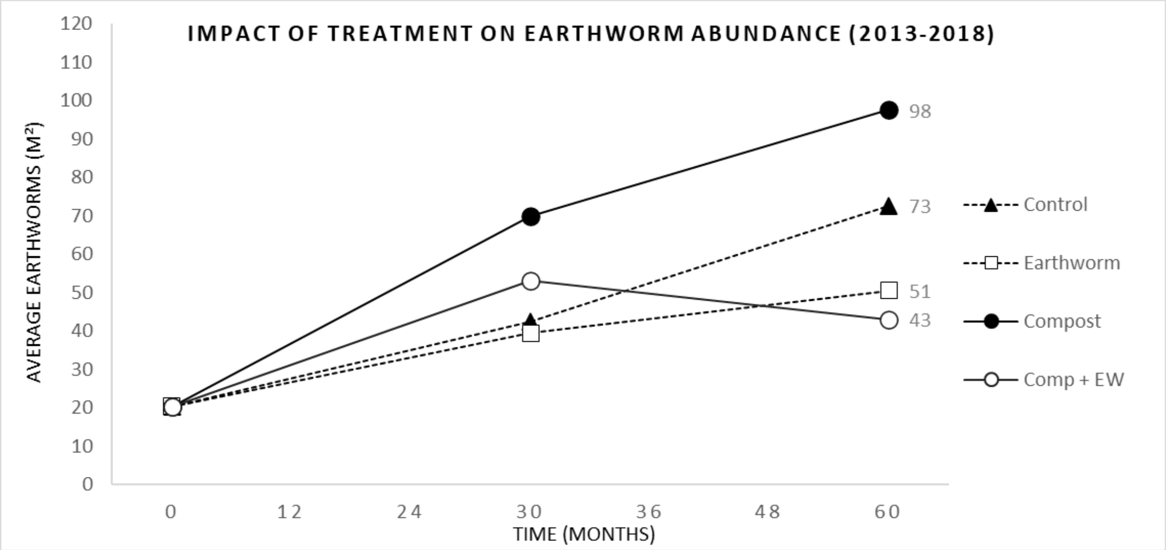


IMPACT OF TREE SPECIES ON EARTHWORM ECOLOGICAL GROUP (m²)

Treatment	Test	F-statistic/ Chi-squared	df	p-value	Effect size	post-hoc test (w/bonferroni)			Overall change (+/-) 0-60 months
						0 - 30 months	30 - 60 months	0 - 60 months	
Endogeic - Maple	F - RMANOVA	22.27	2	0.001	0.653	↑	X	↑	+
Endogeic - Alder	RMANOVA	9.25	2	0.001	0.078				
Anecic - Maple	F - RMANOVA	25.18	2	0.001	0.490	↑	X	↑	+
Anecic - Alder	RMANOVA	12.62	2	0.001	0.114				
Epigeic - Maple	F - RMANOVA	8.64	2	0.01	0.493	↓	X	↓	-
Epigeic - Alder	F - RMANOVA	12.34	2	0.002	0.375				

***FIGURE 5.19** – Changes in earthworm abundance (m^{-2}) over time. For each earthworm ecological group, a similar pattern of change is observed under both tree species from 0 – 30 and 30 – 60 months. From 0 – 30 months the changes (+/-) observed are significant. From 30 – 60 months the rate of change (+/-) slows or switches direction. Post-hoc tests with bonferroni correction revealed changes during 30 – 60 months are not significant.*

FIGURE 5.20 shows the effect of treatment on earthworm abundance (No. m^{-2}) over time. Again, changes in earthworm abundance were significant from 0 – 30 months but not from 30 – 60 months. Furthermore, at 60 months there was no significant difference in earthworm abundance between treatments (one-way ANOVA, F (3,67) = 0.616, p 0.607). Earthworm density appears higher in non-inoculated treatments (i.e., control and compost-only). However, an examination of median and mode earthworm density reveals the difference among treatments is not as great as the mean suggests. For example, median earthworm densities per treatment were control = (35 m^{-2}); earthworms-only = (50 m^{-2}); compost = (40 m^{-2}); comp+EW = (30 m^{-2}). Mode earthworm densities were control = (20 m^{-2}); earthworms-only = (50 m^{-2}); compost = (40 m^{-2}); comp+EW = (20 m^{-2}). Results from Ingrebourne Hill will now be presented and summarised using PCA (principal component analysis).



IMPACT OF TREATMENT ON TOTAL EARTHWORM DENSITY (m^2)

Treatment	Test	Chi-squared	df	p-value	Kendall's W	post-hoc test (w/bonferroni)			Overall change (+/-) 0-60 months
						0 - 30 months	30 - 60 months	0 - 60 months	
Control	F - RMANOVA	12.30	2	0.002	0.462	↑	X	↑	+
Earthworms	F - RMANOVA	16.59	2	0.001	0.351	X	X	↑	
Compost	F - RMANOVA	10.03	2	0.007	0.369	↑	X	↑	
Comp + EW	F - RMANOVA	11.32	2	0.003	0.354	↑	X	↑	

***FIGURE 5.20** – Impact of treatment on earthworm abundance (m^{-2}) over time. Changes were significant from 0 – 30 months except for the earthworms-only treatment. From 30 – 60 months change over time was no longer significant. At the end of the 60-month observation period, the difference between treatments was not significant (one-way ANOVA, F (3,67) = 0.616, p 0.607).*

5.7.4 Overview of results

Principal component analysis (PCA) was used to analyse site data from 2018. This provided an overview of tree performance and CGW impacts on soil quality and earthworm populations. Graphical PCA outputs were assessed by examining distances between data points, and angles between vectors. The PCA map depicted in **FIGURE 5.21** shows a clustering of data points and grouping of variables according to tree species and tree growth. ‘Height’, ‘DBH’, ‘Foliar C’ and ‘Foliar N’ group under Italian alder, whilst ‘Foliar C:N’ ratio and soil ‘NO_3^-’ group under Norway maple. The two groups of variables are positioned at opposite ends of a 180° angle, meaning they have a negative ‘opposing’ correlation. This means Italian alder grew taller, had wider stems, and its foliage contained more carbon and nitrogen than Norway maple. Conversely, Norway maple was smaller, with a higher foliar C:N ratio, but its soils contained more available NO_3^-. Most soil chemical and earthworm related variables are positioned between the two tree-species, indicating the tree species effects observed at 60-months in PCA **FIGURE 5.21** were unrelated to soil chemistry and earthworm density or biomass.

The PCA map depicted in **FIGURE 5.22** illustrates the effect of treatment on soil chemistry and earthworm populations. Data points overlap significantly, hence no strong treatment effect is apparent. Vectors / variables are located in two main groups positioned at a 90° angle from one another, meaning no significant correlation exists between the two groups. However, both sets of variables do associate weakly with the compost-based treatments. For example, ‘% SOM’, ‘Soil C’, ‘Soil N’, and epigeic earthworms align with the compost plus earthworm treatment. Whereas available NH_4^+, anecic, and endogeic earthworms associate with the compost-only treatment. Taking both PCA maps together, measurable differences exist between tree species, and between compost versus non-compost treatments. However, earthworms have only a weak attraction to compost-based treatments, and do not correlate with the tree species effects observed. Furthermore, given

Norway Maple’s poor performance, the tree-species effect may not be responsible for increased NO_3^- and SOM observed in Norway maple plots.

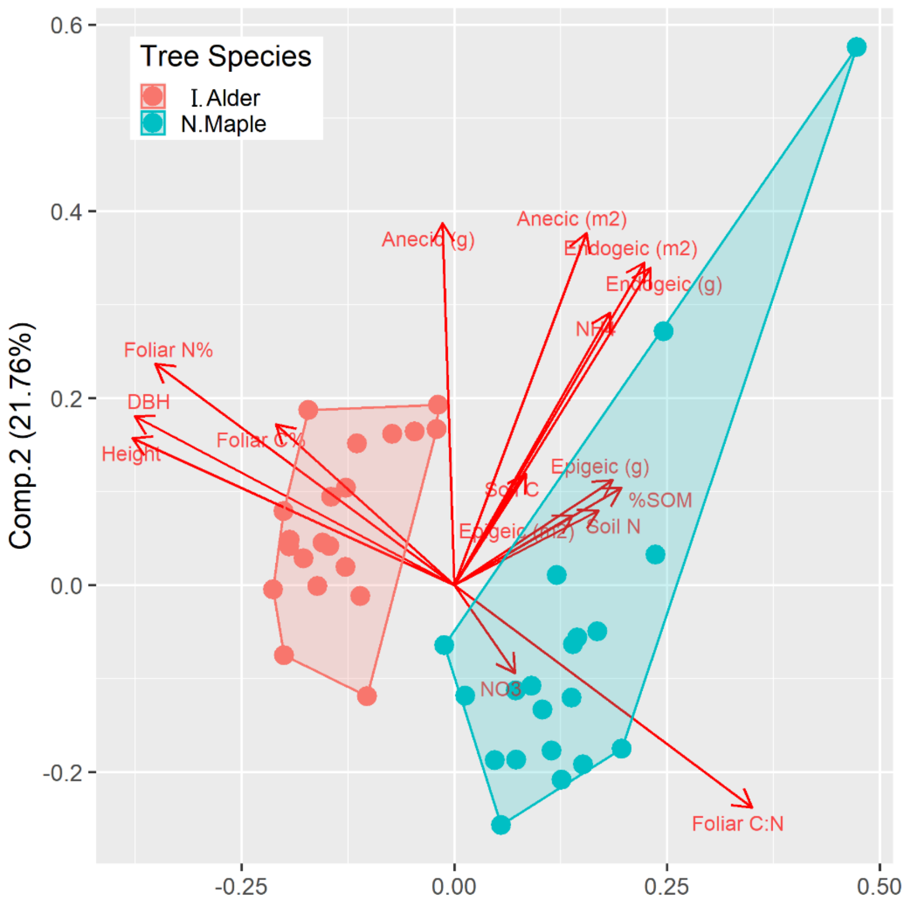


FIGURE 5.21 – PCA illustrating relationships between tree growth, soil chemistry, and earthworms at Ingrebourne Hill. Data separates and clusters according to tree species. This occurs along the primary axis of PCA 1 versus PCA 2 and accounts for around 50% of the variation observed along this axis. Variables positioned between the two main clusters of data points [i.e. 'Anecic (g)', 'Endogeic (g)', NH_4^+ etc] are not correlated with the clustering effect

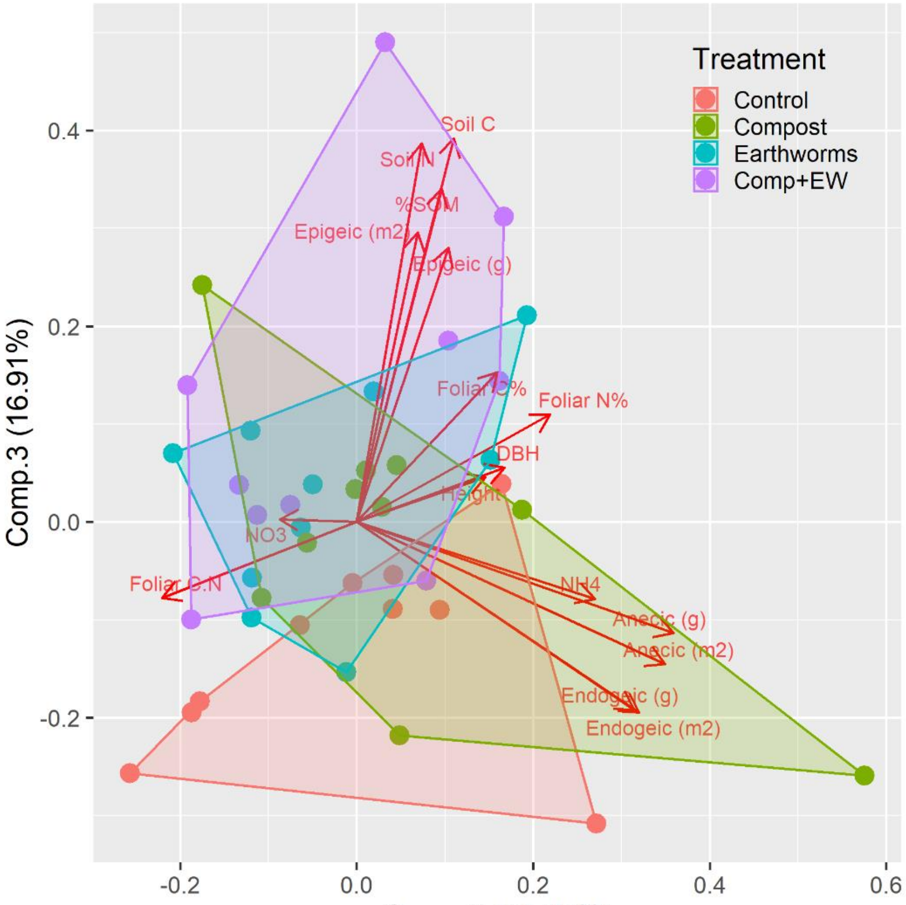


FIGURE 5.22 – PCA illustrating treatment effect on soil chemistry and earthworm populations at Ingrebourne Hill. Data shows a weak separation according to treatment. The two groups of variables associate with compost as indicated by the grouping of variables/vectors in the compost-based treatments. This effect occurs along the secondary axis of PCA 2 versus PCA 3 and accounts for around 37% of the variation observed along this axis.

5.8 Discussion

Composted green waste and earthworms were expected to have a beneficial impact on tree growth, soil carbon, and soil nitrogen. However, the timeframe over which this takes place was unclear. For example, exactly when might CGW begin to stimulate tree growth, organic matter accumulation, and earthworm activity? How long would these effects last? Would the impact of compost increase or diminish over time? Data from Ingrebourne Hill suggests shortly after incorporation, a single application of composted green waste elevates soil resource availability, increasing plant growth, soil carbon, soil nitrogen, and earthworm activity. These beneficial effects unfold over a two-year period, impacting ecological processes between 6 and 24 months. The effect of compost at Ingrebourne Hill appears subtle and short lived, nevertheless remnants of its early impact on plant growth and soil chemistry can still be detected at 60 months.

5.8.1 Impact of compost addition on tree species survival, growth, and foliar nutrition

A previous study by Ashwood et al. (2018) covered site development from 0-30 months. The current study measured tree growth at 60-months and confirms compost increased tree growth during 0-24-months alone. This means the findings of the current study are similar to Ashwood et al. (2018). For example, the difference in Italian alder height and stem diameter found between compost and non-compost treatments was similar at 24 and 60-months. Italian alder grew vigorously during the first 18-months; with no further compost-induced increase observed beyond this point. This shows compost has an acute effect on tree growth, impacting primary production shortly after compost is applied. Yet because this effect remains detectable at 60-months, we can conclude a single incorporation of compost during soil manufacturing makes a lasting difference to this point. Other studies have observed composts beneficial impact on tree growth, however monitoring has rarely if ever been conducted beyond 48-months (Foot et al., 2003; WRAP, 2011a; WRAP, 2012; Palmer and Davies, 2014). Previous studies also measured tree growth annually, making it difficult to pinpoint when CGW takes effect. In contrast, Ashwood et al. (2018) monitored tree growth on a bi-annual basis. Added to recent monitoring at 60-months this creates a more detailed overview of site development. It can now be more confidently asserted that the rapid increase in Italian alder growth observed between 6 and 18 months is the sole period of compost-induced growth, with no further compost related increase beyond 24-months at this site. This suggests Italian alder might benefit from a repeat compost application as recommended by previous authors (Foot et al., 2003; Beesley, 2012; Larney and Angers, 2012). A top dressing of mulch immediately around the base of each tree would be a low cost, low risk strategy worthy of consideration. Although Italian alder fixes its own nitrogen via root symbioses with *Frankia spp.,* Moffat et al. (2008) suspected poor nodulation could limit its performance. After 6 – 7 years of growth at three different landfills, these authors found Italian alder responded positively to chemical N-fertilization thus similar may be possible using surface compost mulch.

Regarding Norway maple, Ashwood et al. (2018) suggested site conditions inhibited this species growth, over-riding CGW beneficial effects. Findings at 60-months agree with this conclusion. Whilst an increase in CGW quantity may improve a given species performance, the experiments' location and configuration make this unlikely for Norway maple. For example, consider that Ingrebourne Hill is located along the Thames Estuary, one of the UK's hottest and most arid regions (Met-Office, 2011c). The experiment itself is situated at the crest of a hill with no surrounding shelter, leaving trees permanently exposed to dry prevailing winds. Further, Ashwood (2016) found soil pH at Ingrebourne Hill was alkaline (around 8.0), which is not ideal for many species in the *Acer* genus. According to Caudullo and de Rigo (2016), Norway maple shows tolerance for the conditions cited above, including drought, exposure, and calcareous soils. However, it is extremely intolerant of prolonged drought and high evapotranspiration, preferring moist, nutrient rich, sub-acid soils. Norway maple is also described as a secondary species, making it unsuitable for monoculture planting. A combination of warm and dry climate, prolonged exposure, and experimental effects arising from single species planting undermined its performance. Norway

Maple survival and growth may improve when planted in sheltered positions as a sub-component of mixed species stands.

5.8.2 Impact of compost addition on soil nitrogen, organic matter, and carbon storage

To understand how compost interacts with trees and reclaimed soils, tree growth and soil chemical variables were assessed. This helped reveal which out of compost and tree growth had a greater impact on soil development. Overall, this study finds compost had the greatest impact on soil development during the first 5-years, interacting positively with Italian alder, but altering soils under both tree species. This is evidenced by increased tree height, stem diameter, soil carbon, and soil nitrogen for Italian alder in both compost-based treatments. However crucially, soils under Norway maple contained significantly more total nitrogen, available nitrogen, and had lower C:N ratio's than the non-compost treatments under Italian alder. Given Norway maples poor performance and lack of nitrogen fixation, factors other than tree growth were likely responsible.

Indeed, regardless of tree species, compost instigated a sequence of soil chemical changes, beginning one year after incorporation. This suggests CGW has a greater effect than tree species on soil development, although the impact is not instantaneous. For example, the compost-induced changes began at 12-months with a spike in available nitrates, followed by a drop in C:N ratio between 12 and 18 months. Then, between 18 and 24-months soil organic matter, total carbon, and total nitrogen increase together, with the strongest effects observed in compost-based treatments. This sequence, plus the difference between compost and non-compost treatments lasting to 60-months, suggests compost-induces changes that occur progressively and sequentially over an extended two-year period, eventually becoming integrated within the soil matrix.

The fact a compost-induced increase in soil organic matter, total carbon, and total nitrogen is not detected until 2-years after application does require explanation. Likely, soil food-web organisms take time to metabolise and mineralise compost; and move the biological products of compost processing through carbon and nutrient cycles. Existing reclamation-based studies similarly indicate this. For example, Beesley (2012) showed compost addition increased microbial respiration in the first 6-months after application, illustrating how CGW stimulates a pulse of microbial activity shortly after application. This would be a prior step to the sequence observed at Ingrebourne Hill. A laboratory experiment by Deeb et al. (2017) demonstrated that for 6-months following CGW application to manufactured soil, CGW induces soil formation only when plant roots and soil dwelling earthworms are present. No matter the quantity of CGW applied, soil formation remained limited when plant roots and earthworms were absent. Daynes et al. (2013) reached similar conclusions to Deeb et al. (2017), albeit by studying soil fungi. Said-Pullicino et al. (2010) found CGW stimulated microbial mineralisation of soil organic matter, resulting in a long-term increase in organic matter content of landfill soils. Taken together these studies re-iterate the dynamics observed at Ingrebourne Hill. The seasonal and climatic cycles of this field experiment simply mean processes take longer and undergo fluctuations that are generally absent under laboratory conditions. Indeed, according to Said-Pullicino et al. (2010), CGW is retained within the soil system over time, further demonstrated by other field based studies which document soil organism's ability to increase soil organic matter, total nitrogen, and total carbon in a reclamation context (Scullion and Malik, 2000; Józefowska et al., 2017; Liang et al., 2017; Angst et al., 2021). The sequence of soil chemical events observed at Ingrebourne Hill support findings of these studies, but effects appeared to slow down as time progressed. Indeed, a build-up of NH4 and increasing soil C:N ratios at 60-months indicate rates of nitrogen cycling rates had in fact slowed.

Norway maple plots similarly outline composts influence, with some soil chemical effects stronger under this species despite poorer survival and growth. Interestingly, under Norway maple the earthworms-only treatment received no compost, yet soil C:N ratio and total nitrogen were equivalent to compost-based treatments. This suggests factors un-related to tree species or compost may be responsible, which agrees with numerous authors. For example, according to Bending and Moffat (1997), tree establishment takes 3 – 5 years, meaning tree contributions to SOM, carbon, and nitrogen within this timeframe are modest at best. Crucially, numerous field studies note how CGW improves growth of non-target vegetation, including herbaceous weeds, annual plants, and grasses (Sparke et al., 2011; Kilbride, 2014; Wall, 2016). Thus, the most likely explanation is that Norway maples absence and failure to close canopy allowed ground vegetation to thrive for an extended period. Norway maples failure to dominate meant the grass seed-mix used to suppress weed growth derived maximum benefit from compost addition through reduced shading and resource competition. Under Norway maple, interactions between ground vegetation, compost, and soil biology likely increased SOM, total nitrogen, and lowered soil C:N ratios. In contrast, Italian alder developed closed canopy within 5-years, shading out ground vegetation. Hence in general, Norway maple did not significantly contribute to soil development, whereas ground vegetation did altering soil chemistry in N. maple plots.

5.8.3 Impact of tree species and compost addition on earthworm populations

From 0 – 60 months compost appears to have impacted earthworm population densities whereas tree species did not. Nevertheless, alongside compost and ground cover plants, earthworms were likely one of several factors influencing ecosystem service provision at Ingrebourne Hill. For example, during the 0 to 30-month observations by Ashwood et al. (2018), earthworm densities increased significantly across the site. Densities were highest in the compost-based treatments, although the treatment effect was not significant. (Ashwood et al., 2018). As time progressed, earthworm population growth slowed, and was no longer significant between 30 and 60-months.

Falling population growth rates from 30 to 60-months suggest compost may have influenced earthworm density early on. Indeed, composts subtle influence on earthworms can be observed in earthworm inoculated plots. For example, when CGW was added to compost-plus-earthworm treatments, population densities were relatively high at 30-months. However, by 60-months they had declined becoming the lowest throughout the site (43 m^{-2}). Examining the earthworms-only treatment over the same period, population densities increased steadily from 0 – 60 months (51 m^{-2}), eventually surpassing the compost-plus-earthworm treatment at the final observation. This suggests compost-stimulated a population rise from 0 to 30-months (Ashwood et al., 2018), however as CGW was utilised by earthworms the supply diminished, causing populations to stabilise then decline. In summary, when compost was added to inoculated plots, earthworm populations increased significantly. When compost was not added to inoculated plots, there was no significant population increase. At Ingrebourne Hill the rise and fall in earthworm numbers was not always statistically significant, yet the pattern resembles dynamics observed in previous landfill reclamation studies.

For example, Butt et al. (2004) and Lowe et al. (2008) conducted multi-year studies of population dynamics at Calvert and Clifton Marsh landfills, following application of composted green waste. They found CGW increased earthworm populations from 0 to 2 years, however after 4-years CGW disappeared, and earthworm populations declined. This suggests compost can stimulate boom and bust dynamics in earthworm populations, provided enough earthworms are present within soil or nearby when compost is applied. Arguably, similar dynamics have now been observed at Ingrebourne Hill, albeit to a subtle degree given smaller earthworm populations were present and less compost was applied (80 t-ha^{-1}).

Moving on, the most common earthworms at 30 and 60-months were *A. chlorotica* and the inoculated species *A. longa*. Ashwood (2016) reported the inoculation technique utilised at Ingrebourne Hill was unsuccessful resulting in widespread mortality. Thus, the presence of *A. chlorotica* and *A. longa* at Ingrebourne Hill is most likely linked to their common and widespread UK distribution. For example, according to Natural England, of the UK's 27 native species *A. chlorotica* is the most abundant and widespread species, whilst *A. longa* is seventh most abundant and fourth most widespread in England (NE, 2014). Similarly, both species demonstrate a unique ability to recolonize reclaimed soils and re-establish viable populations (Piearce et al., 2003; Marashi and Scullion, 2003; Butt et al., 2004; Butt, 2008; Butt and Putwain, 2017). The relative success of *A. chlorotica* and *A. longa* at Ingrebourne Hill may be linked to their preference for disturbed habitat. Thus, the present study provides further evidence of these species suitability to reclaimed and manufactured landfill soils.

Regarding earthworm impacts on tree growth and soil development, earthworm densities increased significantly from 0 to 30-months alongside accelerated tree growth and soil chemical change. This suggests compost directly benefitted earthworm populations present at Ingrebourne Hill. In turn, earthworm bioturbation enhanced soil formation, likely benefitting tree growth. However, earthworm impacts on trees appeared to be secondary or indirect with no obvious trends or statistically significant interactions present between tree species and earthworms. Instead, earthworms remained most closely associated with compost, and were likely one of several factors influencing tree growth.

An examination of soil chemistry in the earthworms-only treatment demonstrates how several factors combined to promote soil and vegetation recovery. Consider, compost was not added to earthworms-only treatments yet soil chemistry differed in this treatment. Soil chemistry in Norway maple earthworms-only was comparable to compost amended treatments. Conversely, soil chemistry in Italian alder earthworms-only was comparable to controls. Why is this? If anything, Italian alders nitrogen fixing capabilities should produce soils with similar nitrogen profiles to compost amended plots. Conversely, Norway maple performance was poor, limiting its impact on soil development, so how did this tree species create soil chemistry similar to compost amended plots. Indeed, unamended Norway maple earthworms-only treatments had: 1). significantly lower soil C:N ratio than Italian alder earthworms-only; 2). significantly more total nitrogen than Italian alder earthworms-only; 3). significantly more SOM than Norway maple control. This is despite the fact that Norway maple was deficient in foliar-N. The most likely explanation is that ground vegetation growth in Norway maple plots altered soil chemistry. Consider, Italian alder performance was relatively good, shading out ground vegetation through fast short-term growth. This rapid growth also allowed Italian alder trees, and not ground vegetation, to utilise resources available throughout the site, whereas Norway maples inability to adapt to site conditions meant ground vegetation benefitted from increased light and soil resources. In Norway maple plots there may have been fewer competitive interactions between trees and groundcover meaning fewer soil resources were stored in above-ground woody biomass. With fewer competitive pressures, interactions between ground vegetation and earthworms increase, concentrating inputs and resources below-ground. Indeed, although large densities of *A. chlorotica* (1060 and 1770 m^{-2}) found at Ingrebourne Hill were located in control and compost-only treatments in a single block, high densities both occurred in Norway maple plots.

In summary, the addition of compost appears to have temporarily boosted earthworm populations, and in turn compost and earthworms appear to have benefitted growth of any plant adapted to site climatic conditions. All introduced resources including seeded ground cover, compost, trees, and earthworm biomass worked together promoting soil development and vegetation growth. From 0 – 24 months earthworms exerted a subtle impact on site development. This is because a small population was present alongside a relatively modest quantity of CGW (80 t-ha^{-1}) limiting the intensity of interactions between soil resources and organisms. Earthworms supported soil chemical development, however during the 0 to 60-month observation this appears to have been primarily mediated via interactions between compost, earthworms, and ground vegetation. Earthworm interactions with tree species appear to have been secondary or supplementary to this over a 5-year period.

CHAPTER 6. ORGANIC AMENDMENTS IN CENTRAL SCOTLANDS GREEN NETWORK: Comparing the effects of composted green waste at two adjacent landfills near Glasgow, 10 years after application

6.1 Greenoakhill – Background

Greenoakhill is an 85-ha landfill near Uddingston, Glasgow, Scotland (Nat Grid Ref: NS 66622 62760)**.** The site is split into two main sections by the M74 motorway. Active landfill operations are ongoing at the northern site, whilst the southern site has been reclaimed to community woodland end-use **(FIGURE 6.1, FIGURE 6.8, FIGURE 6.9)**. The southern section forms part of the Central Scotland Green Network (CSGN), a long-term multi-agency initiative aiming to expand woodland cover and greenspace throughout central Scotland. Scottish Forestry (SF), formerly Forestry Commission Scotland, are responsible for developing the CSGN and manage all community woodland aspects of the southern site. SF also maintain a long-term interest in converting active landfill at Greenoakhill north to community woodland when operations cease (Wall, 2014). The steep slopes and hills which characterise Greenoakhill today are the product of more than 50 years of landfill. Throughout its lifespan the site has followed a typical cycle of industrial land-use, beginning with sand quarrying in the 1940s, evolving to landfill by the 1960s. By 2008, the site was receiving 500,000 tonnes of mixed waste per annum, with the southern landfill nearing completion. At this time, and in anticipation of site conversion to community woodland end-use, two formal experiments were established. Their purpose was to identify how much compost should be added to manufactured soils intended for woodland planting in 2010/11.

FIGURE 6.*1 – Aerial image of Greenoakhill (GOH). The site is divided into two sections by the M74 motorway, Greenoakhill north (GOH NORTH) and Greenoakhill south (GOH SOUTH). In 2008, a formal experiment was established on each section (smaller yellow shapes).*

6.2 Site Conditions

Annual climate data from surrounding weather stations show between 1981 and 2010, Greenoakhill received an average of 1100 – 1250 mm rainfall, 1200 – 1300 sunshine hours, with a mean annual temperature range of 6 – 13 °C (Met-Office, 2011b). Greenoakhill is the coolest, wettest site studied, with fewest sunshine hours of all. A formal experiment has been established on each section, referred to herein as GOH-NORTH (Greenoakhill North - Nat Grid Ref: NS 66622 62760) and GOH-SOUTH (Greenoakhill South - Nat Grid Ref: NS 66412 62135)**(FIGURE 6.1).** The experiments are considered separate, given their distinct locations with conditions differing markedly at each.

6.2.1 Greenoakhill North

Greenoakhill North experiment is located inside 65 ha of privately owned landfill. It is situated near the main entrance and weighbridge area on a capped landfill parcel **(FIGURE 6.2)**. The ground slopes towards a busy access road which runs alongside the experiment. HGVs use this to transport waste materials to active landfill cells **(FIGURE 6.3, FIGURE 6.4)**. A passenger railway and A74 carriageway bound the experiments northern side. Closed canopy woodland sits immediately east and west protecting the experiment from prevailing winds hence exposure is moderate.

6.2.2 Greenoakhill South

Greenoakhill South is located on a smaller 20 ha landfill, reclaimed to community woodland in 2010/11. The terrain surrounding the southern experiment can be described as newly reclaimed developing woodland / scrub. The experiment is located on the crest of a hill on GOH-SOUTH's highest point **(FIGURE 6.5, FIGURE 6.6, FIGURE 6.7)**. Immediately to the experiment's north, a steeply inclined embankment slopes down toward the M74, forming a natural barrier along the site's northern boundary **(FIGURE 6.8, FIGURE 6.9)**. The River Clyde straddles the site's southern boundary separating it from adjacent land. Sewage works, an aggregate processing facility, and reservoir occupy land immediately east. The site is isolated and exposed compared to GOH-NORTH.

FIGURE 6.2 – *The GOH-NORTH experiment as viewed from the main entrance and weighbridge area of the northern site. As seen, the experiment is situated on a roadside bank on completed landfill.*

FIGURE 6.3 – *Landfill access road as viewed from the middle of the GOH-NORTH experiment.*

FIGURE 6.4 – *Side view of an experimental block at GOH-NORTH. The slope incline is more visible from this perspective. Photo taken from middle of northern experiment.*

FIGURE 6.5 – *Experiment at GOH-SOUTH as viewed from the eastern end of the southern site. The experiment is situated on open ground at the crest of a hill. The pylon marks the centre of the southern site and overlooks the sewage works. The M74 can be seen with GOH-NORTH to the right of the motorway.*

FIGURE 6.6 – *Experiment at GOH-SOUTH approaching from the east walking towards the crest of the hill.*

FIGURE 6.7 – *Close up view of Birch trees at the boundary of GOH-SOUTH experiment. This is the control plot of Block A.*

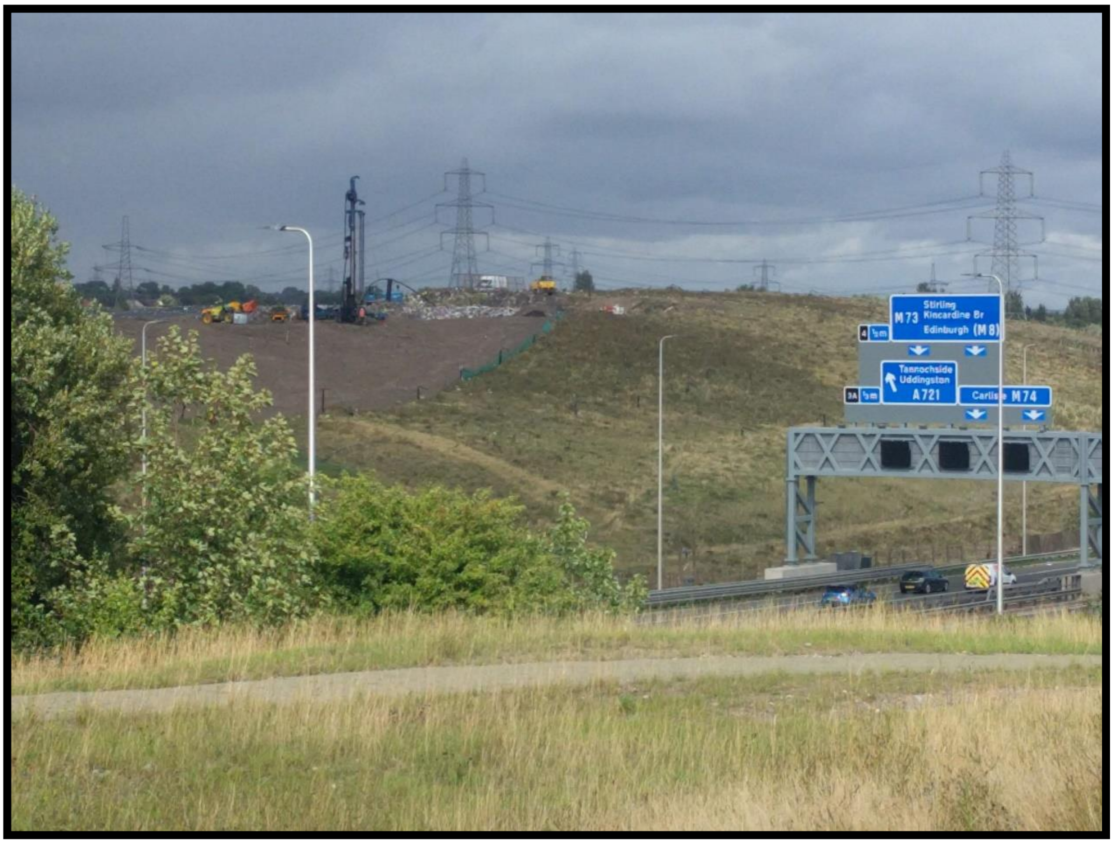


FIGURE 6.8 – *Steep banks at GOH-SOUTH (foreground). The site overlooks the dividing M74, and GOH-NORTH landfill (background) where active areas of landfill can be seen.*

FIGURE 6.9 – *View of M74 from the northern bank of GOH-SOUTH. Woodland was planted along this bank in 2008, however in some areas growth was poor due to compaction and nutrient limitation.*

6.3 Soil Materials

During their lifetime, Greenoakhill North and South have received contrasting types of refuse, producing landfills with different topsoils and different management and engineering strategies utilised to deal with their differing contents (Wall, 2014; Wall, 2016).

6.3.1 Greenoakhill North

Firstly, Greenoakhill North has always received organic and putrescible waste. A 'dilute and disperse' strategy has been deployed to manage waste decomposition. This means refuse materials are spread over large areas then compacted down in layers, with no under lining to prevent leachates migrating into ground water and adjacent ecosystems. An engineered clay cap seals landfill material from above. A manufactured topsoil material has been tipped over the cap, creating a 1 m deep rooting medium. Topsoil is described as silty sand, loose and well-draining (Wall, 2016). The topsoil is rubble free, with the cap separating plant roots from landfill waste **(FIGURE 6.10)**. A pipe network rests between the cap and waste, transporting methane to atmosphere and to on-site engines, generating electricity for National Grid. This diverts landfill gas away from soil, preventing the cap from rupturing and protecting plant roots from suffocation (Wall, 2014; Wall, 2016).

FIGURE 6.10 – *Soil excavated from the 300 t-ha^{-1} treatment in Block D of the northern experiment. Soil around grass roots has an aggregated structure.*

6.3.2 Greenoakhill South

Greenoakhill South similarly relies on 'dilute and disperse', but has received inert commercial, industrial, and geological waste. The lack of decomposing organics means clay capping and gas collection infrastructure are not necessary and landfill discharge is considered inert. Southern soils are defined as clay loam and are composed of geological drift and subsoil from engineering and construction projects (Wall, 2016). Site soils are described as dense, cohesive, with a high clay content, and large amounts of fragmented concrete, masonry, and rocks **(FIGURE 6.11)**. The upper 1 m has been graded with finer materials to support plant root growth. Plants root directly into the substrate with no barrier to prevent root ingress into the sub-surface landfill.

The southern site was reclaimed in two phases. The first undertaken between 2003 and 2006 as the landfill neared completion. This southeastern section, and banks along the northern and western perimeters received grass seeding, plus shrub and broadleaf tree planting. In some areas trees were planted directly into ungraded landfill, however compaction and lack of nutrients limited growth (Wall, 2014). A second phase was completed in 2010/11 with 5.5 ha of woodland planting. This phase was considered a greater success as soils were cultivated to 1 m depth with composted green waste incorporated into the upper 0.5 m (Wall, 2014).

FIGURE 6.11 *– Soils at the southern site illustrated as follows:* ***1).*** *Fragments of concrete and masonry cover the soil surface on land approaching the southern experiment.* ***2).*** *Soil excavated from experimental blocks shows finer graded material mixed with rocky fragments.* ***3).*** *Close up of excavated soil shows cloddy angular structure with broken masonry.*

6.4 Description of field experiment

The experiments at GOH-NORTH (Nat Grid Ref: NS 66622 62760) and GOH-SOUTH (Nat Grid Ref: NS 66412 62135) were both established in 2008 using Waste Resources Action Programme (WRAP) funding. The experiments were used to develop a soil specification for reclamation works planned at Greenoakhill South in 2010/11. Test treatments included an experimental control plus three rates of composted green waste (CGW) set at 300, 600, and 1200 t-ha^{-1}. PAS100 principal grade soil improver, particle size 0-20 mm was incorporated into manufactured soils to 0.5 m depth. Each treatment was replicated 4 times creating 16 randomised plots in each experiment **(FIGURE 6.12)**. The 1200 t-ha^{-1} treatments were not evaluated during this study, as several authors consider such quantities excessive (Beesley, 2012; WRAP, 2012; Wall, 2016). *Betula pendula* (Silver birch) was the only tree species planted, with 36 in each plot planted at 1.5 x 1.5 m spacing. The original trials observations lasted for 18 months from June 2008 to November 2009 (Wall, 2016). Results from this trial suggested CGW increased soil organic matter, and available nutrients at both experiments, however tree growth data remained un-published. Nevertheless, Wall (2016) concluded a 600 t-ha^{-1} treatment was optimal for maintaining acceptable tree performance, whilst avoiding excessive weed growth. Consequently, 750 t-ha^{-1} CGW was specified for woodland planting in the 2010/11 land reclamation to woodland project (Wall, 2014).

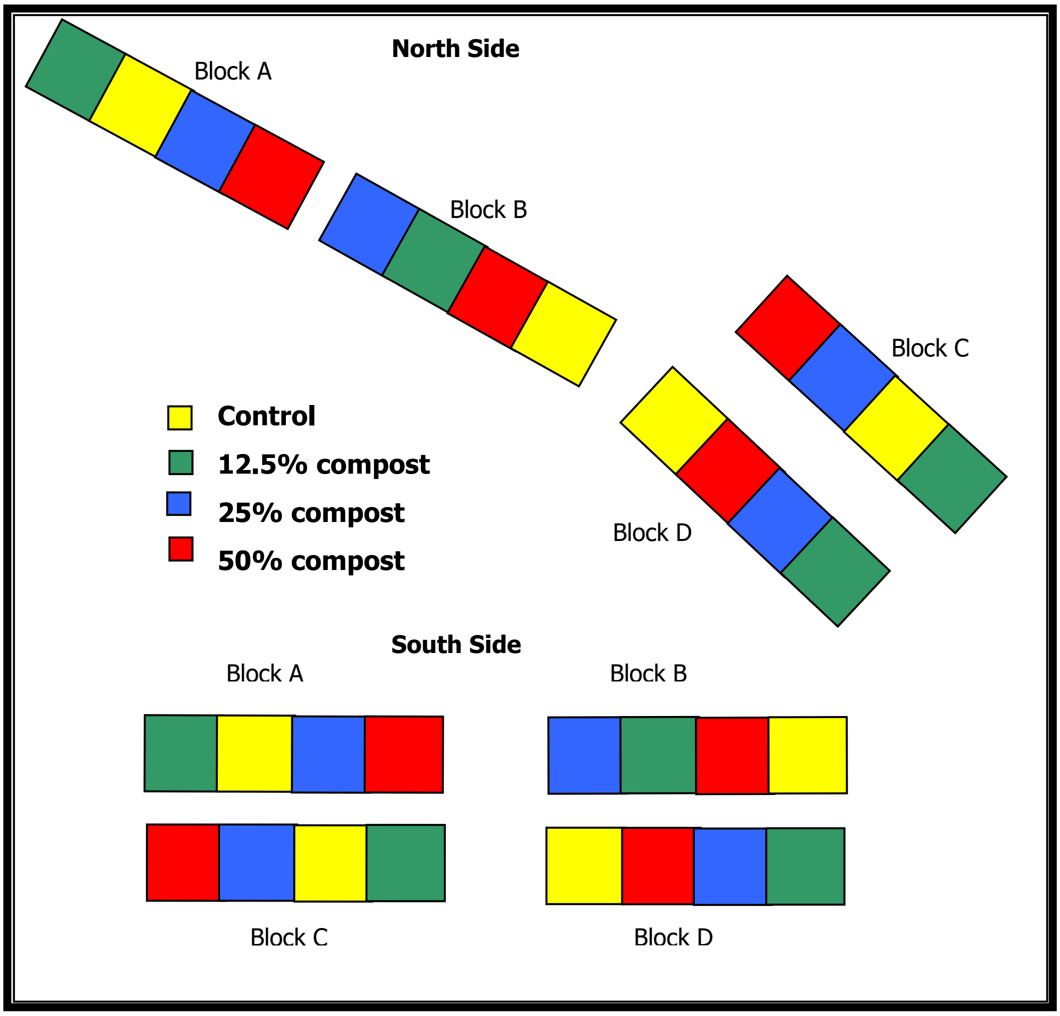


***FIGURE 6.12 –** Layout of Northern and Southern experiments. Each experiment included 16 plots set out in a randomised sequence (4 treatments X 4 replicates = 16 plots). Three rates of CGW were applied as indicated in the figure (Wall, 2016).*

6.5 Selection rationale

Greenoakhill was selected for study based on several factors, namely site age, quantity of compost applied, and presence of duplicate experiments. First, Greenoakhill was 10 years old when sampled for this study. This meant tree growth and soil pedogenesis could be analysed from the mid-point of a 20-year study site age range. Second, Greenoakhill received the largest quantities of compost, with greater increments between rates than any other site. This allowed the efficacy of large CGW application rates to be analysed following an extended 10-year period. In addition, because the experiment had been duplicated on adjacent sites with differing topography, microclimate, and soil materials, the effects of site conditions on site development could be compared. Tree species were also unique at this experiment, with *B. pendula* planted throughout. This differentiated Greenoakhill from the other study sites, and meant interactions between CGW, earthworms, and Silver birch could be observed.

Further, it was recognised the original study by Wall (2016) had limitations which future studies needed to explore, centering on the experiment’s 18-month duration. This short timeframe meant long-term developmental processes could not be observed. Thus, as time had moved along further monitoring could now be conducted by a subsequent study. Similarly, carbon fate was identified for further exploration. The original study concluded CGW mineralisation increased soil organic matter content and available nutrients, however the extent to which CGW had become stabilised as carbon in soil remained unknown. No soil biological observations had been made at Greenoakhill. Hence, by obtaining earthworm related data alongside soil physical and chemical variables, the extent to which CGW and earthworms interactions increased carbon storage could be assessed.

6.5.1 Aims and objectives

Explore CGW legacy impacts (defined as 5+ years) on provision of ecosystem services related to tree growth, soil formation, earthworm activity, and soil organic carbon fate.

1. Determine CGW long term impact on above-ground tree growth, biomass production, and foliar nutrient content at reclaimed sites;

2. Record physical, chemical, and biological attributes of CGW-amended soils to identify potential links between CGW application rate, CGW incorporation depth, soil formation, and carbon storage;

3. Determine how earthworm community composition, density, and biomass impact pedogenesis in CGW-amended soils;

4. Outline CGW and earthworms synergistic impact on supporting and regulating ecosystem services.

6.6 Materials and Methods

The following information summarises the sampling and analysis techniques used to investigate interactions between compost application, tree growth, soil chemistry, and earthworm activity at Greenoakhill. A more detailed description of the techniques used to evaluate all study sites is provided in **CHAPTER 4 – GENERAL MATERIALS AND METHODS**. Similar techniques were used at all sites to ensure data generated could be readily compared and discussed.

Tree growth and foliar chemistry were measured at Greenoakhill to understand how large quantities of compost might influence primary production and nutrient cycling 10-years after application. Tree height was measured in February 2019 when foliage was minimal using a Haglof Vertex IV digital hypsometer and ultrasonic transponder. Stem diameter was recorded using a circumference tape manufactured by Richter. To ensure leaf nutrient concentrations were steady, foliar collections were completed between mid-summer and late summer in July 2019. Foliage was taken from several different trees per plot and bulked to create a plot representative sample. Foliage was oven-dried, homogenised, then sieved to obtain smaller particles 1 – 2 mm in size. A sub-sample was then taken and ground to powder in a planetary ball mill, ensuring foliage was ready for carbon and nitrogen analysis. Further details are provided in **SECTION 4.2.1** and **4.2.2**.

To evaluate compost impact on soil nutrient cycling, and carbon storage, soil cores were taken from each experimental plot in May 2019. A Dutch auger fitted with 15 cm attachment was used to extract duplicate cores from 0 – 15 and 15 – 30 cm depth. Cores were taken from two different locations per plot, prepared for analysis, then bulked to create a plot-representative sample for each depth. Soil organic matter content, total carbon, total nitrogen, C:N ratio, and available nitrates were all assessed. Available-N was determined by taking 10 g of fresh soil, performing a KCL-extraction, then sending soil extracts to Rothamsted Research laboratory for colorimeter analyses. For the remaining soil chemical analyses, cores were air dried, crushed in a motorised blade grinder, and sieved to > 2 mm to remove stones and other large particles. Replicate cores were then bulked and sieved again to 0.5 mm, creating a fine homogenised soil. This fine dried material was used to determine organic matter content via loss on ignition, and total carbon, total nitrogen, and C:N ratio for soil and foliar samples using a Thermo-Scientific CHNS Organic Elemental Analyser. Further details are provided in **SECTIONS 4.3.3** and **4.3.4**.

Soil bulk density (BD) was used as a proxy for soil formation. Cores also provided readings for soil moisture content. Cores were extracted from earthworm pits in April / May 2019 when earthworms were sampled. Three depths for BD (0 - 10, 10 - 30, and 30 - 50 cm) were assessed. A hammer and 100 cm^3 cylindrical steel core were used to extract BD cores from the side and base of two 30 x 30 x 40 cm^3 earthworm pits, providing duplicate cores for each sampling depth (Robertson et al., 1999). Cores were oven dried at 105°C for 24 hours to establish soil bulk density, moisture content, and total porosity. Further details are provided in **SECTION 4.3.1** and **4.3.2**.

Earthworm population density and community structure was measured to explore how interactions among compost, earthworms, and tree species mediate ecosystem service provision. Sampling was conducted in April-May 2019 following the methods described by Butt and Grigoropoulou (2010). In each plot, two earthworm pits were excavated to obtain a plot representative population sample. A 0.1 m^2 quadrat was placed on the soil surface, then 30 x 30 x 40 cm^3 of soil was excavated from beneath this and hand-screened for earthworms. Specimens were placed in plastic bottles and preserved in 4% formaldehyde solution prior to identification. All adult worms were identified to species level following the key of Sims and Gerard (1999). Visual identification was completed using a stereo microscope with adjustable zoom (X10 – X30 magnification). Specimens were dried on absorbent paper then weighed to determine biomass. Population densities found in each 0.1 m^2 pit were scaled up and reported as average number of species / eco-group per m^2. Further details are available in **SECTION 4.4**.

6.7 Statistical Analysis

Statistical analysis was performed using two different open-source software packages. These were 'JASP' (Version 0.9.0.1) statistical software, followed by 'R – script' (Version 4.0.2) and its associated 'R – studio' user interface (Version 1.3.959). JASP was used for: one-way analysis of variance (ANOVA), repeated measures ANOVA (RMANOVA), and regression, whilst 'R' was used for principal component analysis (PCA). The statistical techniques used in JASP required data normality to be assessed in various ways. Firstly, histograms and Q-Q plots were used to graphically / visually evaluate whether data was normally distributed. Next, numerical values for skewness and kurtosis were considered. If either was above 1.96, data was deemed to deviate significantly from normality. Finally, Levene's test was used to evaluate statistical variance between groups. Where Levene's test was significant ($p < 0.05$), variance was deemed to be unequal violating the assumption of normality (Goss-Sampson, 2018). Where data was not normally distributed and log-distribution could not correct it, non-parametric alternatives were applied. Kruskal-Wallis non-parametric test followed by Dunn's post-hoc replaced one-way ANOVA. Non-parametric Friedman's test followed by Connover's post-hoc replaced one-way RMANOVA. (Goss-Sampson, 2018). And finally, to reduce type-1 errors, bonferroni corrections were applied to all significant results.

With compost, tree growth, earthworms, and other factors simultaneously driving site development, it was important to determine which had the greatest influence on restoration processes. Thus, using the same data, PCA was performed in 'R' to reveal which variables were chiefly responsible for the effects revealed in JASP. PCA observes data in three-dimensional space by overlaying multiple linear equations (vectors) over the dataset. This allows PCA to explore which of the multiple factors measured are driving the variance observed. Once dominant variables are identified and ranked by order of influence, PCA re-expresses all data around the dominant variables. Results are illustrated using a biplot which provides a visual representation of site dynamics. PCA is completely non-parametric, making no assumptions about normality or how data is recorded. However, PCA does function using the major assumption that all variables are linear. This simplifies the data by eliminating all secondary correlations, restricting the number of relationships and connections present. PCA is also a descriptive technique, meaning it explains only the data to which it is applied, and cannot be used to make inferences about a larger statistical population (Shlens, 2014; Jolliffe and Cadima, 2016). Using PCA, results from GOH-NORTH and GOH-SOUTH were compared to examine how CGW might impact two replicate experiments with differing site conditions, and how this might influence the restoration process.

6.8 Results

6.8.1 Impact of compost addition on tree survival, growth, and foliar nutrition

Tree growth and survival were recorded in February 2019, 10 years and 8 months after the experiments were established. Comparing tree survival at GOH-NORTH and GOH-SOUTH; there was no significant difference between sites (Kruskal-Wallis non-parametric ANOVA, H (2) = 0.116, p 0.944); or between treatments (Kruskal-Wallis non-parametric ANOVA, H (5) = 5.885, p 0.318). Tree survival was slightly higher at GOH-SOUTH (87%) compared with GOH-NORTH (79%), however in general survival was similar throughout **(TABLE 6.4)**.

In contrast, compost increased tree height at both sites, with trees in amended plots significantly taller than controls (one-way ANOVA, F (5, 712) = 42.58, $p < 0.001$). Post-hoc tests with bonferroni correction revealed Silver birch at GOH-NORTH were significantly taller than at GOH-SOUTH ($p < 0.006$), with a 1.2 – 2 m difference between corresponding treatments. However, at GOH-NORTH only the 300 t-ha^{-1} treatments grew significantly taller than controls ($p < 0.05$). Whereas at GOH-SOUTH, both the 300 and 600 t-ha^{-1} treatments grew significantly taller than controls ($p < 0.004$) **(FIGURE 6.13)**.

A similar pattern was observed for stem diameter, which increased significantly at both sites in compost amended plots (Kruskal-Wallis non-parametric ANOVA, H (5) = 99.10, $p < 0.001$). Again, post-hoc tests for GOH-NORTH showed that only the 300 t-ha^{-1} treatment displayed a significant increase in stem diameter ($p < 0.01$), whereas at GOH-SOUTH both the 300 and 600 t-ha^{-1} treatments were significantly greater than their respective controls ($p < 0.001$) **(FIGURE 6.14)**.

Greenoakhill - Tree Survival (%)		
Treatment	GOH-NORTH	GOH-SOUTH
0 t-ha	79.2 ± 3.5	86.8 ± 5.8
300 t-ha	81.9 ± 7.3	85.4 ± 2.9
600 t-ha	76.4 ± 5.4	88.9 ± 3.6
SITE TOTAL	79.2 ± 5.4	87.0 ± 4.1

***TABLE 6.4** – Percentage survival at the northern and southern experiments.*

FIGURE 6.13 *– Impact of CGW quantity on B. pendula height. A difference in letters indicates significant differences between treatments. Independent one-way ANOVA with bonferroni correction showed CGW increased B. pendula height significantly at both sites compared with controls (F (5, 712) = 42.58, p <0.001). Trees also grew significantly taller at* ***GOH-NORTH*** *compared with* ***GOH-SOUTH*** *(p <0.006).*

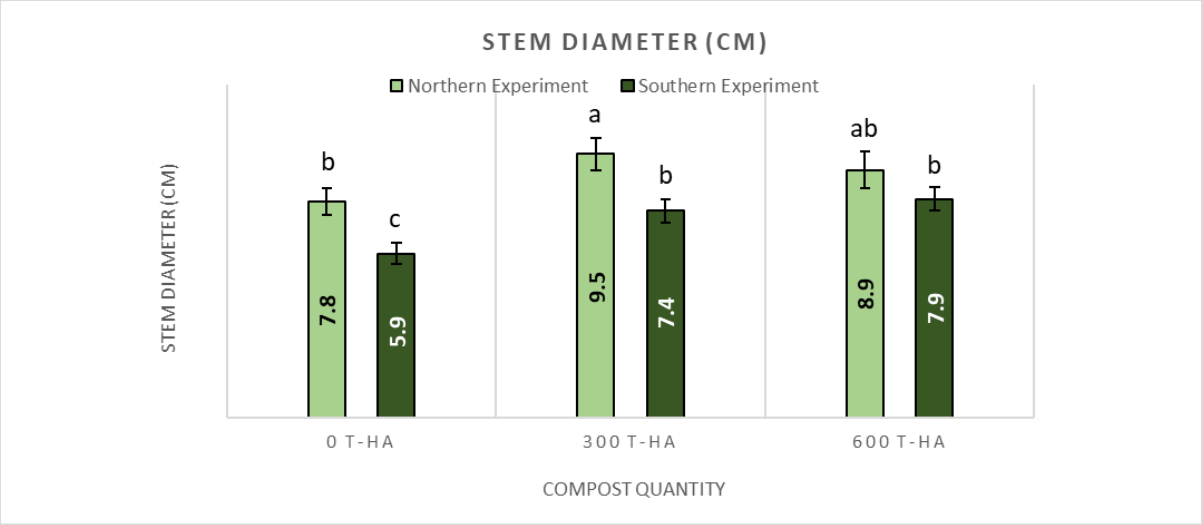


FIGURE 6.14 – *Impact of CGW quantity on B. pendula stem diameter. A difference in letters indicates significant differences between treatments. Kruskal-Wallis H test with bonferroni correction showed CGW increased B. pendula stem diameter significantly compared with controls (H (5) = 99.10, p <001). Trees at GOH-NORTH again outperformed trees at GOH-SOUTH although the difference was not always significant, being largely restricted to differences between controls and 300 t-ha^{-1} treatments.*

Given CGW ability to improve Silver birch growth, its effect on nutrient cycling was evaluated by examining links between compost quantity and foliar nitrogen. At GOH-NORTH, foliar C:N ratio was significantly lower than at GOH-SOUTH (Kruskal-Wallis non-parametric ANOVA, H (1) = 8.910, p <0.003). Indeed, foliar C:N at GOH-NORTH was below 20:1, with 600 t-ha^{-1} treatments lowest of all (17:1). Conversely, at GOH-SOUTH foliar C:N ratios were above 20:1, and highest under 600 t-ha^{-1} treatments (24:1), highlighting a definite contrast **(FIGURE 6.15)**. This opposing trend between sites was the only significant effect compost had on foliar C:N ratio (Kruskal-Wallis non-parametric ANOVA, H (5) = 13.40, p <0.02), and the pattern can be better understood by examining foliar nitrogen alone. For foliar-N, trees at GOH-NORTH surpassed FC guideline values of 2.5 %. Conversely, at GOH-SOUTH foliar nitrogen fell below this guideline value, with the 600 t-ha^{-1} treatments the most deficient (2.07 %) **(FIGURE 6.15)**. Thus, at GOH-SOUTH a lack of foliar-N produced high ratio of C to N in leaves, with the deficiency intensifying at the higher compost rate.

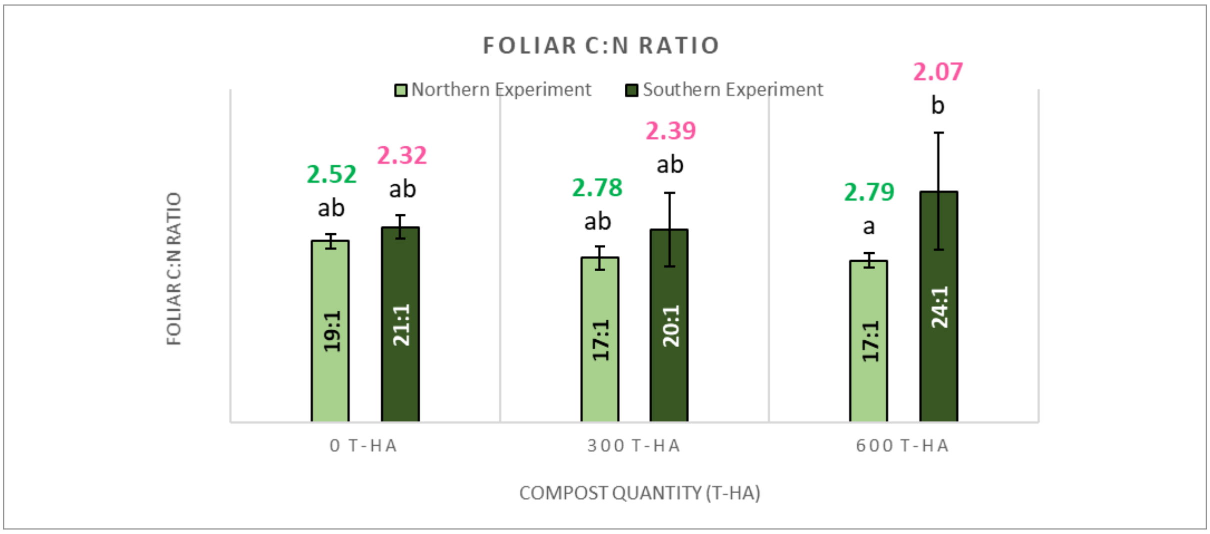


FIGURE 6.15 *– Impact of CGW quantity on* *FOLIAR C:N RATIO. A difference in letters indicates significant differences between treatments. Trees at GOH-NORTH had significantly lower foliar C:N ratio than trees at GOH-SOUTH (Kruskal-Wallis non-parametric ANOVA, H (5) = 13.40, p <0.02). Post-hoc tests revealed the difference was most pronounced in the 600 t-ha^{-1} treatments (p <0.04). Values above bars represent (%) foliar-N.* ***GREEN =*** *above FC guidelines of (2.5 %).* ***PINK =*** *below FC guidelines of (2.5 %). Silver birch at GOH-SOUTH were moderately deficient in foliar-N.*

Although tree growth was slower at GOH-SOUTH and foliar tissues contained less nitrogen, soil C:N ratios were similar at both experiments suggesting the issue was not a lack of nitrogen supplied by compost. Indeed, soil C:N was not significantly different between sites, yet within site, soil C:N in 300 and 600 t-ha^{-1} treatments were significantly different from one another and respective controls (Kruskal-Wallis non-parametric ANOVA, H (5) = 28.61, p <0.001; Dunn's post-hoc test with bonferroni correction, p <0.012). Overall, wherever CGW quantity increased, soil C:N ratios declined, reaching 20:1 or lower when compost was incorporated. Each increase in compost quantity lowered soil C:N further, however the margin diminished with each increase suggesting the CGW effect on soil has upper limits **(FIGURE 6.16)**.

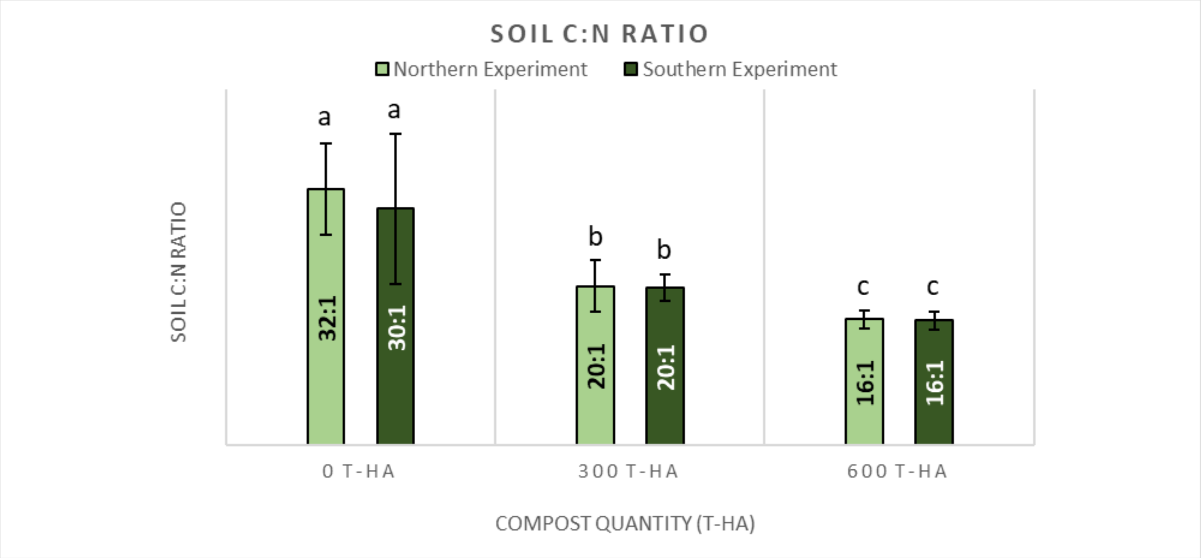


***FIGURE 6.16** – Impact of CGW quantity on SOIL C:N ratio. A difference in letters indicates significant differences between treatments. Soil C:N was significantly lower in amended treatments compared with controls (Kruskal-Wallis non-parametric ANOVA, H (5) = 28.61, p <0.001). Dunn's post-hoc tests revealed all treatments were significantly different from one another (p <0.05). The exact same pattern was observed at both sites, with significant differences between all treatments, but never between sites.*

6.8.2 Impact of compost addition on soil physical quality

After more than 10-years of soil development, soil physical parameters were assessed. Soil bulk density (SBD) served as a proxy indicator for soil formation, and at both sites significant differences were found between controls and amended plots (Kruskal-Wallis non-parametric ANOVA, H (5) = 30.11, $p < 0.001$). At GOH-NORTH, bulk density was significantly lower in the 300 t-ha^{-1} treatments ($p < 0.01$), whereas at GOH-SOUTH bulk density was significantly lower in the 600 t-ha^{-1} treatments ($p < 0.001$). Overall, as compost quantity increased, SBD decreased and was lowest overall in GOH-SOUTH 600 t-ha^{-1} treatments (0.67 g m^{-3}) **(FIGURE 6.17)**.

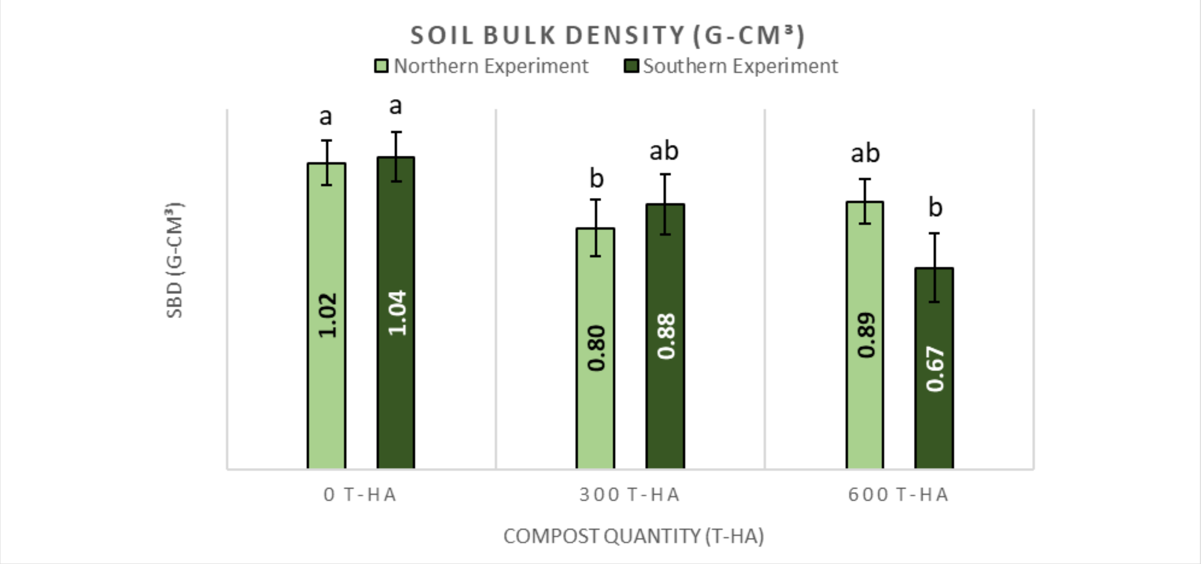


***FIGURE 6.17** – CGW impact on soil bulk density (SBD). A difference in letters indicates significant differences between treatments. SBD was significantly lower in amended treatments compared with controls (Kruskal-Wallis non-parametric ANOVA, H (5) = 30.11, p <0.001). Overall, as compost quantity increases, SBD decreases.*

Examining soil moisture (% water) at both sites and comparing amended plots with controls, CGW addition significantly increased soil moisture (Kruskal-Wallis non-parametric ANOVA, H (5) = 25.67, p <.001). At GOH-NORTH the 300 t-ha^{-1} treatments retained significantly more moisture than controls (p <0.04). However, the greatest effects were observed at GOH-SOUTH where the 600 t-ha^{-1} treatments retained significantly more moisture than controls at either site (p <0.04) **(FIGURE 6.18)**.

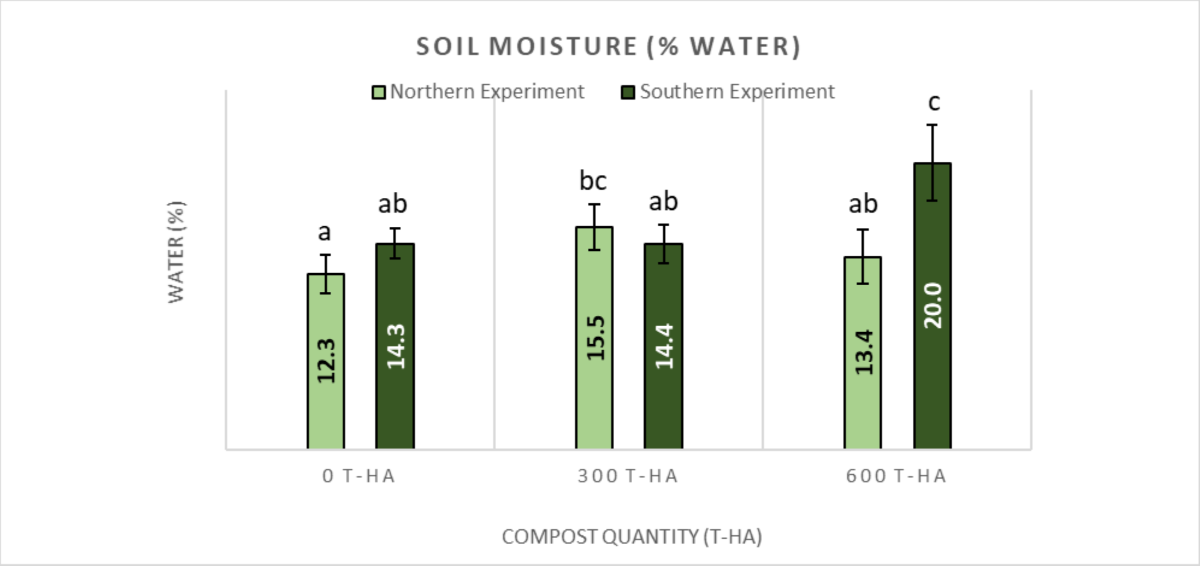


FIGURE 6.18 – CGW impact on soil moisture (% Water). A difference in letters indicates significant differences between treatments. Soil moisture was significantly higher in compost amended plots compared with controls (Kruskal-Wallis non-parametric ANOVA, H (5) = 30.11, p <0.001). Overall, as compost quantity increases, soils can retain more moisture.

6.8.3 Impact of compost addition on soil nitrogen, organic matter, and carbon storage

After 10 years and 8-months, CGW impacts on soil chemistry were assessed by collecting soil cores in May 2019. Cores provided data on soil organic matter (SOM), soil carbon, and soil nitrogen, all of which CGW were expected to increase. Regarding SOM (%), similar trends were observed at both sites. Soils amended with compost contained significantly more SOM than controls (Kruskal-Wallis non-parametric ANOVA, H (5) = 22.08, p <.001). At both sites, SOM rose incrementally with increasing compost quantity. However the effect was non-additive, diminishing with increasing quantities of compost (**FIGURE 6.19**). Consequently, differences between the 300 and 600 t-ha^{-1} treatments were not significant.

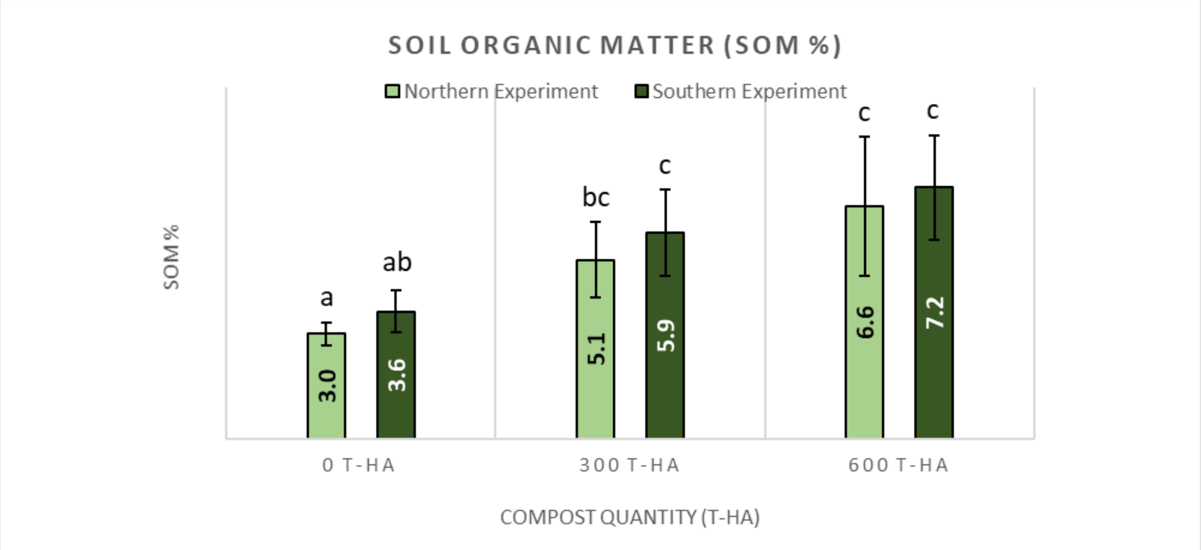


***FIGURE 6.19** – CGW impact on soil organic matter (SOM %). A difference in letters indicates significant differences between treatments. At both sites SOM was significantly higher in compost amended plots compared with controls (Kruskal-Wallis non-parametric ANOVA, H (5) = 22.08, p <.001).*

Using the same soil material, total carbon (t-ha^{-1}) was assessed and significant treatment effects were observed (independent one-way ANOVA F (5, 42) = 2.547, p <0.05). Yet a post-hoc test with bonferroni correction revealed just one significant difference. Namely, the 600 t-ha^{-1} treatment at GOH-NORTH contained significantly more carbon than the control at GOH-SOUTH (p <0.05). Furthermore, when CGW quantity doubled at GOH-NORTH, total carbon increased accordingly **(FIGURE 6.20)**.

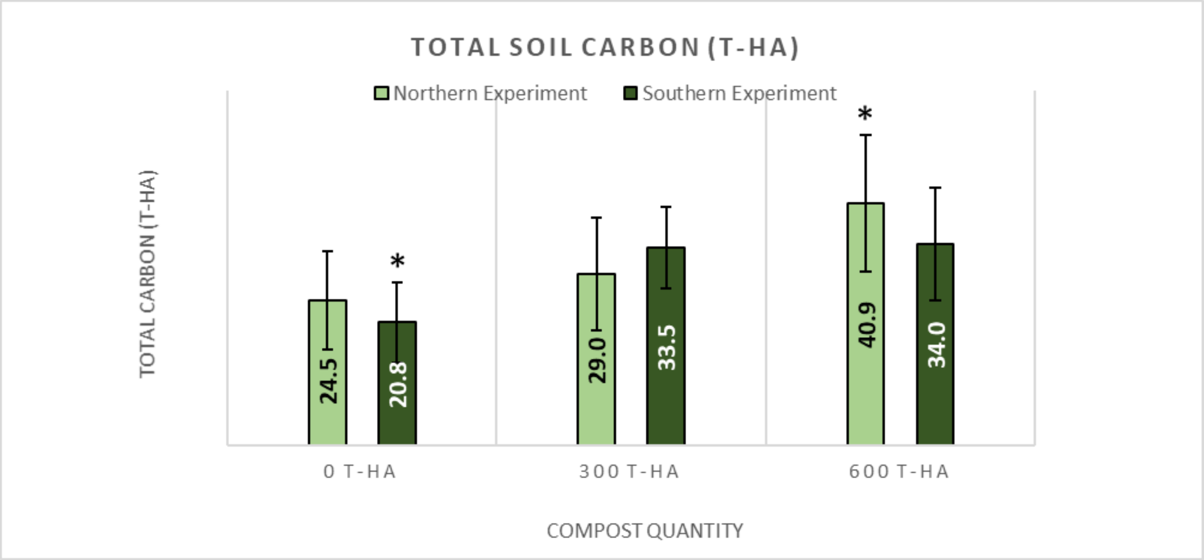


FIGURE 6.20 – *CGW impact on total soil carbon (t-ha^{-1}). Only one significant difference was found as indicated by asterisks (independent one-way ANOVA F (5, 42) = 2.547, p <0.05). At both sites compost amended plots contained more soil carbon than controls, however high variance meant the effect was not always significant. When compost quantity doubled at GOH-NORTH so did total soil carbon. However, at GOH-SOUTH carbon levelled out when CGW rate increased.*

Because CGW is considered an organic matter substitute and SOM contains substantial quantities of carbon, the relationship between soil organic matter and carbon was investigated further. Linear regression was used to predict how varying quantities of SOM might influence total carbon at Greenoakhill. At GOH-NORTH, soil organic matter was a significant predictor of total carbon: $R^2 = 0.602$, F (1, 22) = 33.34, $p < 0.001$; explaining 60% of the variance observed in soil carbon levels **(FIGURE 6.21)**.

Similarly, soil organic matter was also a significant predictor of total carbon at GOH-SOUTH: $R^2 = 0.206$, F (1, 22) = 5.709, $p < 0.03$, however on this occasion SOM quantity accounted for just 20% of the variance observed in soil carbon (**FIGURE 6.22**). Thus, SOM predicted far less variance in soil carbon at GOH-SOUTH, meaning the mechanisms responsible may differ between sites. Nevertheless, at both experiments CGW elevated SOM and carbon to levels that remain detectable 10 years after compost was applied, albeit to a far lesser degree at the southern site.

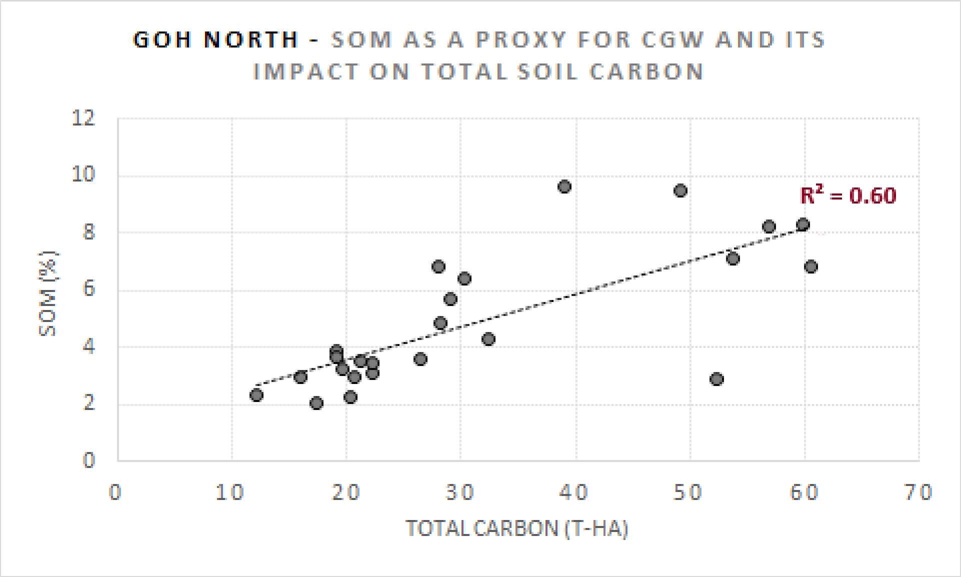


FIGURE 6.21 *– SOM as a predictor of soil carbon at GOH-NORTH. Linear regression shows the quantity of SOM present was a significant predictor of total carbon (t-ha): R² = 0.602, F (1, 22) = 33.34, p <0.001. Thus, SOM accounted for 60% of the variability observed at the Northern site. The following equation: Total Soil Carbon (t-ha⁻¹) = 0.959 + (0.765 x SOM%) provides a predictive model for this relationship.*

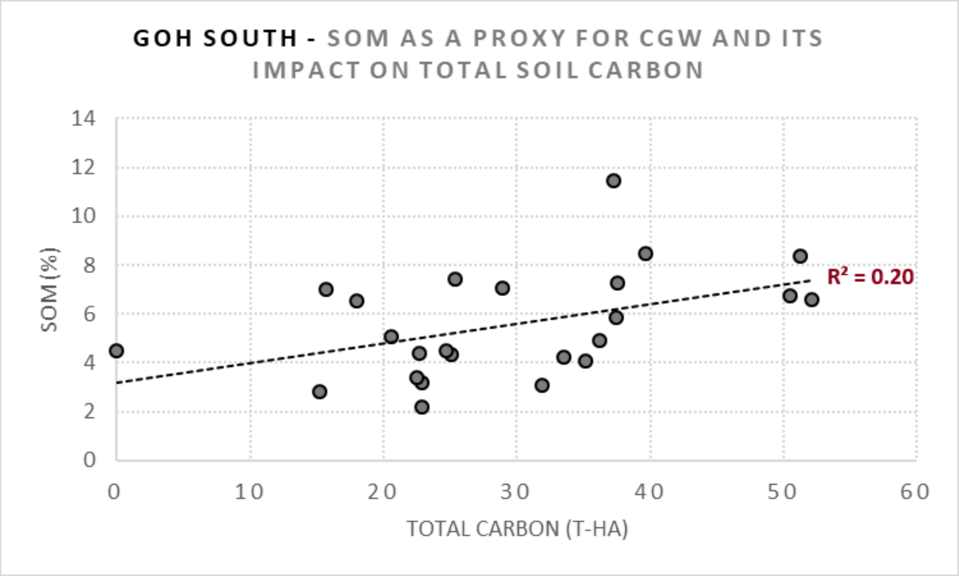


FIGURE 6.22 *– SOM as a predictor of soil carbon at GOH-SOUTH. Linear regression shows the quantity of SOM present was a significant predictor of total carbon (t-ha⁻¹): R² = 0.206, F (1, 22) = 5.709, p<.0.03. Thus, SOM accounted for 20% of the variability observed at the southern site. The following equation: Total Soil Carbon (t-ha⁻¹) = 15.163 + (2.571 x SOM%) provides a predictive model for this relationship.*

CGW is known to contain significant quantities of nitrogen, making it important for reclaimed soils where nitrogen capital is low. Thus, total nitrogen (t-ha^{-1}), and available nitrogen (mg kg^{-1}) were evaluated to understand CGW impacts on this vital nutrient. For total nitrogen (t-ha^{-1}), similar trends were observed at both sites. Independent one-way ANOVA found all compost treatments contained significantly more total-N than controls: F (5, 41) = 12.71, p <0.001. Indeed, at both sites soil total-N rose in tandem with compost rate, increasing by 0.5 – 1.3 t-ha^{-1} with each increase in compost rate. Indeed, CGW addition always increased soil total-N, although the difference between 300 and 600 t-ha^{-1} treatments was not significant. Nevertheless, a compost effect remained detectable after +10 years had elapsed (**FIGURE 6.23**).

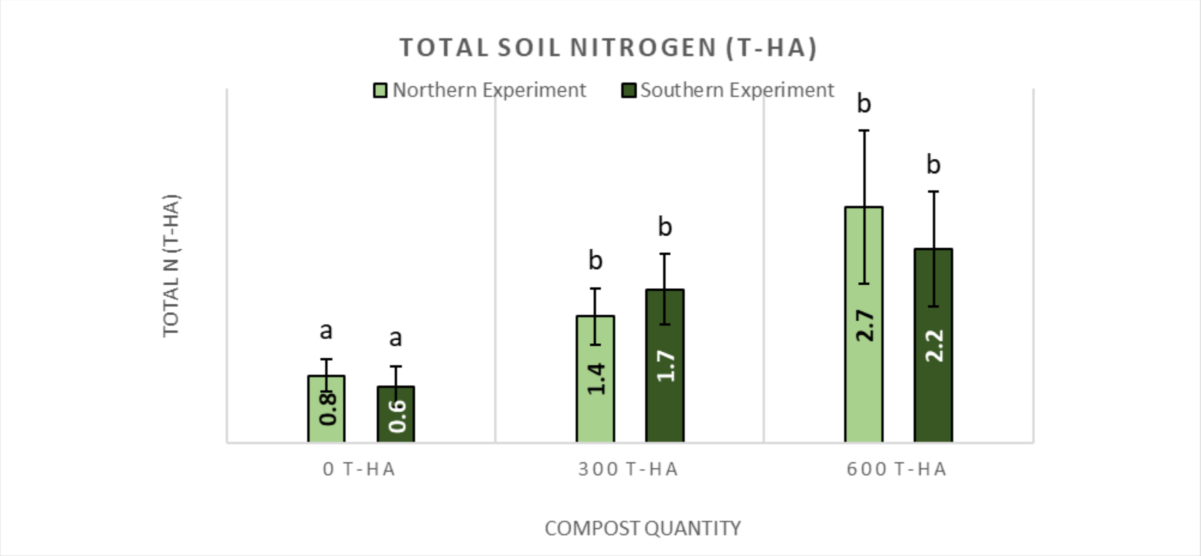


***FIGURE 6.23** – CGW impact on total soil nitrogen (t-ha^{-1}). A difference in letters indicates significant differences between treatments. At both sites, CGW significantly increased total soil nitrogen in 300 and 600 t-ha^{-1} treatments compared with controls (independent one-way ANOVA F (5, 41) = 12.71, p <0.001).*

Compost impact on available nitrogen (kg-ha^{-1}) appeared similar to those observed for total nitrogen. However, upon closer inspection a key difference was apparent. First, as with total-N, available-N rose incrementally when compost rate increased, resulting in significant differences between controls and compost treatments at both sites (Kruskal-Wallis non-parametric ANOVA, H (5) = 26.93, p <.001). At GOH-NORTH only the 600 t-ha^{-1} treatments contained significantly more soil available-N (p <0.01) than controls, whilst at GOH-SOUTH both the 300 and 600 t-ha^{-1}

treatments contained significantly more available-N (p <0.05) **(FIGURE 6.24)**. Yet, by examining nitrate (NO_3^-) and ammonium (NH_4^+) separately, further insight into nitrogen cycling can be obtained. Specifically, at GOH-SOUTH the 300 t-ha^{-1} treatments were dominated by ammonium (NH_4^+) whilst available nitrates (NO_3^-) remained low. Ammonium (NH_4^+) build-up suggests nitrification may be inhibited, resulting in lack of NO_3^- and thus foliar-N deficiency at GOH-SOUTH.

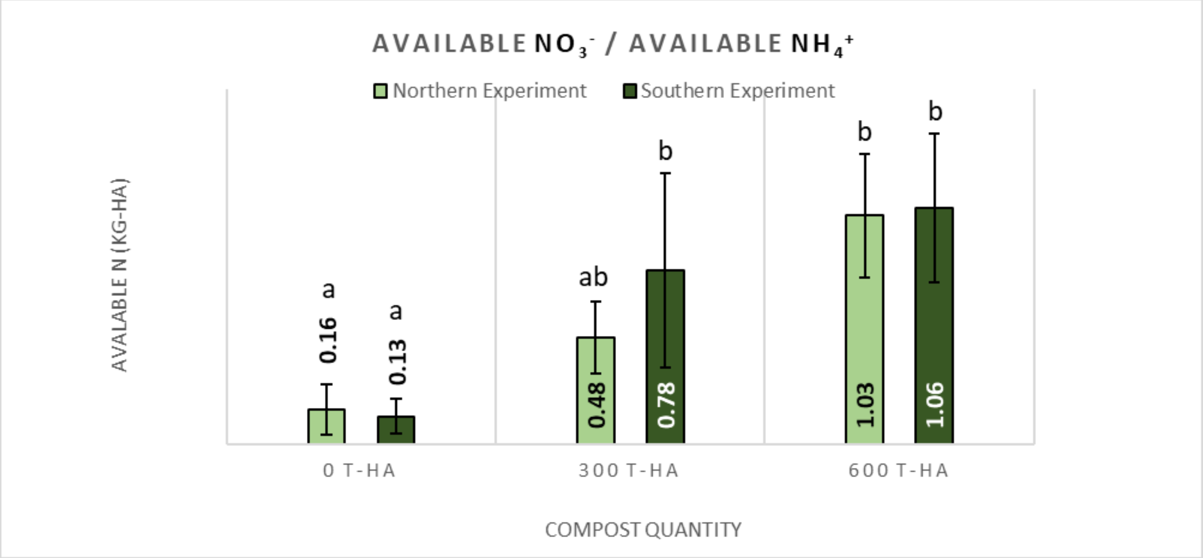


FIGURE 6.24 – CGW impact on available nitrogen (kg-ha^{-1}). A difference in letters indicates significant differences between treatments. CGW significantly increased available nitrogen at both sites compared with controls (Kruskal-Wallis non-parametric ANOVA H (5) = 26.93, p <.001). At GOH-NORTH, available-N was significantly higher in the 600 t-ha^{-1} treatments (p <.009). At GOH-SOUTH, available-N was significantly higher in both the 300 and 600 t-ha^{-1} treatments (p <0.045).

***FIGURE 6.25** – CGW impact on available nitrate (NO_3^- kg-ha^{-1}). Differences in letters indicate significant differences between treatments. CGW increased available NO_3^- significantly at both sites compared with controls (Kruskal-Wallis non-parametric ANOVA H (5) = 26.93, p <.001). The exception was GOH-SOUTH 300 t-ha^{-1} treatment where nitrate levels were similar to controls, with ammonium (NH_4^+) making up the bulk of available-N in this treatment. In all other amended plots, NO_3^- and NH_4^+ were present at a 1:1 ratio, suggesting reduced levels of nitrification in this treatment.*

6.8.4 Impact of compost addition on earthworm populations and soil development

Earthworms were sampled in April-May 2019, 10 years after the experiments were established. Stark contrasts in earthworm population establishment and development emerged between sites. At GOH-NORTH, nine earthworm species were identified within the experimental area, all at moderate to low levels. Mean densities for the nine species were: *Aporrectodea caliginosa* (47 m^{-2}), *Allolobophora chlorotica* (23 m^{-2}), *Aporrectodea longa* (19 m^{-2}), *Aporrectodea rosea* (16 m^{-2}), *Octolasion tyrtaeum* (16 m^{-2}), *Lumbricus rubellus* (3 m^{-2}), *Lumbricus castaneus* (1 m^{-2}), *Octolasion cyaneum* (0.4 m^{-2}), *Lumbricus terrestris* (0.4 m^{-2}) **(FIGURE 6.26)**. This gave an overall earthworm community density of 125 m^{-2}. In contrast, at GOH-SOUTH no earthworms were detected in any of the 24 pits sampled, suggesting viable populations failed to establish within the southern experiment.

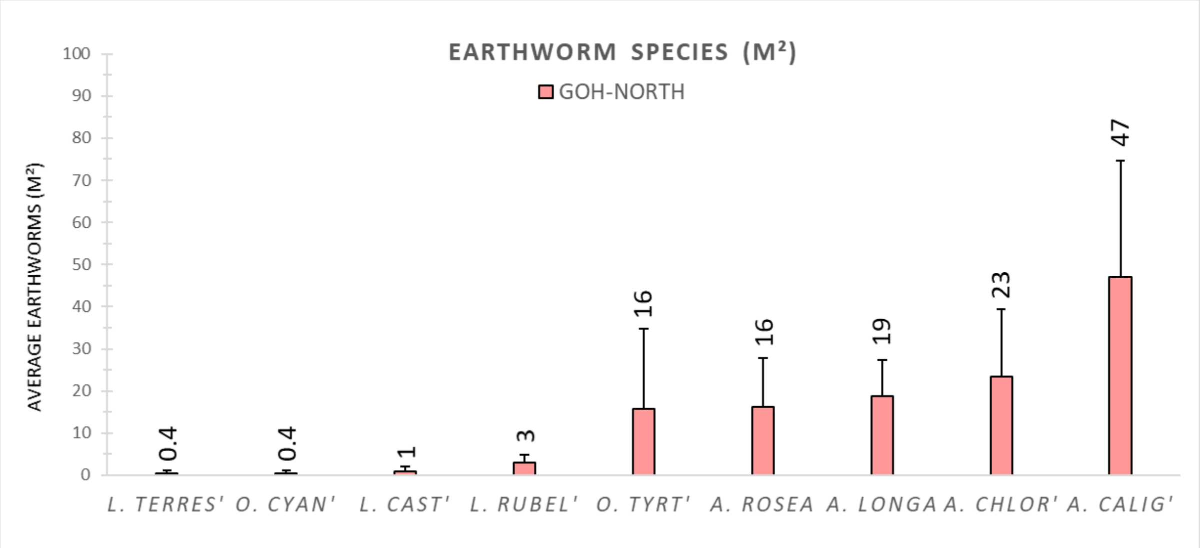


FIGURE 6.26 *– Earthworm abundance (m^{-2}) at GOH-NORTH following 10-years of site development. No significant differences were found between the densities of different earthworm species. However, the community appears to be dominated by species utilising endogeic strategies as their primary or secondary burrowing and feeding behaviour*

Regarding specifics of community structure, species which were abundant at GOH-NORTH employ an endogeic strategy as their primary or secondary burrowing and feeding behaviour (A. caliginosa, A. chlorotica, A. longa, A. rosea, O.tyrtaeum). Conversely, species of low abundance employed an epigeic / anecic strategy as their primary or secondary burrowing and feeding behaviour (L. rubellus, L. castaneus, L. terrestris). This suggests site conditions have evolved to favour a predominantly endogeic community. Indeed, characterizing the GOH-NORTH community according to conventional ecological groups, endogeic density (87 m^{-2}) was significantly higher than anecic (20 m^{-2}) and epigeic (20 m^{-2}) density combined (Kruskal-Wallis non-parametric ANOVA, H (2) = 17.15, $p < .001$) **(FIGURE 6.27)**. In summary, GOH-NORTH hosts a predominantly endogeic community, whilst at GOH-SOUTH a viable population had failed to establish. Differences in soil development might be expected due to the presence / absence of earthworms.

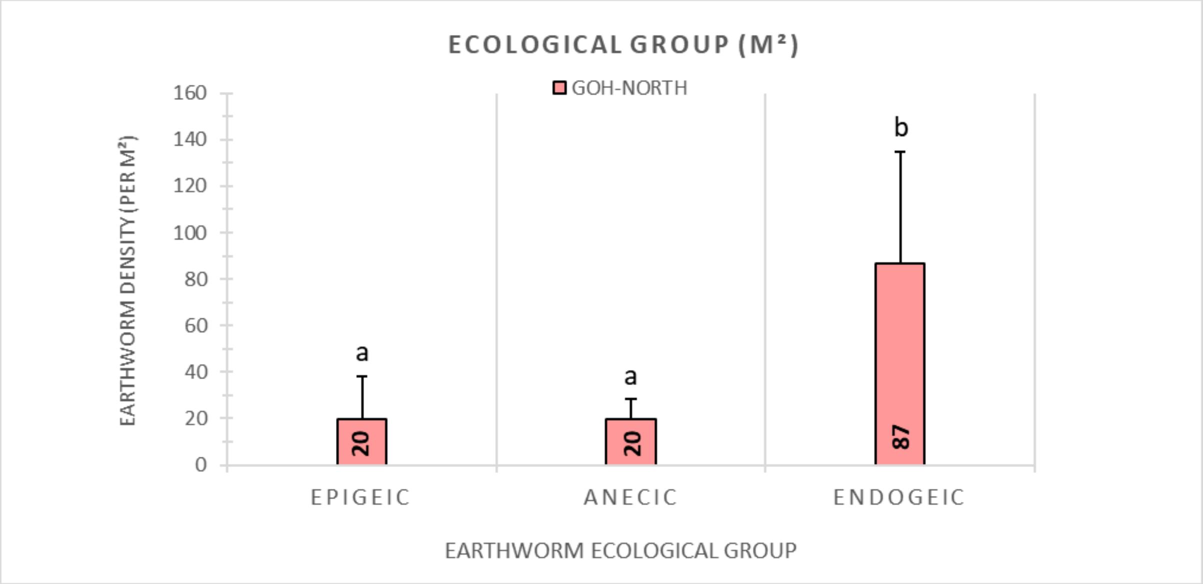


FIGURE 6.27 – Average abundance of earthworm ecological group (m^{-2}) at GOH-NORTH according to the three conventional classifications. A Kruskal-Wallis non-parametric ANOVA found significant differences in ecological group density [H (2) = 17.15, $p < .001$]. A post-hoc test with bonferroni correction found there were significantly more endogeic earthworms (87 m^{-2}) than anecic (20 m^{-2}) or epigeic (20 m^{-2}) ($p < 0.005$).

Regarding the treatment effect, compost quantity seemed to have a statistically significant impact on earthworm abundance (Kruskal-Wallis non-parametric ANOVA, H (8) = 22.08, $p < 0.005$). Yet this was not a true effect, and earthworm ecological group was considered a better descriptor of population dynamics. For example, Dunn's post-hoc test without bonferroni correction found endogeic density was significantly higher than epigeic *and* anecic density in *all* treatments and controls ($p < 0.05$). This simply means that endogeic species dominated the entire site and earthworm community, with densities similar in amended plots and controls **(FIGURE 6.28)**.

Consequently, compost quantity could not be the sole driver of earthworm community dynamics when sampling was conducted in Spring 2019.

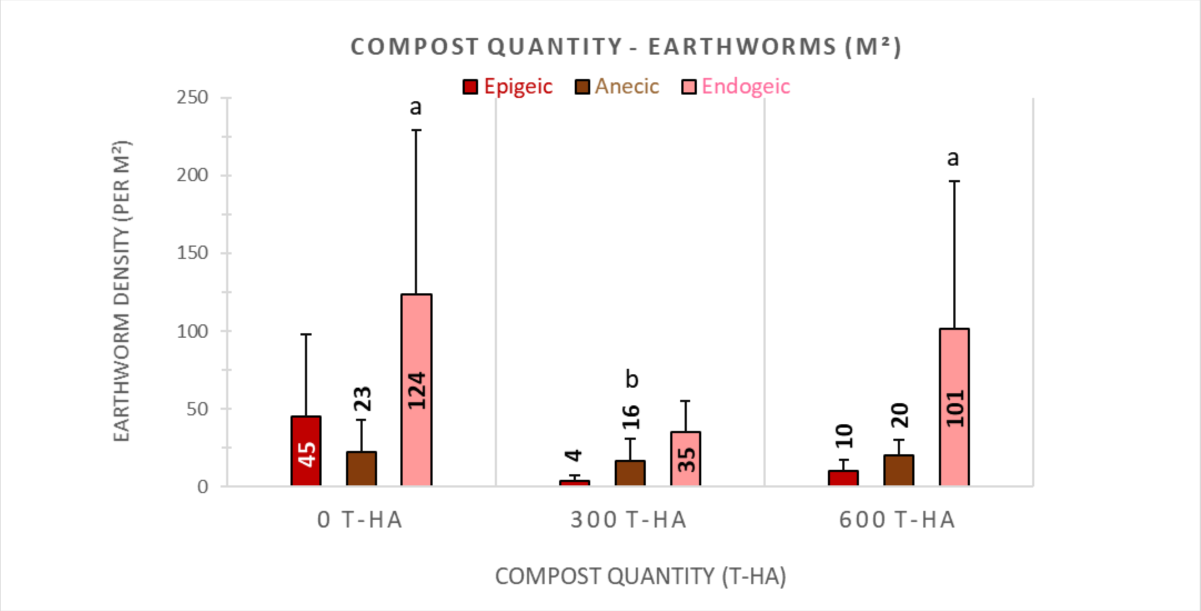


***FIGURE 6.28** – CGW impact on earthworm abundance (m^{-2}). Significant treatment effects were identified (Kruskal-Wallis non-parametric ANOVA, H (8) = 22.08, $p < 0.005$). Post-hoc tests with bonferroni corrections revealed two main effects. Namely, endogeic densities in control and 600 t-ha^{-1} treatments were significantly higher than epigeic density in the 300 t-ha^{-1} treatments ($p < 0.01$). In addition, epigeic density was significantly higher in controls compared with the 300 t-ha^{-1} treatments ($p < 0.05$).*

Indeed, sampling data revealed an unusually large number of earthworms had congregated in two adjacent plots located in BLOCK D at GOH-NORTH, namely, the control (350 endogeics per m^{-2}), and 600 t-ha^{-1} plots (255 endogeics per m^{-2}). No other plots contained such high densities suggesting endogeic dominance was concentrated in one area. Thus, high endogeic densities in controls and amended plots were based on an uneven distribution throughout the site. The following will now provide a graphical summary of compost impact on tree growth, soil development, and earthworm populations at Greenoakhill, using visual outputs produced by principal component analysis (PCA).

6.8.5 Overview of results

Principal component analysis (PCA) was used to evaluate all site data obtained in 2019. Graphical outputs from the PCA known as 'biplots' provided a visual representation of total site dynamics. The interplay between site conditions', compost, tree growth, and earthworms was assessed by examining the distribution of data points and vectors and comparing these between sites. For example, a clustering of data points indicates a greater association among variables. Vector length indicates effect size and strength. Whilst angles between vectors, indicate the nature of a relationship. For example, vectors following similar trajectories are closely associated, vectors at opposite ends of 180° angles have an 'opposing' or 'polar' correlation, whilst vectors at right angle generally have no relationship or correlation.

The PCA biplots in **FIGURE 6.30** and **FIGURE 6.29** reveal data separated according to site. GOH-NORTH and GOH-SOUTH diverged from one another at the 0-15 cm sampling depth. 'Height', 'DBH', 'Foliar N', 'Soil C', 'Soil N', 'SBD', and 'NO_3^-' were all higher at GOH-NORTH (earthworms present) causing them to associate with the northern site. Conversely, 'Porosity', 'Water', 'Foliar C:N' ratio, 'Foliar C' and 'NH_4^+' were all higher at GOH-SOUTH (no earthworms) causing these variables to associate with the southern site. This means Silver birch grew taller, had broader stems, and had higher foliar nitrogen at GOH-NORTH (earthworms present), with soils retaining more carbon, nitrogen, and available nitrates. Conversely, at GOH-SOUTH (no earthworms present) the opposite was found. Tree growth and foliar nitrogen were lower, and soils contained less carbon and nitrogen. Additionally, soils at GOH-SOUTH were higher in ammonium indicating differences in nutrient cycling between sites. Interestingly, although GOH-SOUTH soils were lower in carbon and nutrients, they were more porous with higher soil moisture.

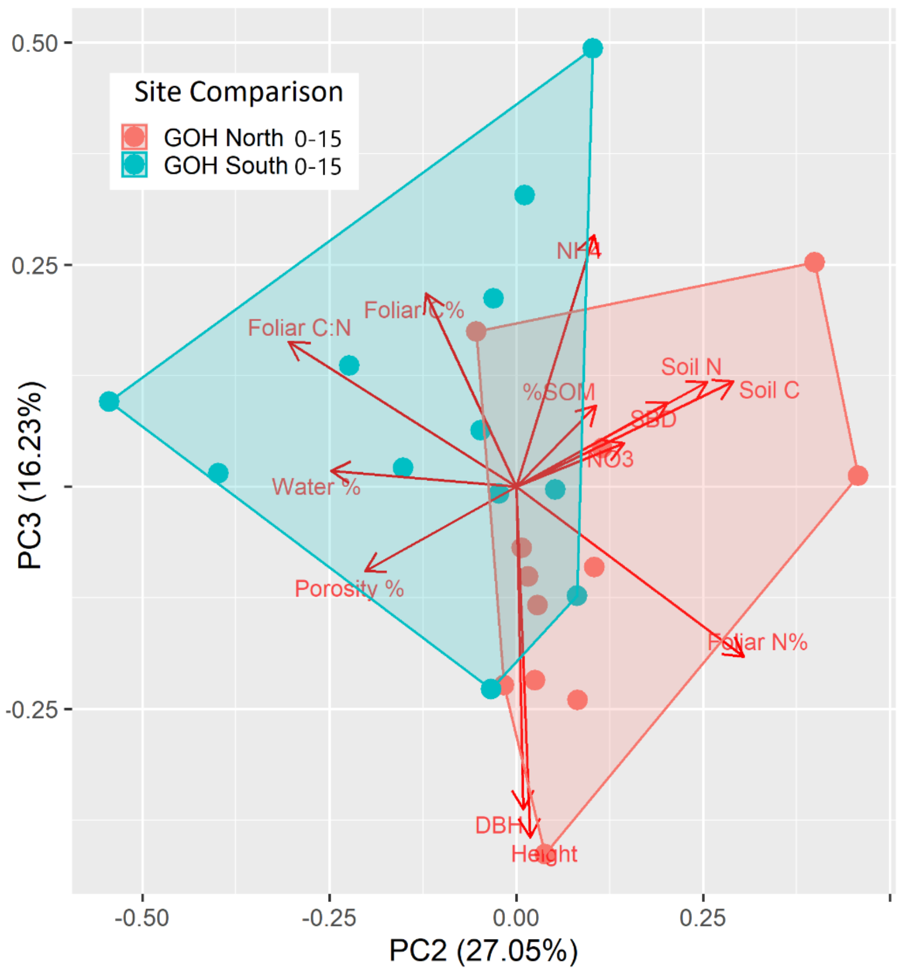


FIGURE 6.30 *– PCA examining relationships between soil physical, soil chemical, and tree growth related variables <u>0-15cm sampling depth</u>. Data clusters and separates according to site. This suggests developmental processes at GOH-NORTH and GOH-SOUTH have begun to diverge. Tree growth and soil chemical variables are primary drivers of the separation, which occurs along the secondary axis of PCA 2 versus PCA 3 and accounts for around 43% of the variation observed between sites.*

FIGURE 6.29 *– PCA examining relationships between soil physical, soil chemical, and tree growth related variables at <u>15-30 cm sampling depth</u>. Data still clusters and separates according to site. However, the effect is not as strong at the lower sampling depth. Although similar variables associate with each site there is more overlap among variables and data points. The separation occurs along the secondary axis of PCA 2 versus PCA 3 and accounts for around 38% of the variation observed.*

Regarding treatment effects, the PCA biplots depicted in **FIGURE 6.32** and **FIGURE 6.31** illustrate the impact of compost quantity on tree growth, soil development, and earthworm populations at GOH-NORTH. Data separates according to compost quantity, and different groups of variables correlate with specific quantities of compost. First, 'SBD' and 'Foliar C:N' were associated with controls, meaning wherever compost is absent, soil bulk density is high and foliar nitrogen is low. Second, 'Height', 'DBH', 'Porosity', and 'Water' are associated with the 300 t-ha^{-1} treatments, meaning tree growth was greatest, and soils were most porous and moist under the lower application rate. Finally, 'NO_3', 'NH_4^+', 'Soil N', 'Soil C', 'SOM', 'Foliar N', and all earthworm variables associate with the 600 t-ha^{-1} treatments. This means when the larger rate is used, soil total and available nitrogen, total carbon, organic matter, and foliar nitrogen are at their highest, whilst earthworm density and biomass correlate with these soil chemical effects. In summary, at GOH-NORTH, the 300 t-ha^{-1} treatments had the greatest effect on tree growth and soil physical parameters, whilst the 600 t-ha^{-1} treatments had the greatest effect on soil chemical and biological parameters.

The PCA biplots depicted in **FIGURE 6.33** and **FIGURE 6.34** illustrate the impact of compost quantity on tree growth and soil development at GOH-SOUTH. Earthworms were not present at the southern site hence do not appear in biplots. As before, data separates according to compost quantity, however different variables are associated with each treatment and effects are depth dependent. First, at both sampling depths controls were the same as the northern site, with bulk density high and foliar nitrogen low when compost is absent. However, when compost was added, opposite dynamics occurred compared with GOH-NORTH. For example, at GOH-SOUTH 0-15 cm, 'Soil C', 'Soil N', 'Foliar N', and 'NH_4^+' associate with the 300 t-ha^{-1} treatments. Conversely, 'Height', 'DBH', 'Porosity', 'Water', and 'NO_3^-' are associated with 600 t-ha^{-1}. This means the 300 t-ha^{-1} treatments had the greatest effect on soil chemical parameters, whilst 600 t-ha^{-1} treatments had the greatest effect on tree growth and soil physical parameters. Hence, the pattern observed at GOH-SOUTH is opposite to that observed at GOH-NORTH. Furthermore, unlike the northern site, treatment effects at GOH-SOUTH were depth dependent. For example, at 15-30 cm most parameters correlate with the 600 t-ha^{-1} rate of compost, suggesting the higher application rate had a strong influence deeper in the soil profile. In summary, CGW effects on tree growth and soil physical, chemical, and biological interactions were slightly different at each site. The reasons for this are now discussed, with site conditions and earthworm presence / absence thought to be responsible for many of the differences observed between sites.

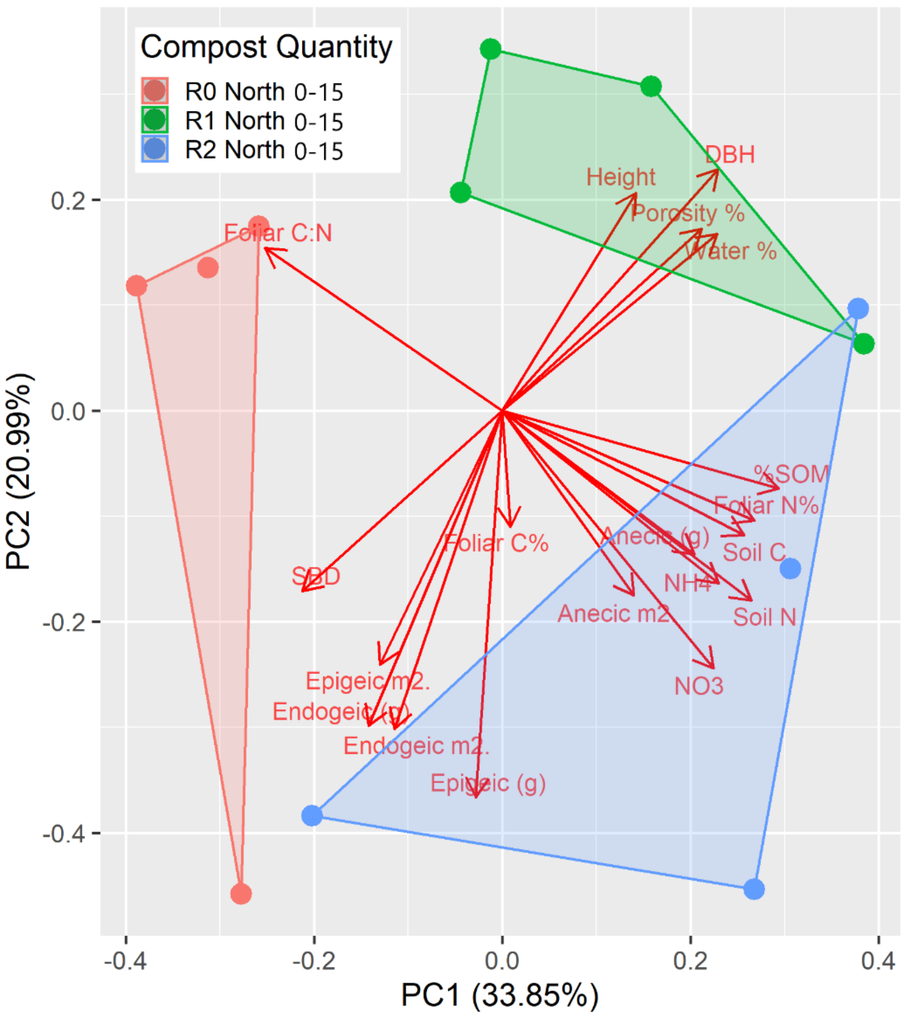


FIGURE 6.32 – PCA illustrating relationships between compost quantity, tree growth, earthworms, and soil physical and chemical parameters at GOH-NORTH 0-15 cm sampling depth. Data separates according to compost quantity along the primary axis of PCA 1 versus PCA 2, accounting for 55% of the variation observed. Controls (R0 = RED) have compacted soils and trees with low foliar-N. Tree growth is greatest in 300 t-ha^{-1} treatments (R1 = GREEN). Soil carbon, soil nitrogen, organic matter, and abundance of burrowing earthworms are greatest in the 600 t-ha^{-1} treatments (R2 = BLUE).

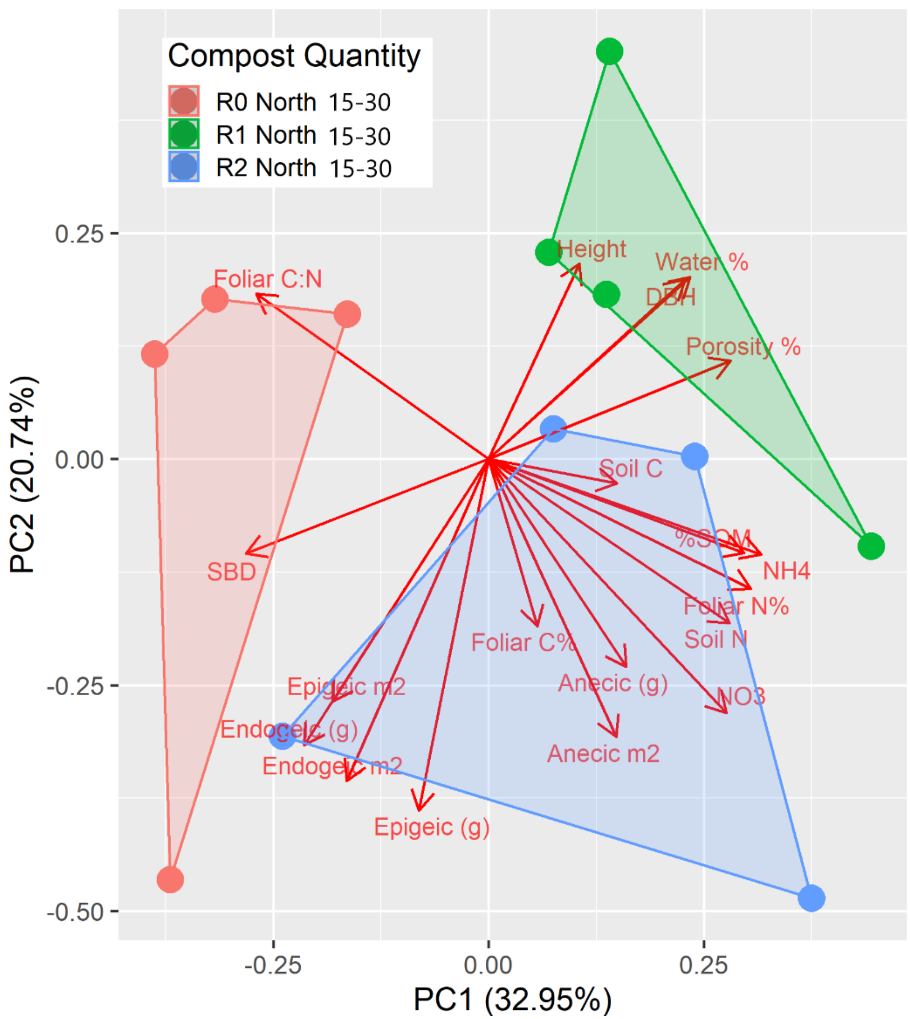


FIGURE 6.31 – PCA illustrating relationships between compost quantity, tree growth, earthworms, and soil physical and chemical parameters at GOH-NORTH 15–30 cm sampling depth. Data separates according to compost quantity along the primary axis of PCA 1 versus PCA 2, accounting for 54% of the variation observed in the dataset. Variables associate with the same treatments as the 0-15 cm sampling depth, suggesting uniformity of soil physical and chemical parameters throughout the depth sampled.

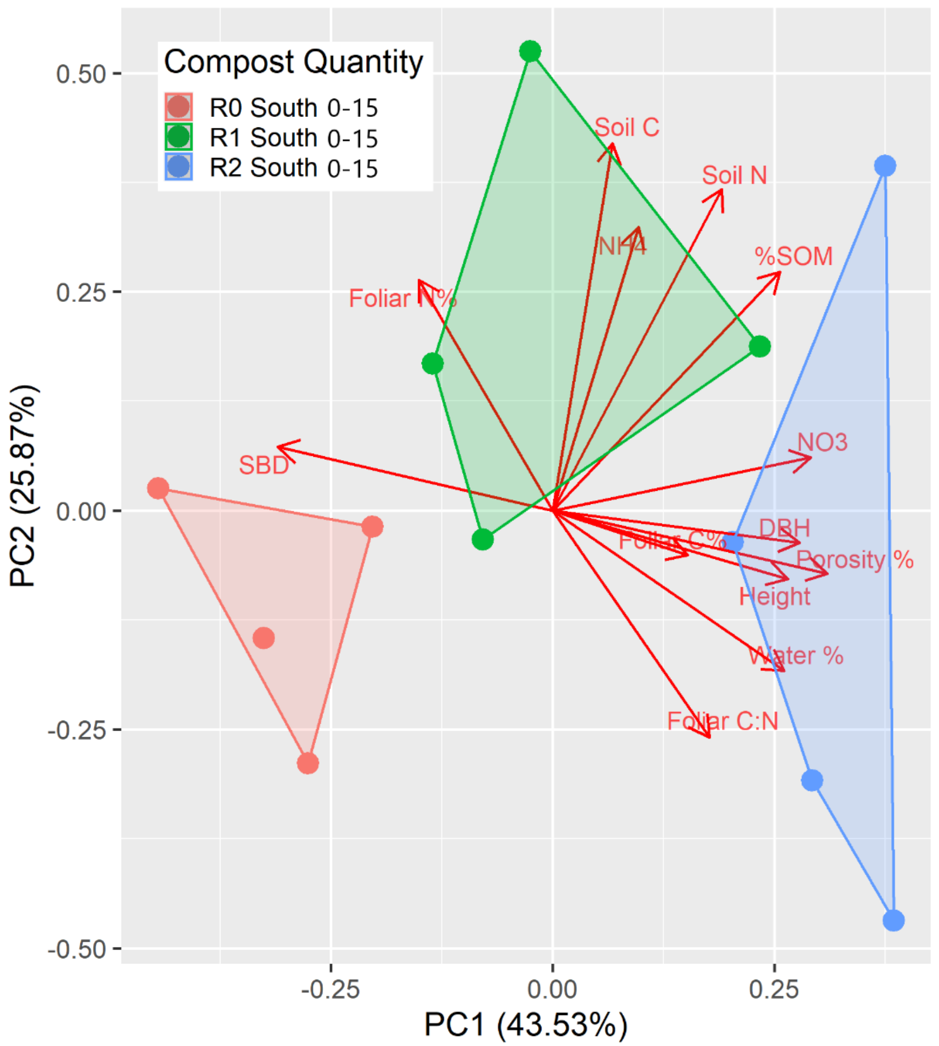


FIGURE 6.33 *– PCA illustrating relationships between compost quantity, tree growth, earthworms, and soil physical and chemical parameters at GOH-SOUTH 0-15 cm sampling depth. No earthworms were present at this site. Data separates according to compost quantity along the primary axis of PCA 1 versus PCA 2, accounting for 70% of the variation observed in the dataset. Again, controls (R0 = RED) have compacted soils and trees with low foliar-N. However, compared with GOH-NORTH compost treatments associate with different variables suggesting alternate dynamics are in operation.*

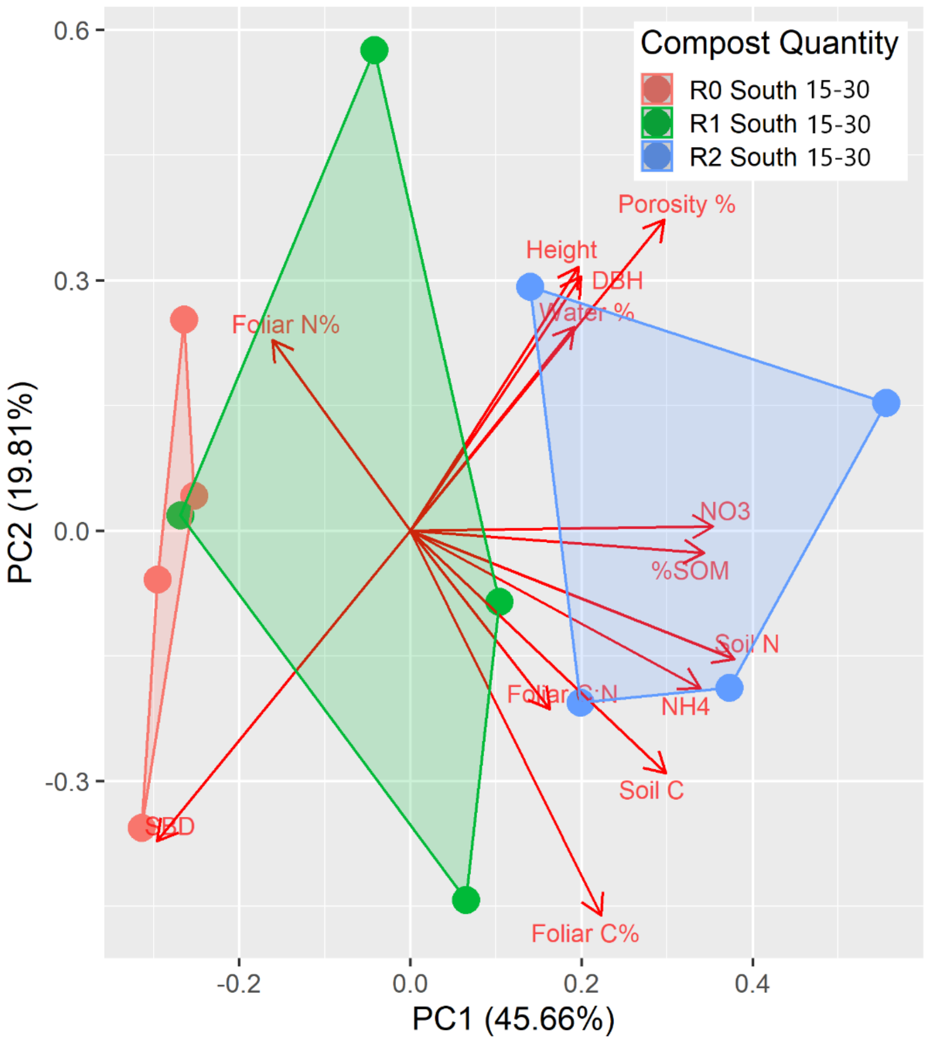


FIGURE 6.34 *– PCA illustrating relationships between compost quantity, tree growth, earthworms, and soil physical and chemical parameters at GOH-SOUTH 15-30 cm sampling depth. Data separates according to compost quantity along the primary axis of PCA 1 versus PCA 2, accounting for 67% of the variation observed in the dataset. At this deeper sampling depth most variables are oriented towards the 600 t-ha^{-1} treatments (R2 = BLUE), especially SOM and NO$_3^-$ suggesting both are in greater supply at this depth.*

6.9 Discussion

The experiments established at Greenoakhill provide rare insights into composted green waste's legacy effects. Compost was incorporated to improve native broadleaf performance at two adjacent landfills, and undoubtedly improved Silver birch growth. Indeed, wherever compost was incorporated soil resources increased; with organic matter, soil carbon, and soil nitrogen remaining higher in amended plots 10-years after application suggesting CGW has long lasting legacy effects. Yet the mechanisms behind CGW longer acting effects have to be inferred, by comparing quantitative data and qualitative observations made at each site. For example, at GOH-NORTH earthworms were present, surface litter was absent, and a predominantly endogeic community produced characteristic vermiform mull soil. In contrast, at GOH-SOUTH earthworms were absent, surface litter was abundant, fungal hyphae dominated the upper horizons, fewer macroaggregates were visible, and undecomposed CGW was occasionally found below the soil surface. This shows pedogenesis differed in earthworms absence, with organic matter decomposition and incorporation into the soil matrix greater in their presence. However, despite these longer acting soil-based effects, CGW impact on tree growth appeared short-lived. Tree growth was similar in 300 and 600 t-ha^{-1} treatments, suggesting a doubling of compost quantity does not provide additional short or long-term benefits to tree growth. Indeed, the versatility of Silver birch meant tree performance at Greenoakhill was always satisfactory, regardless of whether compost or earthworms were present. Indeed, when earthworms were absent at GOH-SOUTH compost still improved Silver birch growth, suggests other soil organisms were mediating tree growth and soil formation in earthworms' absence. Nevertheless, compost had the greatest impact on carbon storage when earthworms were present with 600 t-ha^{-1} composted green waste. This larger application supported earthworm metabolism, resulting in lower soil organic matter content, but statistically significant gains in soil carbon storage, outlining CGW and earthworms interactive effects. In summary, CGW impact on above and below-ground processes was positive, remaining detectable after 10-years and demonstrating compatibility with Silver birch and earthworms.

6.9.1 Impact of compost addition on tree survival, growth, and foliar nutrition

Composted green waste significantly improved tree growth at both sites, however trees at GOH-NORTH outperformed those at GOH-SOUTH even in controls. Superior growth between site controls suggests factors additional to compost influenced tree growth. Indeed, a combination of site microclimate and topography, soil forming materials, and earthworm interactions with vegetation appear responsible for superior tree growth at the northern site.

First, concerning the impact of compost on trees, wherever compost was incorporated growth always excelled. Yet, increasing compost quantity from 300 t-ha^{-1} to 600 t-ha^{-1} provided no additional benefit suggesting upper limits were reached at the lower rate. Yield charts published in Hynynen et al. (2009) reveal Silver birch growth at Greenoakhill was comparable to natural forest ecosystems. For example, during the first 10-years of growth in northern and western European climates, the maximum average height for Silver birch is 10 m, with a stem diameter of 8.7 cm (Hynynen et al., 2009). At GOH-NORTH, all trees approached this limit, with average height ranging from 8.6 to 9.3 m, stem diameter from 7.8 to 9.5 cm, being greatest in 300 t-ha^{-1} treatments and lowest in controls. Thus, compost increased height by 0.7 m, and stem diameter by 1.1 cm. These gains were similar to those observed at Ingrebourne Hill when Italian alder received 80 t-ha^{-1} PAS-100 CGW, suggesting composts tree growth promoting effect is mostly short acting.

In contrast, at GOH-SOUTH growth was slower and more moderate, but still competitive relative to Northern European yields (Hynynen et al., 2009). Average height was 6.6 to 7.8 m, with stem diameter from 5.9 to 7.9 cm, being tallest in the 600 t-ha^{-1} treatments and lowest in controls. This means compost increased height by 1.2 m, and stem diameter by 2.0 cm, which was almost double the improvement observed at GOH-NORTH. Hence, at GOH-SOUTH compost stimulated a greater increase in growth, but overall height was superior at GOH-NORTH.

Compost was not solely responsible for superior growth at GOH-NORTH, as evidenced by examining compost type, quantity, and application techniques. These were the same at both sites, producing soils with similar nitrogen profiles at either site (i.e. total nitrogen, available nitrogen, and C:N ratio). Yet despite these similarities, tree growth differed between sites, and Silver birch were foliar-N deficient at GOH-SOUTH. Interestingly, foliar-N deficiency at GOH-SOUTH has been a long-term problem, as evidenced by a site-wide survey conducted by the Forestry Commission in 2014 (Wall, 2014). This survey found newly established trees near the experiment were also foliar-N deficient, despite having received 750 t-ha^{-1} PAS-100 compost three years prior to the survey when the site was restored (Wall, 2014). Having received a substantial quantity of organic amendment, lack of compost could not be responsible for reduced tree growth and low foliar-N at GOH-SOUTH. Monitoring at 10-years can now reveal that several interlocking factors caused this deficiency, namely site microclimate and topography, soil manufacturing techniques, tree species selection, and the absence of earthworms, all of which are now discussed.

Regarding microclimate and topography, GOH-SOUTH was a challenging site. The experiment was situated at the crest of a hill and exposed on all sides, reducing tree growth rates. Conversely, at GOH-NORTH microclimate was favourable with the experiment resting in a slight depression and sheltered on all sides. This improved the performance of newly established Silver birch even in controls. Examining tree selection, the problem of foliar-N deficiency was species specific and fairly localised. For example, Wall (2014) found only Silver birch and European ash *(Fraxinus excelsior)* were foliar-N deficient, both being fast-growing, shorter lived species with higher nutrient demands. Conversely, Alder *(Alnus glutinosa)* which has N-fixing capabilities, and Larch *(Larix decidua)*, Oak *(Quercus spp.)*, and Scots pine *(Pinus sylvestris)* displayed no signs of foliar N-deficiency (Wall, 2014). These species are either slow growing, have longer lifespans, and / or have lower demand for nutrients (Da Ronch et al., 2016; Eaton et al., 2016; Houston Durrant et al., 2016). Hence, although soils at GOH-SOUTH contained sufficient nitrogen; Silver birch at this site could not access the quantities of nitrogen needed to fuel its rapid growth .

Regarding soil manufacturing techniques, Wall (2014) attributed the problem of inadequate nitrogen uptake in Silver birch to improper mechanical mixing of CGW in the heavy clay soil. Indeed, when incorporating compost into manufactured soil, improper mechanical mixing can result in uneven organic matter distribution throughout the soil profile (Moffat, 1996b; Bending et al., 1999; WRAP, 2009; WRAP, 2012). Consequently, soil organisms may have difficulty locating organic food resources because amendments are locked away out of reach. Yet crucially, in addition to inadequate mechanical mixing of soil by machinery, earthworms were absent from GOH-SOUTH, meaning a vital mechanism for soil formation and nutrient cycling was lost. Earthworm bioturbation intimately mixes organic and mineral substrates together. Or in other words, earthworm feeding, burrowing, and casting activities process and homogenise the soil substrate continually, accelerating organic matter mineralisation, increasing nitrogen availability, and stimulating nutrient turnover (Blouin et al., 2013; Liu et al., 2019; Medina-Sauza et al., 2019). Additionally, the biogenic aggregates earthworms create concentrate nutrients, acting as reservoirs of soil fertility (Van Groenigen et al., 2019). Earthworm activity has even been shown to promote growth of nitrifying bacteria and enhance nitrogen mineralisation (Araujo et al., 2004; Carrillo et al., 2011; Blouin et al., 2013). Thus, with earthworm bioturbation absent from GOH-SOUTH, soil processing and turnover was slower, resulting in less nitrogen mineralization to NO_3^-. Consequently, resources remained bound to clay and CGW, and fast growing trees were unable to access optimal quantities of mineralised nitrates (NO_3^-) (Cui and Song, 2007).

Yet despite foliar-N deficiency at GOH-SOUTH, Silver birch performance remained satisfactory even in controls. It may be that adjacent amended plots boosted tree growth, by leaching nutrients and attracting soil organisms into the vicinity. However, Silver birch success both with and without compost, highlights the primary importance of selecting tree species that are compatible with reclaimed site conditions. Site conditions have a primary, overarching effect on tree species' performance that can supersede CGW effects. The adaptability of Silver birch justifies its widespread use in a variety of reclamation projects, and its large dispersal range make it ideal for natural colonization (Moffat and McNeill, 1994; Moffat and Laing, 2003; Frouz et al., 2009; Beck et al., 2016; Józefowska et al., 2017). Nevertheless, it is important to remember compost has wider impacts that extend beyond tree performance and into the soil environment. These are explored in the following sections and linked to tree growth and nitrogen dynamics already discussed.

6.9.2 Impact of compost addition on soil nitrogen, organic matter, and carbon storage

CGW is composed of bio-organic residues which living organisms utilise to support their metabolic needs. Thus, by providing an influx of organic residues compost was expected to promote soil formation and increase nutrient availability. At both experiments, CGW lowered soil C:N ratios, and increased organic matter, total carbon, total nitrogen, available nitrogen, and soil moisture. After 10-years these parameters remained elevated, indicating compost had belowground legacy effects. CGW was particularly effective at raising nitrogen capital, providing more than the recommended minimum of 1000 kg-ha^{-1} wherever applied (Bradshaw, 1983; Bradshaw, 1997). Yet only the 600 t-ha^{-1} treatments reached the 2000 kg-ha^{-1} total nitrogen recommended for cool temperate systems (Bradshaw, 2000). CGW ability to create nitrogen rich soils is beneficial, yet in certain contexts may be undesirable as some tree species prefer soils with less nutrients. Large influxes of compost can also contaminate watercourses, which is why nitrogen vulnerable zones (NVZ) have legal limits of 500 kg Total N ha^{-1} per year. When CGW was first incorporated at Greenoakhill in 2008, the 300 t-ha^{-1} treatments contained 2500 kg Total N-ha^{-1}, whilst the 600 t-ha^{-1} treatments contained 5000 kg Total N-ha^{-1}, considerably more than the legal limit for a potential NVZ. By 2019 only half of this nitrogen remained, but it remains unclear how much was lost annually and how much was transferred to receptors (Wall, 2014). Still, higher application rates risk contamination of watercourses, and 600 t-ha^{-1} provided no additional benefit to tree growth. Consequently, the higher rate of compost can only be justified based on its ability to increase carbon, which paradoxically occurred only in the 600 t-ha^{-1} treatments when earthworms were present. Earthworms' failure to colonise the experimental area means a 600 t- ha^{-1} treatment would have remained problematic if GOH-SOUTH was an NVZ. Consequently, compost quantity must be carefully matched to tree species, site conditions, soil biology, and project requirements, and should be considered as one of several integrated reclamation interventions.

Indeed, by adjusting CGW quantity different 'soil resource environments' can be created. The 600 t-ha^{-1} rate produced a 'resource rich' soil, whilst the 300 t-ha^{-1} rate produced 'resource moderate' soil. Data for soil organic matter and nitrogen cycling at Greenoakhill outline this, with below-ground organisms key considerations when selecting a given strategy. For example, when earthworms were present at GOH-NORTH, key differences in soil formation and compost utilisation were observed, centring on more rapid turnover of SOM and nitrogen in earthworms' presence. According to Haynes et al. (2015), when CGW mineralisation and nitrification are restricted, NH_4^+ accumulates and NO_3^- availability remains low. This exact phenomenon occurred at GOH-SOUTH, evidenced by ammonium build-up, foliar-N deficiency, and slower tree growth in earthworms' absence. This problem occurred even though soils at GOH-SOUTH retained more SOM and had similar C:N ratios and total nitrogen to GOH-NORTH, where earthworms remained present. This suggests compost provided sufficient quantities of SOM, carbon, and nitrogen at both sites, however with earthworms absent CGW remains underutilised with fewer resources available to plants.

Further evidence for lack of compost utilisation was found when conducting soil and earthworm sampling at GOH-SOUTH. Surprisingly, in several amended plots, undecomposed intact compost was excavated from beneath newly formed surface horizons. This indicated a lack of decomposition and soil turnover in earthworms' absence, a view supported by principal component analysis. PCA data for GOH-SOUTH 15-30 cm sampling depth, shows a correlation between SOM and the 600 t-ha^{-1} treatments. This points towards the presence of undecomposed compost lower down the profile, meaning that when a higher rate of compost was incorporated to 0.5 m, an absence of earthworm bioturbation allowed some CGW to persist. Additionally, at GOH-SOUTH the 600 t-ha^{-1} treatments also had the lowest bulk density (0.67 g m^{-3}), and highest soil moisture (20.0 %) of all soils sampled. Again, this could be explained by the persistence of undecomposed compost below the soil surface, increasing moisture retention and lowering BD. Tellingly, this pattern was not apparent at GOH-NORTH where 600 t-ha^{-1} treatments contained significantly less moisture. Consequently, both quantitative and observational data corroborate the notion of slower decomposition and nutrient cycling in earthworms' absence. By lowering CGW supply from the outset, a more resource 'moderate' soil environment can thus be created, albeit with less opportunity for gains in carbon storage.

Indeed, soil carbon data from Greenoakhill supports the perspective that earthworms are suited to high turnover 'resource rich' systems. For example, with earthworms present at GOH-NORTH, the 600 t-ha^{-1} applications produced the greatest and only statistically significant increase in total carbon (40.9 t-ha^{-1}). Likely, CGW was metabolically processed by earthworms and associated micro-organisms, with the products of this metabolic activity becoming stabilised in biogenic structures (Liang et al., 2017; Liu et al., 2019; Angst et al., 2021). Indeed, when earthworms were present SOM content was generally lower, but total carbon was generally higher. This was most prevalent in the 600 t-ha^{-1} treatments, suggesting the higher rate of compost facilitated earthworm mediated carbon storage. Regression analysis further supports this notion. When earthworms were present at GOH-NORTH, SOM content explained 60% of variance in total carbon. In contrast, with earthworms absent from GOH-SOUTH, SOM explained only 20% of total carbon variance.

The correlation between SOM content and total soil carbon at GOH-NORTH strongly suggests earthworms' involvement. Their ability to transfer organic inputs into physically stabilised fractions is well documented and has been demonstrated in post-mining soils (Scullion and Malik, 2000; Frouz et al., 2013; Deeb et al., 2017). The 600 t-ha^{-1} treatments at GOH-NORTH appeared to produce a high input 'resource rich' system, with greater compost availability supporting the metabolic demands of earthworms and their associated biota, enhancing carbon storage. Earthworm effects on soil formation beyond carbon storage are now discussed.

6.9.3 Impact of compost addition on earthworm populations and soil development

Divergent patterns of soil formation were observed at GOH-NORTH and GOH-SOUTH, primarily because earthworms failed to colonize the southern site. Establishment failure was related to topography and surrounding land-use, which impeded natural colonization into GOH-SOUTH. In contrast, earthworms impact at GOH-NORTH was profound, shaping soil pedogenesis by mixing organic inputs into the mineral fraction. Several inter-locking factors influenced earthworm pedogenesis, namely, site and soil conditions, ground vegetation, trees, and compost.

Firstly, at GOH-NORTH a predominantly geophageous community emerged, strongly influencing soil pedogenesis. All dominant and co-dominant species were capable of horizontal burrowing, with *A. caliginosa* (47 m^{-2}), *A. rosea* (16 m^{-2}), and *O. tyrtaeum* (16 m^{-2}) all endogeic, whilst *A. longa* (19 m^{-2}) and *A. chlorotica* (23 m^{-2}) are endo-anecic and intermediate respectively (Capowiez et al., 2015; Bottinelli et al., 2020). In contrast, the least abundant earthworms were epigeic or surface feeding, with *L. castaneus* (1 m^{-2}), *L. rubellus* (3 m^{-2})*,* and *L. terrestris* (0.4 m^{-2}), classified as epigeic, endo-epigeic, and epi-anecic respectively (Bottinelli et al., 2020).

At GOH-NORTH, soil and vegetation development reflected the activities of the dominant species. Positive feedback emerged between grasses, Silver birch, and earthworms, with bioturbation and surface litter removal supporting grassland expansion. In turn, dense grass roots supported organo-mineral mull-soil formation **(FIGURE 6.35, FIGURE 6.36)**. Indeed, according to Natural England, soil feeding earthworms favour grasslands with neutral to base-rich soil, with A. caliginosa, A. chlorotica, A. longa, and A. rosea particularly abundant in such habitat (NE, 2014). O. tyrtaeum similarly thrives in neutral to base-rich woodland (NE, 2014), and has been shown to increase grass and herb seedling emergence in the deciduous forest understory (Eisenhauer et al., 2009b). In line with these studies, earthworm dynamics at Greenoakhill promoted development of non-acidic grassland and aggregated mull soil (Scullion and Malik, 2000; Marashi and Scullion, 2003; Eisenhauer and Scheu, 2008; Eijsackers, 2011; Birkhofer et al., 2012; Rutgers et al., 2016).

To attract migrating earthworms, Eijsackers (2010) highlights the importance of initial habitat quality with soil pH, moisture, depth, temperature, and organic matter availability influencing species arrivals. CGW influence on these properties may have encouraged geophageous species arrivals, promoting emergence of vegetation and soils presently observed. For example, at GOH-NORTH incorporation of 600 t-ha^{-1} compost to 0.5 m depth increased organic matter content, soil moisture, and pH, thus attracting geophageous species. Indeed, numerous studies highlight CGW ability to maintain neutral to basic pH for multiple years following its incorporation into landfill soil (Beesley, 2012; Wall, 2014; Wall, 2016; Deeb et al., 2017; Ashwood et al., 2018). Therefore, plants that favour such conditions may be compatible with geophageous earthworms and CGW.

FIGURE 6.35 – *Silver birch in BLOCK A and BLOCK B at GOH-NORTH. The canopy is light and only 6 trees deep, allowing sunlight penetration to ground level. Grasses dominate ground vegetation, and surface litter is completely absent.*

FIGURE 6.36 – 30 x 30 x 30 cm block of soil taken from a 600 t-ha^{-1} plot at GOH-NORTH. This sample was homogenous throughout the profile, composed almost completely of macro-aggregates with no obvious remnants of PAS-100 CGW.

Knowing when geophageous earthworms arrived at GOH-NORTH and under what circumstances can improve understanding of landfill pedogenesis. Certain authors document successional patterns wherein deeper burrowing species arrive after surface feeding epigeic earthworms (Eijsackers, 2011; Frouz et al., 2001; 2013). Indeed, pedogenesis often begins with epigeics such as *L. rubellus* and *L. castaneus,* who digest surface organic material and create superficial burrows, producing a thin surface O-horizon. This ‘primes’ the soil habitat for endogeic arrivals. Endogeics then use the newly created organic platform to incorporate more organic material into aggregates thus increasing soil mixing and A-horizon depth (Frouz et al., 2001; Eijsackers, 2010; 2011). However, according to Eijsackers (2010), Caro et al. (2013), and Chatelain and Mathieu (2017), ecological group is not *the* most important factor governing colonization. Rather, the primary factor is a species' general adaptability to a wide range of soil conditions. Hence, the now residual epigeic population of GOH-NORTH may have been abundant previously but has now been superseded by deeper burrowing earthworms. Alternatively, it's possible several species arrived together based on shared tolerance and adaptability to early soil conditions improved by CGW. Potential candidates for simultaneous species arrivals include *L. rubellus* (endo-epigeic)*, L. castaneus* (epigeic)*, A. caliginosa* (endogeic)*,* and *A. chlorotica* (intermediate) (Eijsackers, 2010; Bottinelli et al., 2020). Temporal studies would be needed to verify the true sequence of arrival.

Regardless, over a 10-year period endogeic behaviour came to dominate GOH-NORTH, with epigeic species and O-horizons declining as a result. Previous research demonstrates endogeic feeding plays an important role in removing surface litter from GOH-NORTH. For example, laboratory studies by Rajapaksha et al. (2013) show *A. caliginosa, A. chlorotica,* and *A. longa* preferentially feed on Silver birch leaves. Similarly, in reclaimed mine soils, Silver birch foliage is favoured by resident earthworms and associated micro-organisms, increasing soil carbon and nitrogen stocks (Józefowska et al., 2016; Józefowska et al., 2017). At GOH-NORTH carbon and nitrogen stocks increased in the 600 t-ha^{-1} treatments where Silver birch growth and CGW quantity was greatest. Research from reclaimed and agricultural soils suggests earthworm interactions with grasses and bacterial metabolism can help achieve this increase (Liu et al., 2019; Józefowska et al., 2017) **(FIGURE 6.35)**. Litter removals by earthworms may have facilitated grassland expansion, allowing light to reach the forest floor increasing below-ground carbon inputs. Geophageous earthworms used the abundance of inputs provided by trees, ground vegetation, and CGW to develop highly aggregated mull soil observed at GOH-NORTH. Deeb et al. (2017) demonstrated this effect experimentally using lab-scale mesocosms. *A. caliginosa* and ryegrass (*Lolium perenne*) were added to treatments alone and together with varying quantities of CGW (0, 10, 20, 30, 50% soil volume). The combination of *A. caliginosa,* ryegrass, and 30% compost promoted the greatest levels of carbon storage and aggregation. The 600 t-ha^{-1} treatments at GOH-NORTH produced a similar finding, albeit at field scale with inputs from Silver birch. In contrast, GOH-SOUTH lacked an earthworm population and pedogenesis clearly differed. Rather than being covered with grasses, GOH-SOUTH soils were covered with intact or partially decomposed litter from previous seasons (**FIGURE 6.37**). Soils also contained fewer aggregates and were not uniformly structured, containing angular clods and clumps of un-mixed soil forming material **(FIGURE 6.38)**. Indeed, contrary to bulk density readings soils at GOH-SOUTH were compacted making digging and soil sampling difficult. Intact deposits of CGW were found in amended plots suggesting CGW was under utilised. Evidently earthworms absence resulted in a lack of soil mixing and development. This is why Silver birch at GOH-SOUTH were foliar N-deficient, particularly in the 600 t-ha^{-1} where compost quantity was greatest but without earthworms was under-utilised.

FIGURE 6.37 – *Silver birch in BLOCK C at GOH-SOUTH. Unlike the northern site, leaf litter from previous seasons remains on the surface, covering the forest floor.*

FIGURE 6.38 – *30 x 30 x 30 cm block of soil taken from a 600 t-ha*$^{-1}$ *plot at GOH-SOUTH. Unlike the northern site, soils were not uniformly structured, and contained larger, angular, more cohesive peds. Soils in control and 300 t-ha*$^{-1}$ *treatments were even less well formed. This is indicative of poor mixing in the absence of macro-faunal bioturbation.*

Overall, the organic horizons that develop in any ecosystem, in terms of size, structure, and state of decomposition, indicate which plant and soil biological communities synthesised them (Ponge et al., 2010; Zanella et al., 2011). Soils at GOH-NORTH were evidently mull-soils produced by earthworm processing, whereas the O-horizons at GOH-SOUTH and lower aggregation suggest limited faunal processing and bioturbation (Ponge et al., 2010; Zanella et al., 2011). Assigning a classification to GOH-SOUTH humus layers is difficult and nuanced. First because earthworms were the only soil biological community sampled, hence the effects of other organisms could not be evaluated. Second, soils were not formally classified 'in field', meaning post-hoc classification is far less accurate. Third, no formal system of classification exists for manufactured soils, as they are young, constructed using novel recycled materials, and have undergone limited biological, geological, and climatic processing. Nevertheless, using the humus-forms provided in Ponge et al. (2010) and Zanella et al. (2011) alongside site observations, the biologically active upper layers of

GOH-SOUTH fall somewhere broadly between ‘moder’ and ‘mor’. This is due to the existence of different O-horizons, namely an undecomposed OL ‘litter’ horizon and a ‘humified’ OH horizon, both of which indicate arthropod and enchytraeid activity. Extensive fungal hyphae were also present in the O and A-horizons, suggesting fungi played a more prominent role in soil formation at GOH-SOUTH (Ponge et al., 2010; Zanella et al., 2011). These organisms are associated with slower organic matter turnover and nutrient cycling processes (Wardle et al., 2004; Kardol and Wardle, 2010), further explaining poor inter-mixing of organic and mineral fractions, incomplete CGW decomposition, and foliar-N deficiency at GOH-SOUTH. Acceptable tree growth at GOH-SOUTH shows some soil organisms were clearly present. However, with earthworms absent, soil formation, nutrient cycling, and carbon storage evidently differed. The reasons why earthworms failed to establish at GOH-SOUTH are now discussed.

Earthworms failed to establish at GOH-SOUTH due to site geography and surrounding land-use. GOH-SOUTH rests atop a steeply banked hill, surrounded by compacted stony man-made soils (**FIGURE 6.11**). The site is cut-off and disconnected along three of its four site boundaries by motorways, rivers, and poor-quality soils. For example, the M74 motorway straddles the sites northern boundary and hosts a constant stream of traffic (**FIGURE 6.9**), whilst The River Clyde snakes along the southern boundary separating GOH-SOUTH from adjacent land. Motorways and Rivers are formidable obstacles for earthworms and those in question stretch far beyond the Greenoakhill site, This would prevent rapid colonisation from adjacent land and farther afield. Similarly, immediately east of GOH-SOUTH experiment is an aggregate processing facility and sewage works. These are situated on concrete and compacted land, completing an array of inhospitable land-uses and geographical features surrounding the southern experiment. These restrict or entirely prevent earthworm movement above or belowground. Realistically, grassy fields to the experiments’ west are the most viable resource for colonisation. However, earthworms must traverse 300 – 350 m of newly manufactured soil to reach the southern experiment. To complete this journey within 10-years is more than double the maximum dispersal rate recorded for various earthworm species, currently thought to be 10 – 14 m^{-1} per year over arable and grassland (Butt, 1999; Eijsackers, 2011). Earthworms' failure to establish at GOH-SOUTH suggests the site may benefit from active inoculation, making it an interesting test site for post reclamation inoculation and the use of different inoculation techniques (Butt, 1999). Further, simultaneous tracking of changes in soil formation, carbon storage, and above-ground vegetation induced by inoculated earthworms would mean the benefits and trade-offs of earthworm introductions to established reclaimed woodlands could be explored. Simply sampling the wider site and ascertaining the number and location of current earthworm populations would allow earthworm migration patterns and progress at GOH-SOUTH to be more fully understood. Whether existing populations require supplementation via means of inoculation could then be evaluated more precisely.

CHAPTER 7. ORGANIC AMENDMENTS AROUND THE HUMBER ESTUARY: The impact of composted green waste at two separate landfills, 20 years after application

7.1 Background

In 1997 'Enventure Northern Ltd' commissioned a project to evaluate whether recycled composts could improve the quality of restored landfill. Project funding was obtained through credits from the 'Landfill Tax', a levy introduced in the 1990's to discourage landfill use and promote waste recycling. During the 1990's plant biomass from urban gardens, greenspaces, and highways accounted for a significant proportion of waste sent to landfill. However, by converting this biomass to compost significant quantities could be diverted from landfill, provided suitable end-uses for this so called 'composted green waste' (CGW) could be found. Accordingly, Forest Research (FR) were asked to investigate whether CGW could improve establishment of woodland on restored landfill sites. Consequently, in 1997/98 FR designed and ran field experiments at three different landfills to measure composts impact on tree performance over four years (Foot et al., 2003). Results suggested CGW could improve tree growth when appropriate quantities (250 t-ha^{-1}) were incorporated throughout the profile (0.6 m^{-1} depth) of a restored landfill soil. Further, when applied at shallow depths (0.1 m^{-1}) CGW significantly increased growth of ground vegetation leading FR to conclude this compost product was suitable for general greening of degraded sites. Unfortunately, the experiments' short observation period and failure to analyse soils meant results were inconclusive (Hislop and Harding, 1999; Foot et al., 2003). A subsequent attempt to extend monitoring from four to six years was thwarted by lack of funding, therefore composts long-term impacts remained uncertain. To better understand composts legacy effects the present study re-visited two of these experiments and evaluated tree growth and soil development after 20-years. The following chapter presents background information on the experiments, detailed results from the monitoring exercise, and discussion of the findings obtained after 20-years of site development.

7.1.1 Winterton

Winterton landfill is an 84-ha site located 6 km north of Scunthorpe, North Lincolnshire, England (Nat Grid Ref: SE 91427 20056)**.** It is one of three landfills around the Humber estuary hosting composted green waste (CGW) related field experiments **(FIGURE 7.1)**. Between 1860 and 1980 the Winterton area was one of the UK's largest producers of iron ore and steel with the landfill itself originally quarried for iron ore. By the 1970's production had dwindled hence in the 1980's the site was converted to landfill (Symes, 1987; Hislop and Harding, 1999). Landfill activities expanded progressively throughout Winterton, beginning at the northern section then moving gradually southwards. The southern section remains active but is currently nearing completion, with capacity falling from 13,000,000 m^3 in 1996 to around 800,000 m^3 in 2019 (NLLP, 2003; Wiki-waste, 2022b). In contrast, the northern section was capped and completed 30-years ago in 1992 during a time of heightened interest in land reclamation to forestry end-use (Moffat and McNeill, 1994; 1996a; Hislop and Harding, 1999). Forest Research (FR) established three near identical field experiments on landfills around the River Humber, one being situated on Winterton's northern section **(FIGURE 7.1).** Forest Research hypothesized organic amendments made from recycled plant biomass could improve soil structure, fertility, and moisture retention, thus enhancing tree performance on restored landfill soils (Hislop and Harding, 1999; Foot et al., 2003). Yet, unlike the other experiments established around the Humber, the Winterton experiment was installed several years after landfill capping and soil restoration (Hislop and Harding, 1999). This meant soils settled for several years prior to tree planting and compost incorporation, potentially increasing the risk of soil compaction.

FIGURE 7.1 – Aerial photograph of Winterton landfill. The site is split into two sections shown here by a dividing RED line. The northern section (N) was the first used for waste disposal and was completed in 1992. Five years later in 1997, Foot et al. (2003); (Hislop and Harding, 1999) established an experiment on this section (outlined in RED). This is 1 of 3 near identical experiments established throughout Humberside.

7.1.2 Immingham

Immingham landfill is a 32-ha site situated on the southern shore of the River Humber, immediately east of the Port of Immingham, England (Nat Grid Ref: TA 20310 14086)**.** It is the second of three landfills hosting composted green waste (CGW) related field experiments, and is one of several landfills known to have emerged around the port since the 1930's. These landfills were used to supply, store, and dispose of raw materials and wastes moving through the port or generated by surrounding industries. Indeed, ordnance survey data shows the site attached to Immingham was used to dispose of gypsum during the 1950's (OS-Maps, 1956) **(FIGURE 7.2)**. Presently, Knauf the well-known manufacturer of gypsum-based building materials continue to operate a plant opposite Immingham landfill, illustrating the sites decades long relationship with surrounding industry **(FIGURE 7.2).**

Immingham landfill was formerly farmland until the 1960's, when quarrying for sand commenced (OS-Maps, 1966; OS-Maps, 1956). However, by the late 1970's this was deemed un-economical and the quarry was converted to landfill. Presently, landfill activities continue at Immingham, which like Winterton is owned and operated by FCC environment. However, whereas Winterton receives hazardous inputs, Immingham receives non-hazardous waste from industrial, commercial, and domestic sources (EA, 2016; Wiki-waste, 2022a). This includes concrete, bricks, clay, sands, soils, mixed municipal waste, and road dusts (Wiki-waste, 2022a). Forest Research designed, assembled, and evaluated the CGW field experiment located in the eastern corner of Immingham landfill (Foot et al., 2003) **(FIGURE 7.2).** However, in contrast to Winterton the experiment at Immingham was installed immediately after capping and restoration minimizing settlement prior to tree planting and compost incorporation (Hislop and Harding, 1999).

FIGURE 7.2 *– Satellite image of Immingham landfill outlined in RED. The second of three near identical CGW related experiments was established here in 1997 (small RED square). The landfill receives non-hazardous waste and remains active at present. The large patch of white ground near the top of the image is the gypsum disposal bed.*

7.2 Site Conditions

Both Humberside sites share similar climatic conditions (Met-Office, 2011a). Indeed, annual climate data from surrounding weather stations show between 1991 and 2020 Winterton and Immingham received an average of 600 – 700 mm rainfall, 1550 – 1600 sunshine hours, and a mean annual temperature range of 6.8 – 14.5 °C (Met-Office, 2011a). The Humberside experiments are also near identical in terms of format and layout. However, being located 30 km apart means the experiments are clearly separate, whilst they also have differing micro-climate and soil conditions.

7.2.1 Winterton

The Winterton experiment is located inland from the Humber estuary inside 84-ha of privately owned landfill. Winterton beck watercourse runs adjacent to the western boundary, roughly 500 m beyond the site. The landfill is surrounded by agricultural fields on all sides however past iron ore extraction remains visible in the landscape. For example, ore extraction created cliff’s which straddle the eastern border, and land slopes downwards from the cliffs toward the western perimeter creating a sloping gradient which spans the site **(FIGURE 7.1).** Consequently the experiment resides in a slight depression sheltering it from prevailing wind and weather. It is also receives protection from nearby trees and hedgerows which alongside site topography mean exposure is moderate **(FIGURE 7.3, FIGURE 7.4, FIGURE 7.5)**. The experiment is located level ground on a section of landfill capped in 1992, using engineered clay and variable depth subsoil (0.7 – 1.2 m). The experiment was installed five years after soil restoration giving ground time to settle, meaning land had consolidated prior to compost incorporation and tree planting.

FIGURE 7.3 – *North facing end of Winterton field experiment (photograph is looking South). Experiment is on level ground and is surrounded by grassland. Vegetation in the foreground is dominated by couch-grass and sow-thistle. Dead trees (snags) are visible amongst Italian alder and Sycamore trees.*

FIGURE 7.4 – *Winterton field experiment from a distance (North facing end). Experiment is at the centre of image. Sloping land is visible in foreground and contrasts with level ground where experiment resides. Hedgerow to far left and copse of trees at far right shelter the experiment from prevailing winds.*

FIGURE 7.5 – *View from inside Winterton experiment (February 2019). Level ground can be seen once more as can the hedgerow at the experiments far South facing end. Numerous dead trees are also visible in this image, either leaning over or lying horizontal across the ground.*

7.2.2 Immingham

The experiment at Immingham is located near the banks of the Humber estuary inside a smaller 32 ha landfill. Three of the landfills four boundaries are surrounded by a large port, heavy industry, commercial land, and the town of Immingham. Only the southern boundary differs, in that land beyond the A1173 carriageway is agricultural rather than urban-industrial**.** North beck drain watercourse runs alongside the eastern boundary, flowing directly into the Humber estuary **(FIGURE 7.1).** Being just 10 m above sea level the landfill is low lying, however the experiment is situated on high ground on an unprotected corner of the landfill meaning climatic exposure is significant **(FIGURE 7.6, FIGURE 7.7, FIGURE 7.8).** Like Winterton, the Immingham experiment was established on restored and levelled-out sections of landfill with waste capped using engineered clay and variable depth subsoil (0.7 – 1.2 m). However, in contrast to Winterton, tree planting and compost incorporation were completed within months of landfill restoration meaning soil settlement and consolidation was minimal prior to establishing the experiment.

FIGURE 7.6 – *North-west facing end of Immingham field experiment (photograph is looking due East). Trees lean due to exposure from prevailing winds. Vegetation around the experiment is rough grassland. Unlike Winterton, there are no dead trees (snags) visible.*

FIGURE 7.7 – *Immingham field experiment from elevated vantage point. Experiment is isolated and situated on open land at the top of the landfill hence exposure is significant. Industrial and commercial buildings are visible in the background. The lean of the trees is also evident.*

FIGURE 7.8 – *View looking South-west from edge of experiment towards active area of landfill. Waste and debris blowing into the experiment caused significant losses in the year-1 (1998/99). After 20-years wind-blown waste no longer threatens trees but shows how prevailing winds move in from the South-west towards the experiment.*

7.3 Soil Materials

7.3.1 Winterton

Soils at Winterton were constructed in 1992 when landfill cells were complete, by compacting an engineered anaerobic clay cap compacted directly over landfill waste. A variable depth soil (0.7 – 1.2 m) was placed over this to create a growing medium for plants. The variable depth growing medium was constructed using locally available inorganic materials namely clay, waste gypsum, and foundry sand sourced from Scunthorpe steelworks. Inorganic materials were mixed together in varying proportions then placed over the cap in different areas of the northern section **(FIGURE 7.9).** Mixtures applied were pure clay, clay and foundry sand (1:1), and clay and waste gypsum (1:1). Exactly which mixtures were placed where remains uncertain (Drobig, 1999). Nevertheless, the description of 'sandy brown earth topsoil' given by previous authors is attributable to significant quantities of clay and foundry sand which comprise the growing medium (Drobig, 1999; Foot et al., 2003) **(FIGURE 7.10).**

***FIGURE 7.9** – Laying of subsoil treatments at Winterton in 1992. Manufacturing of subsoils was the first phase of 'The Humberside Project: Making New Soils from Waste'. Organic amendment trials followed subsoil construction, and were a secondary phase of the Humberside project (Drobig, 1999; Hislop and Harding, 1999; Foot et al., 2003).*

FIGURE 7.10 *– Soil described as 'sandy brown earth' excavated from Winterton in October 2018. The left-hand soil is the upper 0 – 10 cm topsoil and is darker and blacker suggesting higher organic matter content. The right-hand soil is browner and lighter, matching descriptions by previous authors of 'sandy brown earth' (Drobig, 1999; Hislop and Harding, 1999).*

7.3.2 Immingham

Soils at Immingham are similar to Winterton in terms of general construction and composition. However, slight differences exist between sites with potential impacts on soil development. For example, Immingham soils were constructed in 1997 immediately after landfill restoration hence delay between soil construction and compost incorporation was a matter of months, instead of years as at Winterton. Further, according to Drobig (1999) different input materials were used at Immingham. Instead of using foundry sand, Immingham soils were composed of local marine alluvium mixed with clay and waste gypsum (Drobig, 1999). This led Hislop and Harding (1999), and Foot et al. (2003) to describe Immingham's soils as 'clay-based' and 'heavy clay' respectively. High clay content is likely the reason why Immingham soils remained dry and prone to cracking when the experiment was established (Foot et al., 2003; Weil and Brady, 2016). Yet, although soil texture at Immingham differs from Winterton, soils at both sites were described as severely compacted prior to tree planting and compost incorporation (Hislop and Harding, 1999; Foot et al., 2003). To alleviate this compaction soils at both sites were ripped to a depth of 0.5 to 0.75 m, during a period of dry weather in November 1997 (Bending and Moffat, 1997; Hislop and Harding, 1999; Foot et al., 2003). Even after ripping Immingham's soils appeared to retain their large cloddy physical structure (Hislop and Harding, 1999). However, after 20-years of soil development Immingham's previously poor structure is less apparent **(FIGURE 7.11)**.

FIGURE 7.11 *– Soil excavated from a sampling pit at Immingham. In 2019 soils excavated at Immingham were more loosely structured and less cloddy than those at Winterton.*

In summary, soil construction was similar at both sites but soil texture and composition differ with Immingham soils containing marine alluvium instead of the foundry sand used at Winterton. Further, at Immingham organic amendments were incorporated immediately after landfill restoration, whereas at Winterton years had elapsed between landfill soil re-construction and compost incorporation. Finally, although soils at Winterton and Immingham are described as 'sandy brown earth' and 'heavy clay' respectively, soils at both sites were extremely heterogenous. Heterogeneity likely arises from imperfect mixing of organic and inorganic soil materials during soil reconstruction and compost incorporation, as well as the variety of treatments deployed at the experiments. Indeed, a wide variety of treatments were deployed potentially influencing results obtained by the present study, given a large number of treatments were compressed into a relatively small area. The arrangement of each experiment is now illustrated and described in greater detail.

7.4 Description of field experiments

In November 1997 works to establish experiments at Winterton (Nat Grid Ref: SE 91427 20056) **(FIGURE 7.12)** and Immingham (Nat Grid Ref: TA 20310 14086) **(FIGURE 7.19)** began under the guidance of Forest Research (FR). Their objective was to ascertain whether CGW could improve growth and survival of newly planted trees. The first act in establishing the experiments was to cultivate soils using an excavator via ripping to 0.5 to 0.75 m depth. Compost was incorporated during this process. The CGW used was produced onsite by 'Waste-wise' but was not PAS100 certified as the standard did not exist in 2001. Four CGW quantities were trialed at two different depths to understand the relative benefits of different application depths and rates **(TABLE 7.1)**. DEEP incorporation (0.6 m) was performed during soil cultivation by excavator and tested whether organic matter distribution throughout the profile could improve soil structure and moisture retention as roots expanded over time. In contrast, SHALLOW incorporation (0.1 m) was performed after cultivation, by mulching CGW over soils surface then incorporating it using a rotavator. This tested whether concentrating compost around newly planted roots might boost early growth by increasing organic matter and nutrient content in the root-zone of saplings (Hislop and Harding, 1999; Foot et al., 2003).

Nutrient content (kg/ha) of compost used on Humberside			
Nutrient	% content	Application rate (t/ha)	Equivalent nutrient (kg/ha)
Nitrogen (N)	0.03%	50	15
		100	30
		250	75
		500	150
Phosphorus (P)	0.024%	50	12
		100	24
		250	60
		500	120
Potassium (K)	0.30%	50	150
		100	300
		250	750
		500	1500

TABLE 7.1 *– Typical nutrient content of the four CGW application rates used on Humberside (kg/ha). These are generic guideline values from Forest Research project reports by (Hislop and Harding, 1999). Exact nutrient content of CGW applied on Humberside remains unknown. 500 t-ha^{-1} plots were not analysed by the present project.*

Once compost was incorporated in November 1997, trees were planted in the following spring of March 1998. This allowed a cycle of freeze thaw to break down soils cloddy structure. Experiments were then fenced off to protect young trees from herbivory prior to tree planting. All planted trees were bare-rooted transplant stock with a height of 40 – 60 cm. Two species were planted, Italian alder (*Alnus cordata*) and Sycamore (*Acer pseudoplatanus*) as both are capable of enduring harsh conditions typical of reclaimed sites (Moffat and Laing, 2003). For example, both species tolerate dry, wet, and heavy soils and extreme alkaline pH. Further, Sycamore tolerates acidic soils and exposure, whilst Italian alder can fix its own nitrogen (Moffat and McNeill, 1994; Hislop and Harding, 1999). When trees were planted all plots were free of weed growth. However, the area around each tree was periodically treated with Roundup and Kerb during the first year. Experimental observations began in April 1998, continuing for 4-years through to 2001. Where necessary dead trees were replaced in March 1999 after the first full 12-month cycle (Hislop and Harding, 1999). Throughout the four-year monitoring period tree growth, survival, foliar nutrition, and ground vegetation cover were analysed. In the early years there were no significant relationships between tree performance and CGW. However, by 2001 results showed increasing quantities of compost improved tree growth, foliar nutrition, and ground vegetation cover. DEEP incorporation was particularly beneficial for tree height, whilst SHALLOW incorporation increased ground vegetation cover (Hislop and Harding, 1999; Foot et al., 2003). Visual evidence of these dynamics was captured by a series of photographs taken between 1998 and 2001 by Forest Research (see **FIGURE 7.13** to **FIGURE 7.25**).

Regarding layout, experiments were set out in a randomised split-split plot design. The design was repeated in four separate blocks, creating four replicates for each experimental treatment. Moving down through each 'level' of the experiments from largest to smallest scale, compost 'incorporation depth' was the main plot treatment, whilst 'compost quantity' was the sub-plot treatment. Finally below this was 'tree species' as the sub-sub plot treatment**.** The experimental layouts provided in **FIGURE 7.12** and **FIGURE 7.19** show the layout of each experiment.

Evidently a large number of plots and treatments were compacted into a relatively small area (0.25 ha). Indeed, each experiment contains 80 plots in total, with every plot sized 7.5 x 3 m, and containing a monoculture of 10 trees spaced at 1.5 x 1.5 m. Overall, experimental plots on Humberside are the smallest of those featured in this thesis and contain the fewest trees of any experiment studied. This is because experiments on Humberside were designed to evaluate composts early impacts on woodland establishment and were not intended for use as long-term study sites.

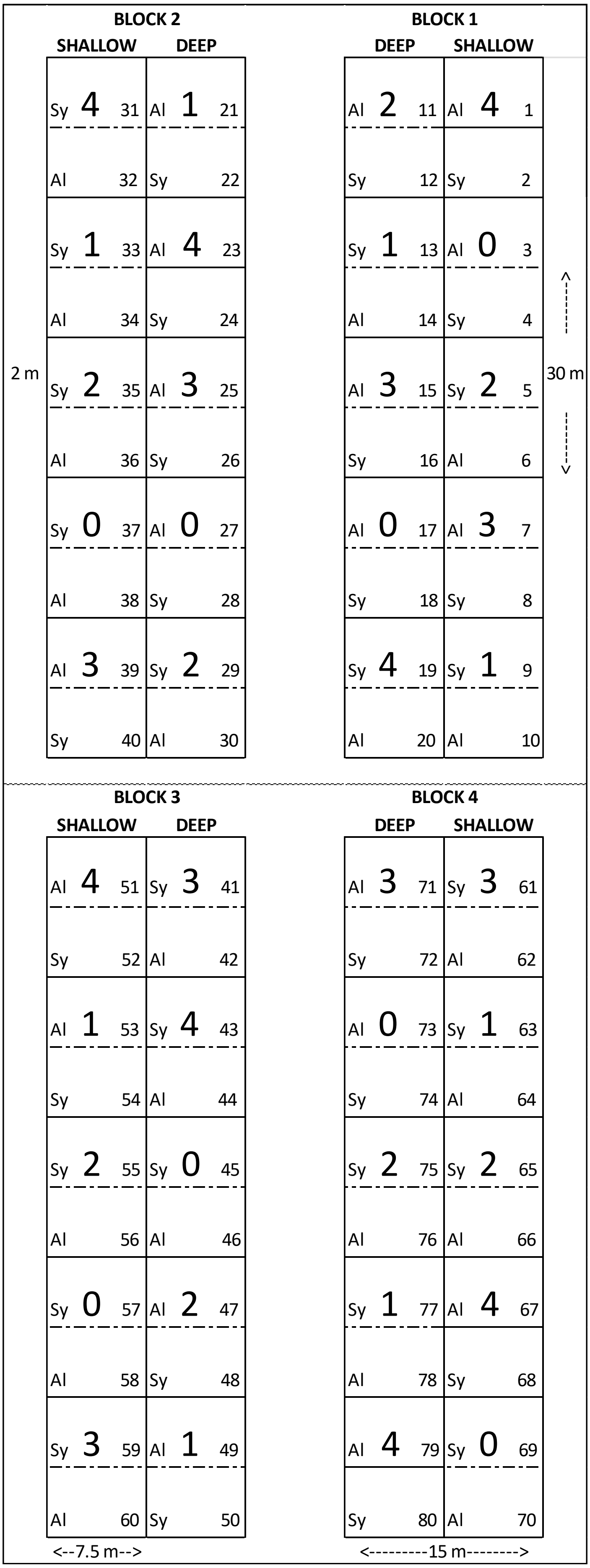


***FIGURE 7.12** – WINTERTON experiment layout with four replicate blocks set out in a 'split-split plot' design. Plots are 'split' into three different levels. The main level is compost <u>incorporation depth</u> (i.e. SHALLOW = 0.1 m or DEEP = 0.6 m). The sub-plot level is <u>compost quantity</u> (i.e. 0 = CONTROL, 1 = 50 t-ha^{-1}, 2 = 100 t-ha^{-1}, 3 = 250 t-ha^{-1}, 4 = 500 t-ha^{-1}). The sub-sub plot level is <u>tree species</u> (i.e. Al = Italian Alder, Sy = Sycamore). 500 t-ha^{-1} plots were not analysed by the present project.*

__FIGURE 7.13 –__ Winterton experiment approximately 12-months after planting. Photograph facing north-west. Young trees (whips) are visible in the foreground where the ground is bare. Ground vegetation cover is increasing in some areas whilst others remain bare.

__FIGURE 7.14 –__ Winterton experiment approximately 12-months after planting. Photograph facing south-west. Again, young trees (whips) are visible in the immediate foreground however ground cover vegetation is more extensive in this photograph.

***FIGURE 7.15** – Italian alder (Alnus cordata) at Winterton. A single tree is visible at the centre of the photograph surrounded by dense cover of White clover (Trifolium repens).*

***FIGURE 7.16** – Winterton experiment in 2001, the fourth and final year of Foot et al (2003) original observations. Photograph facing north. Italian alder is starting to dominate the site. For the first time tree growth is significantly greater for both species in DEEP and 250 t-ha^{-1} treatments.*

FIGURE 7.17 – *Winterton experiment also in 2001. Photograph facing north-west. On the right hand-side of the image Sycamore trees can be seen, identifiable by their light green leaves. On the left hand-side of the photograph Italian alder can be seen, as can their rapid and superior growth rate. On-site production of CGW is on-going in the background.*

FIGURE 7.18 – *On-site production of CGW at Winterton in 2001. Waste-wise, the landfill operator at this time were managing on-site production. PAS-100 specification did not exist in 2001, hence end CGW products could be highly variable. For example, the compost in the picture is extremely course and appears to contain non-compostable plastic wastes.*

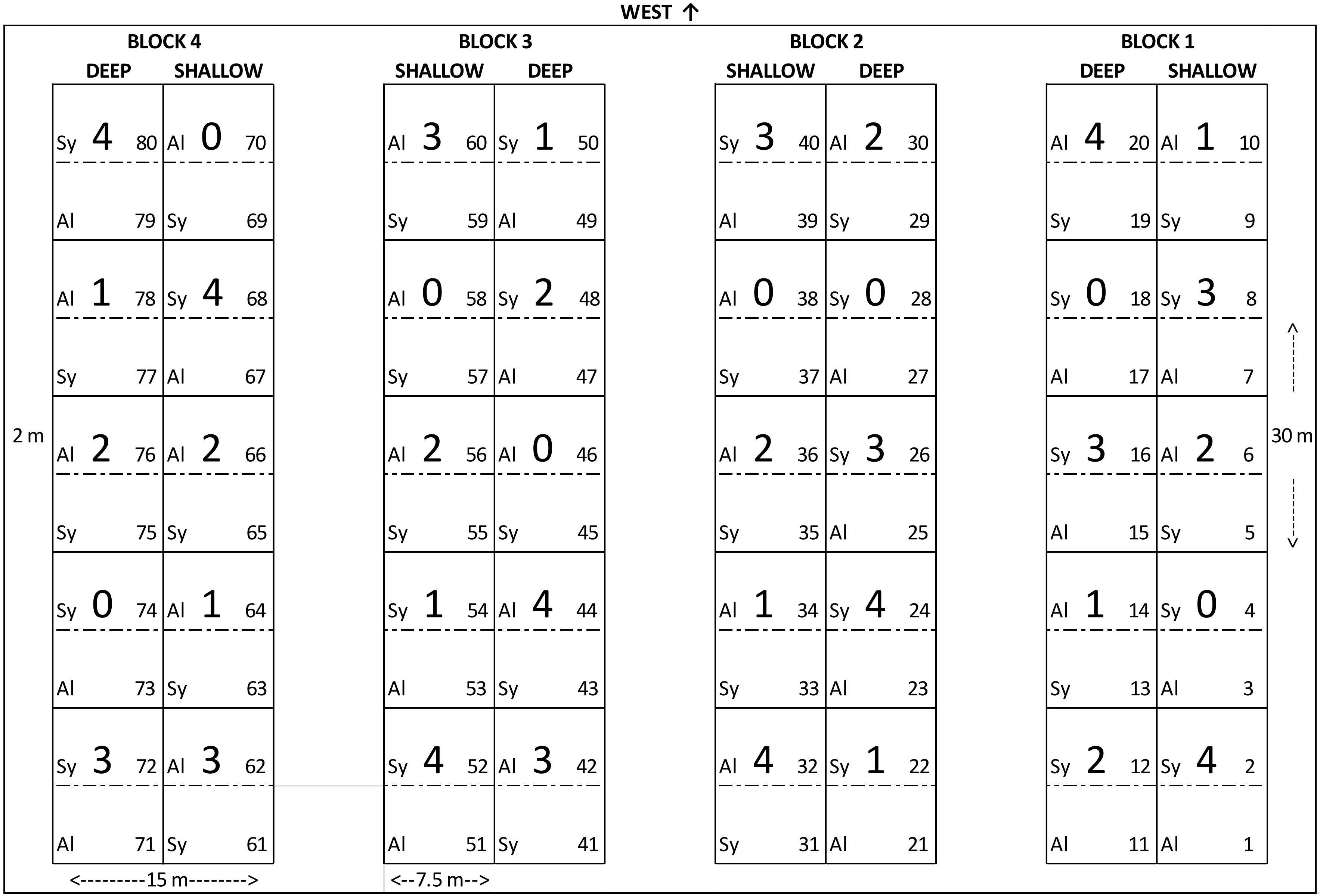


FIGURE 7.19 – *IMMINGHAM experiment layout. The only difference between Winterton and Immingham experiments is the position / orientation of the four replicate blocks. The blocks are set out in a 'split-split plot' design with the 'split' made up of three different levels. The first and main level is compost incorporation depth (i.e. SHALLOW = 0.1 m or DEEP = 0.6 m). The sub-plot level is compost quantity (i.e. 0 = CONTROL, 1 = 50 t-ha^{-1}, 2 = 100 t-ha^{-1}, 3 = 250 t-ha^{-1}, 4 = 500 t-ha^{-1}). The sub-sub plot level is tree species (i.e. Al = Italian Alder, Sy = Sycamore). 500 t-ha^{-1} plots were not analysed by the present project.*

FIGURE 7.20 – *Immingham experiment 12-months after planting. Photograph facing south-west. Young trees (whips) are visible in the foreground. At this stage ground at Immingham is bare and vegetation cover is low, partly because soils reconstruction occurred just a few months before the experiment was established.*

FIGURE 7.21 – *Immingham experiment 12-months after planting. Young trees (whips) are visible in the picture amongst bare and dry ground which has cracked due to low vegetative cover and high clay content. On the right of the image is a young Sycamore sapling with survival at 59 % for this species. Italian alder survival was extremely low with only 11% surviving after 12-months.*

FIGURE 7.22 – Immingham experiment 24-months after planting. Photograph facing north-east. Young trees (whips) are again visible throughout the image. Ground remains bare but vegetation cover is beginning to increase.

FIGURE 7.23 – Immingham experiment 36-months after planting. Photograph facing south. Italian alder is visible in this image with tree size increasing following 18-months of growth. Ground vegetation cover remains fairly sparse but continues to increase.

FIGURE 7.24 – Immingham experiment in 2001, the fourth and final year of Foot et al (2003) observations. Sycamore occupies the foreground identified by light green leaves. Italian alder occupy the midground, having increased their foliar mass and grown much taller than Sycamore. Ground vegetation cover is now extensive and is dominated by White clover (T. repens) and Black medick (Medicago lupulina). Photograph facing north towards the Port of Immingham.

FIGURE 7.25 – Immingham experiment in 2001, the fourth and final year of Foot et al (2003) observations. Photograph facing north-west towards the Port of Immingham. Again, Sycamore occupies the foreground and can be identified by its light green leaves. Ground vegetation is extensive with Black medick (Medicago lupulina) covering Sycamore plots. The remaining trees are all Italian alder and have increased their foliar mass and height since 1998.

7.5 Selection rationale

The Humberside experiments were chosen primarily due to site age, being the oldest CGW related field experiments to remain in existence throughout the UK. The ability to evaluate compost incorporation depths impact on soil development was similarly important, as was the presence of duplicate experiments which enabled experimental dynamics to be compared and contrasted at different sites. Concerning duplicate experiments, the original intention was to re-visit all three studies established throughout Humberside, however time constraints meant Carnaby was excluded from analysis. Nevertheless, the ability to study site dynamics at Winterton and Immingham provided an unprecedented opportunity to evaluate CGW legacy impacts on tree growth and soil pedogenesis at 20-years.

Evaluating the effects of different incorporation depths on Humberside was similarly attractive, especially given other sites visited by this study trialed incorporation at depth alone. The legacy effect of shallow incorporation on woodland development had not been studied in the UK literature. Hence, Winterton and Immingham provided an opportunity to compare the effects of deep versus shallow incorporation on soil physical, chemical, and biological development, and tree performance after 20-years. Concerning tree species, those planted on Humberside were similar to species planted at the 5-year-old study site (Ingrebourne Hill). This provided an added dimension wherein long-term tree growth dynamics could be envisioned over time.

Finally, it was recognised Foot et al. (2003) original study had significant limitations future studies were required to investigate. Namely, soil development was not assessed. This omission meant trends in tree growth and foliar nutrition could not be understood in the context of soil development, functions, and processes. Therefore, by revisiting the experiments and evaluating above and below-ground interactions, trends in tree performance and soil development could be cross-referenced and compared. As an example, if tree performance was influencing soil organic matter, carbon storage, or earthworm activity, these relationships could be recorded and confirmed.

7.5.1 Aims and objectives

Explore CGW legacy impacts (defined as 5+ years) on provision of ecosystem services related to tree growth, soil formation, earthworm activity, and soil organic carbon fate.

1. Determine CGW long term impact on above-ground tree growth, biomass production, and foliar nutrient content at reclaimed sites;

2. Record physical, chemical, and biological attributes of CGW-amended soils to identify potential links between CGW application rate, CGW incorporation depth, soil formation, and carbon storage;

3. Determine how earthworm community composition, density, and biomass impact pedogenesis in CGW-amended soils;

4. Outline CGW and earthworms synergistic impact on supporting and regulating ecosystem services.

7.6 Materials and Methods

The following information summarises the sampling and analysis techniques used to investigate interactions between compost application, tree growth, soil chemistry, and earthworm activity on Humberside. A more detailed description of the techniques used to evaluate all study sites is provided in **CHAPTER 4 – GENERAL MATERIALS AND METHODS**. Similar techniques were used at all sites to ensure data generated could be readily compared and discussed.

Tree growth and foliar chemistry were measured at two experiments, both located in landfills on Humberside, to understand how large quantities of compost influence primary productivity and nutrient cycling 20-years post CGW application. To evaluate tree growth, a Haglof Vertex IV digital hypsometer and ultrasonic transponder were used to measure tree height in February 2019 when foliage was minimal. Stem diameter was then recorded using a circumference tape manufactured by Richter. Foliar chemistry was evaluated to understand how different compost treatments influenced quantities of foliar nutrients. Foliar collections were completed in mid-summer to ensure leaf nutrient concentrations were stable. Foliage was collected from Winterton in July 2018 and from Immingham in July 2019. To create a plot representative foliar sample leaves were taken from several different trees per plot then bulked together. Once returned to the laboratory foliage was oven-dried, homogenised, and sieved to obtain smaller particles 1 – 2 mm in size. A sub-sample was taken from the sieved leaf particles then ground to powder in a planetary ball mill ensuring foliage was ready for carbon and nitrogen analysis. Further details are provided in **SECTION 4.2.1** and **4.2.2**.

To evaluate compost impact on soil nutrient cycling and carbon storage, soil cores were taken from each experimental plot in November 2019. A Dutch auger fitted with 15 cm attachment was used to extract duplicate cores from 0 – 15 and 15 – 30 cm depth. Cores were taken from two different locations per plot, prepared for analysis, then bulked to create a plot-representative sample for each depth. Soil organic matter content, total carbon, total nitrogen, C:N ratio, and available nitrates were all assessed. Available-N was determined by taking 10 g of fresh soil, performing a KCL-extraction, then sending soil extracts to Rothamsted Research laboratory for colorimeter analyses. For the remaining soil chemical analyses, cores were air dried, crushed in a motorised blade grinder, and sieved to > 2 mm to remove stones and other large particles. Replicate cores were then bulked and sieved again to 0.5 mm, creating a fine homogenised soil. This fine dried material was used to determine organic matter content via loss on ignition, and total carbon, total nitrogen, and C:N ratio for soil and foliar samples using a Thermo-Scientific CHNS Organic Elemental Analyser. Further details are provided in **SECTIONS 4.3.3** and **4.3.4**.

Soil bulk density (SBD) was used as a proxy for soil formation. SBD cores also provided readings for soil moisture content. SBD cores were extracted from earthworm pits when earthworms were sampled. This was done in September-October 2018 at Winterton, and April-May 2019 at Immingham. Three depths for SBD were assessed (0 - 10, 10 - 30, and 30 - 50 cm). A hammer and 100 cm^3 cylindrical steel core were used to extract SBD cores from the sides and base of each 30 x 30 x 40 cm^3 earthworm pit (two per plot), providing duplicate cores for each sampling depth (Robertson et al., 1999). Cores were oven dried at 105°C for 24 hours to establish soil bulk density, moisture content, and total porosity. Further details are provided in **SECTION 4.3.1** and **4.3.2**.

Earthworm population density and community structure was measured to explore how interactions among compost, earthworms, and tree species mediate ecosystem service provision. Sampling was conducted in in September-October 2018 at Winterton, and April-May 2019 at Immingham following the methods described by Butt and Grigoropoulou (2010). In each plot, two earthworm pits were excavated to obtain a plot representative sample. Sampling points were created by placing a 0.1 m^2 quadrat on the soil surface, excavating 30 x 30 x 40 cm^3 of soil from beneath the quadrat, then hand-screening excavated soil for earthworms. Earthworm specimens were then placed in plastic bottles and preserved in 4% formaldehyde solution prior to identification. All adult worms were identified to species level following the key of Sims and Gerard (1999). Visual identification was completed using a stereo microscope with adjustable zoom (X10 – X30 magnification). Specimens were dried on absorbent paper then weighed to determine biomass. Population densities found in each 0.1 m^2 pit were scaled up and reported as average number of species / eco-group per m^2. Further details are available in **SECTION 4.4**.

7.7 Results - Winterton

7.7.1 Impact of compost addition on tree survival, growth, and foliar nutrition

When the Humberside experiments began in 1998, survival was poor during the first two years. Italian alder survival was particularly low at two sites and all dead trees were replaced annually throughout this time. Of the original Italian alder planting, only 11% survived at Immingham, and 15% at Carnaby. In contrast survival at Winterton was much higher, with three quarters (74 %) of the original Italian alder and over half (57 %) the original Sycamore surviving in 2001 (Foot et al., 2003). Yet, despite Alders good early performance at Winterton, monitoring in February 2019 found Alder survival had declined to between 53 and 28 % depending on the treatment **(FIGURE 7.26** and **FIGURE 7.27)**. Indeed, Alder survival was actually lowest in 250 t-ha^{-1} treatments (28 %), and highest (53 %) in SHALLOW-CONTROLS. In contrast, at the same 2019 monitoring point, Sycamore survival was high in the 250 t-ha^{-1} treatments (80 %) and lowest in controls (50 %) **(FIGURE 7.28** and **FIGURE 7.29)**. Thus, compost had the opposite effects on species survival, with larger application rates increasing Sycamore survival, but decreasing Alder.

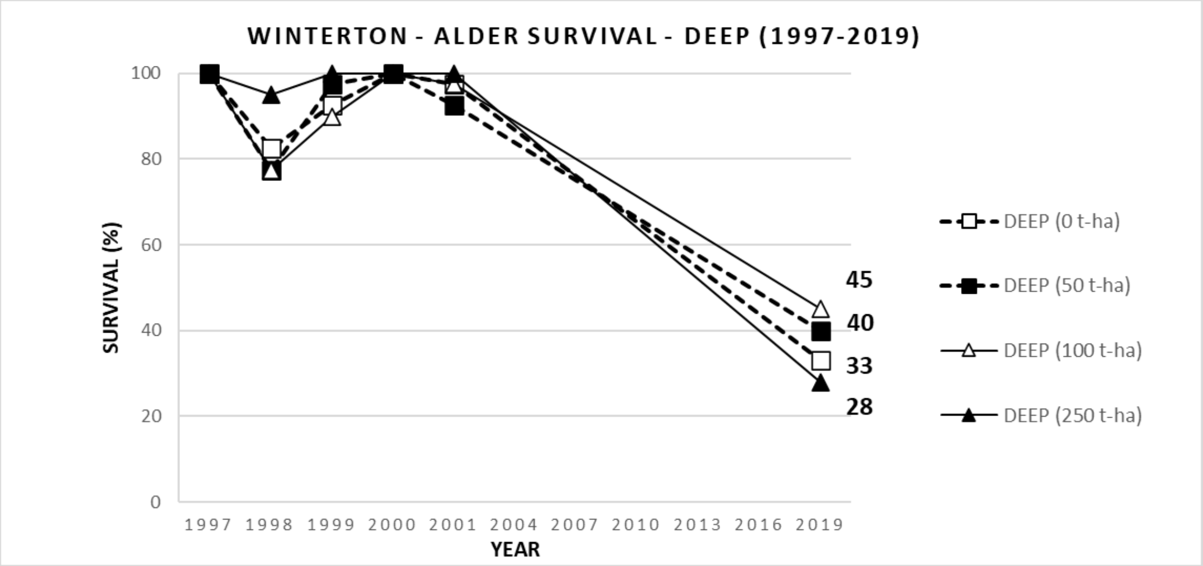


FIGURE 7.26 *– Italian Alder (%) survival at WINTERTON in the DEEP (0.6 m) compost incorporation from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise throughout the first four years. In 2019 survival was highest in DEEP 100 t-ha^{-1}, and lowest in DEEP 250 t-ha^{-1}, with the latter being significantly lower than that predicted by chi-square (χ^2) test for independence: χ^2 (15, 640) = 107.3, p < .001.*

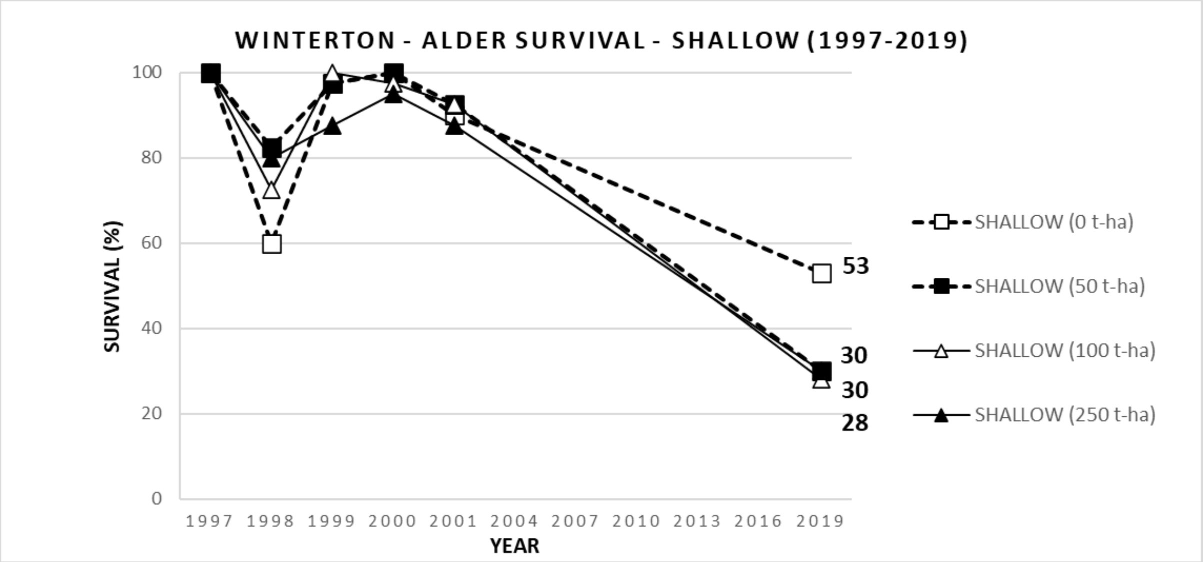


FIGURE 7.27 – *Italian Alder (%) survival at WINTERTON in the SHALLOW (0.1 m) incorporation from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise early on. In 2019, survival was highest in SHALLOW-CONTROL (0 t-ha^{-1}), and lowest in all other CGW applications. Indeed, chi-square (χ^2) test for independence found only the SHALLOW-CONTROL (0 t-ha^{-1}) reached the predicted survival rate. Seemingly, wherever compost was incorporated Italian alder survival fell significantly below the predicted rate: χ^2 (15, 640) = 107.3, p < .001.*

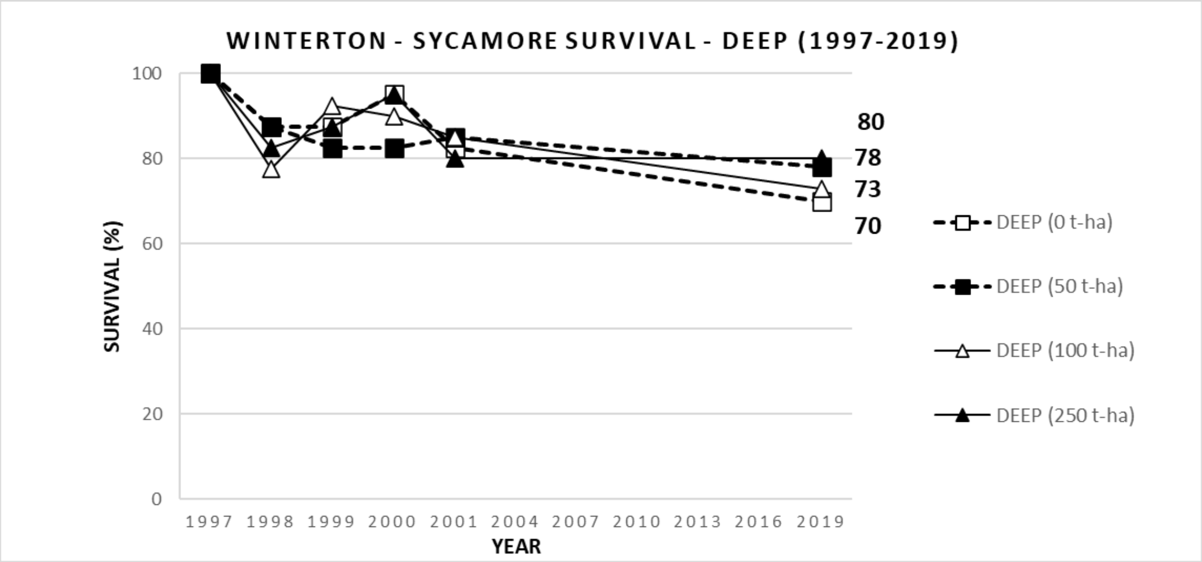


FIGURE 7.28 *– Sycamore (%) survival at WINTERTON in the DEEP (0.6 m) compost incorporation from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise throughout the first four years. Sycamore reached its highest rates of survival in DEEP 250 t-ha^{-1} and 50 t-ha^{-1} treatments, and lowest rate of survival in CONTROL, a pattern opposite to that found for Italian alder. Indeed for Sycamore, survival in DEEP 250 t-ha^{-1} treatments was significantly higher than that predicted by chi-square (χ²) test for independence: χ² (15, 640) = 107.3, p < .001.*

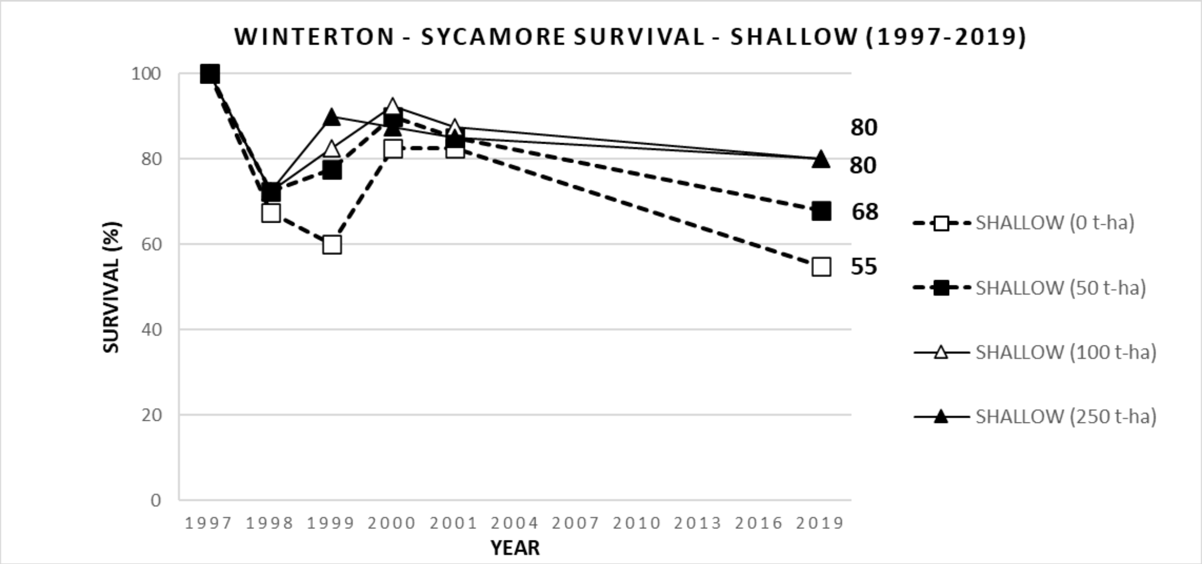


FIGURE 7.29 *– Sycamore (%) survival at WINTERTON in the SHALLOW (0.1 m) compost incorporation from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise during the first four years. In SHALLOW incorporations, Sycamore achieved its highest survival in the 250 t-ha^{-1} and 100 t-ha^{-1} treatments, and lowest in CONTROL, a pattern opposite to that found for Italian alder. Indeed, both of these application rates achieved significantly higher survival than predicted by chi-square (χ²) test for independence: χ² (15, 640) = 107.3, p < .001.*

Because Italian alder survival was low and appeared to differ between treatments, a chi-square (χ^2) test for independence (also known as Pearsons χ^2 test, or χ^2 test of association) was used to check how each treatment impacted survival. The test functions by comparing actual survival recorded on-site, with 'predicted' survival data calculated by a model. The model creates a prediction by spreading total survival (n = 347 trees) equally across all experimental treatments (n = 16), thus creating an equal distribution of trees surviving per treatment (n = 21.7 trees). This 'prediction' then acts as a reference against which actual data (unequally distributed) can be compared. Results are summarised in **TABLE 7.2**. The model predicted 21.7 of 40 trees (54.25%) would survive in each treatment. Yet, the chi-square (χ^2) test for independence found actual data differed significantly from this in certain cases: χ^2 (15, 640) = 107.3, $p < .001$. In general, Sycamore survival was significantly *higher* than predicted, particularly when larger quantities of compost were applied. Conversely, Alder survival was *lower* than predicted, mainly in SHALLOW treatments where only SHALLOW-CONTROL achieved predicted survival. Thus in summary, large concentrations of compost produced higher than expected survival for Sycamore (77.5 – 80 %), but lower than expected survival for Alder (27.5 – 30 %) **(TABLE 7.2)**. Tree growth will now be examined.

WINTERTON - TREE SURVIVAL (20 years)

Tree / CGW-Depth	CGW (t-ha)	Predicted	Actual	% Survival	Higher	Lower	z-score	p = value
Alder Deep	250	21.7	11	27.5		↓	-2.2953448	0.05
Alder Shallow	50	21.7	12	30		↓	-2.0806259	0.05
Alder Shallow	100	21.7	11	27.5		↓	-2.2953448	0.05
Alder Shallow	250	21.7	12	30		↓	-2.0806259	0.05
Sycamore Deep	50	21.7	31	77.5	↑		1.9990327	0.05
Sycamore Deep	250	21.7	32	80	↑		2.2137516	0.05
Sycamore Shallow	100	21.7	32	80	↑		2.2137516	0.05
Sycamore Shallow	250	21.7	32	80	↑		2.2137516	0.05

TABLE 7.2 – *Impact of treatment on predicted survival at WINTERON. Results from a chi-square (χ^2) test for independence found predicted survival was significantly different from observed survival in certain cases: χ^2 (15, 640) = 107.3, $p < .001$. RED arrows indicate treatments where survival was significantly lower than expected. GREEN arrows indicate treatments where survival was significantly higher than expected. In general, large quantities of compost were associated with high survival for SYCAMORE (77.5 – 80 %), but low survival for ALDER (27.5 – 30 %).*

Regarding tree height, when the initial experiment concluded in 2001, several statistically significant effects were noted by Foot et al. (2003). Namely, 1). Italian alder grew significantly taller than Sycamore in all treatments; 2). Sycamore height increased alongside increasing quantities of compost; 3). Trees in DEEP incorporations grew significantly taller than those in SHALLOW (Foot et al., 2003). When monitoring was conducted again in February 2019 these effects remained detectable, albeit with a subtle change. Firstly, regarding Alder's superior growth, a Kruskal-Wallis non-parametric ANOVA followed by Dunn's post-hoc with bonferroni correction found Alder grew significantly taller than Sycamore throughout Winterton: H (7) = 259.7, p 0.001 **(FIGURE 7.30)**. A similar independent one-way ANOVA was performed for stem diameter (DBH) and again found Alder DBH was significantly greater than Sycamore in all treatments: F (7, 339) = 22.13, p 0.001 **(FIGURE 7.32)**. Clearly, Alder's faster growth habit allowed it to grow taller with broader stems. Further, Alder achieved its greatest height of 9.9 m^{-1} in the 250 t-ha^{-1} treatments (0.9 m^{-1} taller than ALDER-CONTROL). However, a caveat to Alder's superior growth also occurred in the 250 t-ha^{-1} treatment. Namely, by 2019 Sycamore height had increased so much (7.3 m^{-1}) it was statistically comparable to Alder. Indeed, under this higher 250 t-ha^{-1} rate of compost Sycamore grew significantly (+ 1.4 m^{-1}) taller than its 100 t-ha^{-1} counterpart (p 0.05) **(FIGURE 7.30)**.

Finally, incorporation depth produced significant differences in tree growth. In 2001, trees in DEEP (0.6 m^{-1}) incorporations had grown significantly taller than trees in SHALLOW (0.1 m^{-1}). A Kruskal-Wallis non-parametric ANOVA followed by Dunn's post-hoc with bonferroni correction revealed trees in ALDER-DEEP grew 1.1 m^{-1} taller, whilst SYCAMORE-DEEP grew 0.9 m^{-1} taller, than their SHALLOW counterparts: H (7) = 259.7, p 0.001 **(FIGURE 7.31)**, with much of this occurring post 2001. For example, Alder's annual growth increments show that in 2001, ALDER-DEEP was only 0.64 m^{-1} taller than ALDER-SHALLOW. Thus, to reach the 1.1 m^{-1} difference recorded in 2019, the gap increased by a further 0.5 m^{-1} post 2001. Concerning Sycamore, in 2001 the difference between DEEP and SHALLOW was 0.09 m^{-1}. Thus, to achieve the 0.9 m^{-1} difference recorded in 2019, the height gap increased by a further 0.81 m^{-1} post 2001 (an increase of 1000 %) **(TABLE 7.3)**.

FIGURE 7.30 – *Impact of CGW quantity on tree height. Where letters are not shared a significant difference exists (p<0.05). Kruskal-Wallis H test followed by Dunn's post-hoc with bonferroni correction revealed all ALDER treatments grew significantly taller than all SYCAMORE* <u>*H (7) = 259.7, p 0.001*</u>*. The exception was SYCAMORE 250 t-ha^{-1} which grew so tall the difference with ALDER was no longer signifcant. Compost appears to have improved ALDER height, as evidenced by incremental height increases with increasing quantity of compost. This resulted in ALDER (250 t-ha^{-1}) growing 0.9m taller than ALDER-CONTROL, however differences among ALDER treatments were not statistically significant. 6.675*

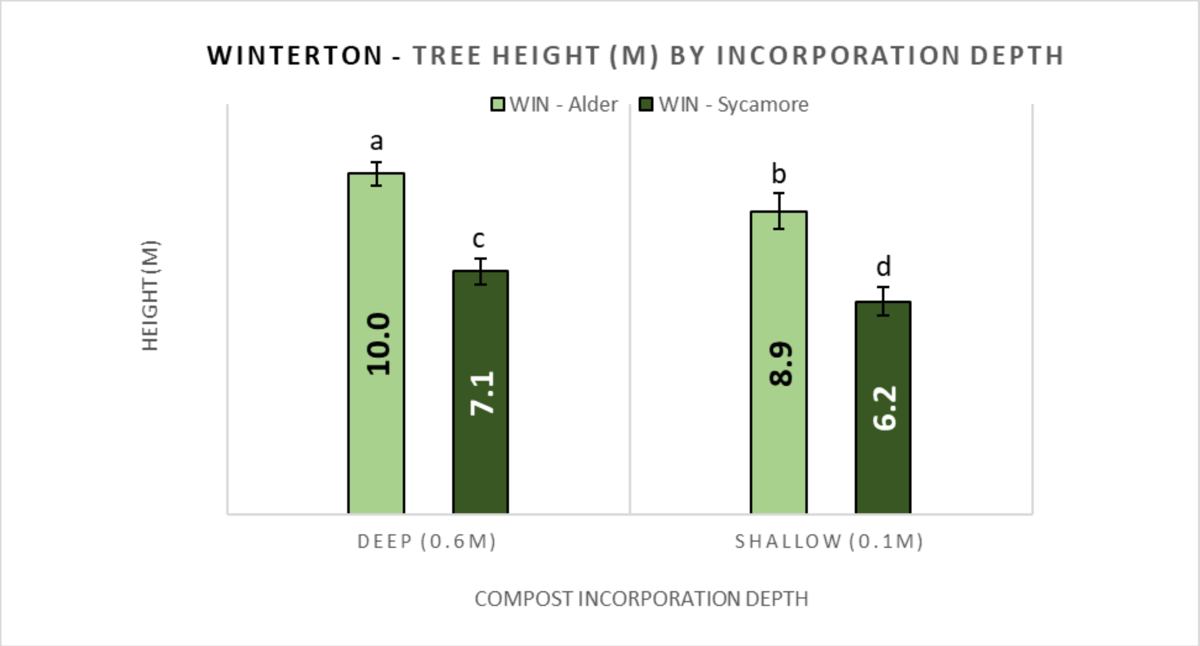


FIGURE 7.31 – *Impact of incorporation depth on tree height. Where letters are not shared a significant difference exists (p<0.05). Kruskal-Wallis H test with bonferroni correction compared tree height in DEEP versus SHALLOW incorporations. Dunn's post-hoc test revealed trees in DEEP treatments grew significantly taller (roughly 1.0 m taller) than trees in SHALLOW incorporations. Additionally, ALDER grew around 3.0 m taller than SYCAMORE* <u>*H (3) = 115.8, p <0.001*</u>*.*

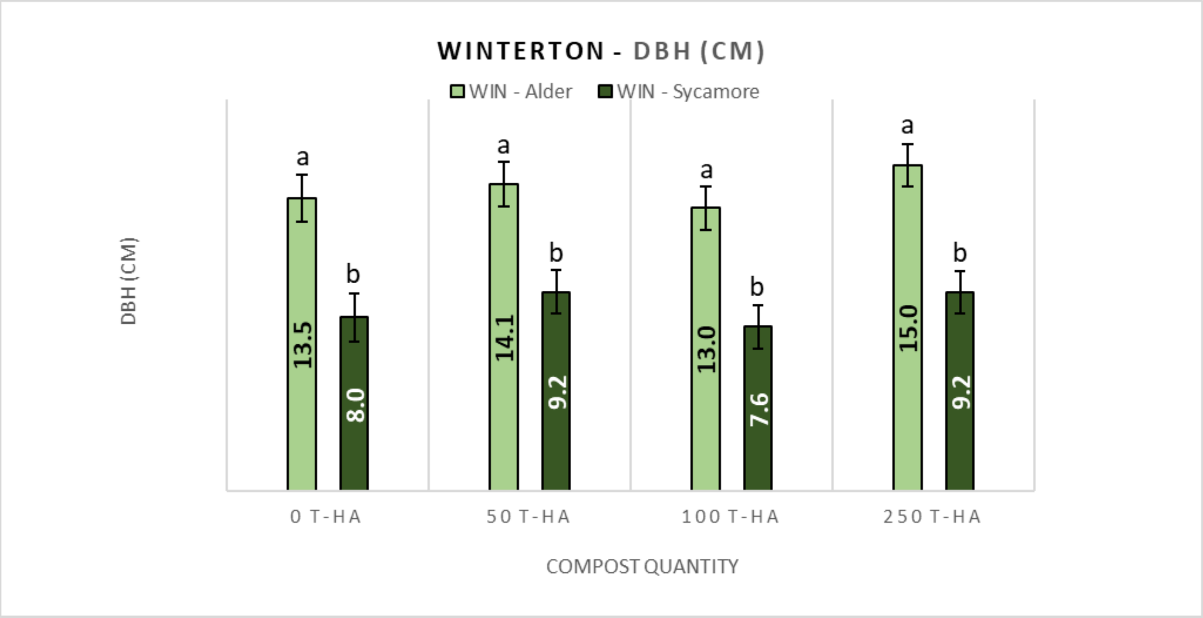


***FIGURE 7.32** – Impact of CGW quantity on tree stem diameter at WINTERTON. Where letters are not shared a significant difference exists (p<0.05) between treatments. CGW quantity did not have any impact on stem diameter (DBH) for either tree species. However, independent one-way ANOVA followed by post-hoc test with bonferroni correction showed all ALDER treatments had significantly greater DBH than all SYCAMORE treatments F (7, 339) = 22.13, p 0.001.*

WINTERTON - ANNUAL (%) GROWTH RATES (1997 - 2019)

Site	Treatment	**ALDER-DEEP**		**ALDER-SHALLOW**		**SYCAMORE-DEEP**		**SYCAMORE-SHALLOW**	
	Year	Height (m)	Growth (%)	Height (m)	Growth (%)	Height (m)	Growth (%)	Height (m)	Growth (%)
Winterton	1997	0.45		0.45		0.40		0.42	
	1998	0.52	16	0.51	13	0.39	-3	0.39	-7.14
	1999	0.84	62	0.72	41	0.36	-8	0.37	-5.13
	2000	1.66	98	1.31	82	0.46	28	0.43	16.22
	2001	2.44	47	1.80	37	0.64	39	0.55	27.91
	2002 - 2019	10.00	8	8.90	9	7.10	14	6.20	14

***TABLE 7.3 –** Annual tree height and (%) growth increments at WINTERTON between 1997 and 2019. Trees planted in DEEP treatments grew 1.1 m taller for ALDER, and 0.9 m taller for SYCAMORE. However, in 2001 DEEP treatments had only grown 0.64 m taller for ALDER and just 0.09 m for SYCAMORE. Thus, to achieve the height increase recorded in 2019, a further 0.5 m for ALDER-DEEP and 0.81 m for SYCAMORE-DEEP must have occurred post-2001.*

In reclaimed soils, Italian alder and composted green waste (CGW) are thought to improve soil fertility, nitrogen availability and nutrient cycling. To evaluate whether these processes were indeed influenced by different quantities of compost and the growth of Italian alder, foliar and soil C:N ratios were measured throughout Winterton at 20-years. Italian alder and Sycamore were expected to have different soil and foliar C:N ratios, given Alder's ability to symbiotically fix atmospheric-N. However, very little difference was found between soil and foliar C:N ratios associated with either tree species **(FIGURE 7.33** and **FIGURE 7.34)**. Indeed, the degree of homogeneity found across soil and foliar C:N values at Winterton, suggests Alder, compost, and possibly other site dynamics may have lowered C:N ratios to a similar level across the entire experiment.

For example, throughout Winterton, all soil and foliar C:N ratios were under 20:1, ranging 19:1 – 13:1. Interestingly, the lowest C:N ratio measured was soil C:N in ALDER-CONTROL (13:1), which did not receive composted green waste **(FIGURE 7.34)**. Concerning foliar C:N, little difference existed between species, with Sycamore ranging (19:1 – 17:1), and Alder (17:1 – 16:1) respectively. Soil C:N values were even more homogenous, with Sycamore ranging (16:1 – 14:1) and Alder (16:1 – 13:1). Overall, the greatest difference between species was in foliar C:N CONTROLS, with Alder at (16:1) and Sycamore (19:1). However, this difference was not apparent in corresponding CONTROL soils **(FIGURE 7.33)**. In summary, similar C:N ratios were found both above and below-ground throughout Winterton (<20:1), with little difference between species apart from foliar C:N CONTROLS. In all likelihood this means that over a 20-year period, Alder, compost, and perhaps ground vegetation have homogenised C:N ratios, despite the fact that large numbers of Alder no longer survive. This is the first indication of a possible 'Alder-effect', outlined further in data on soil chemistry.

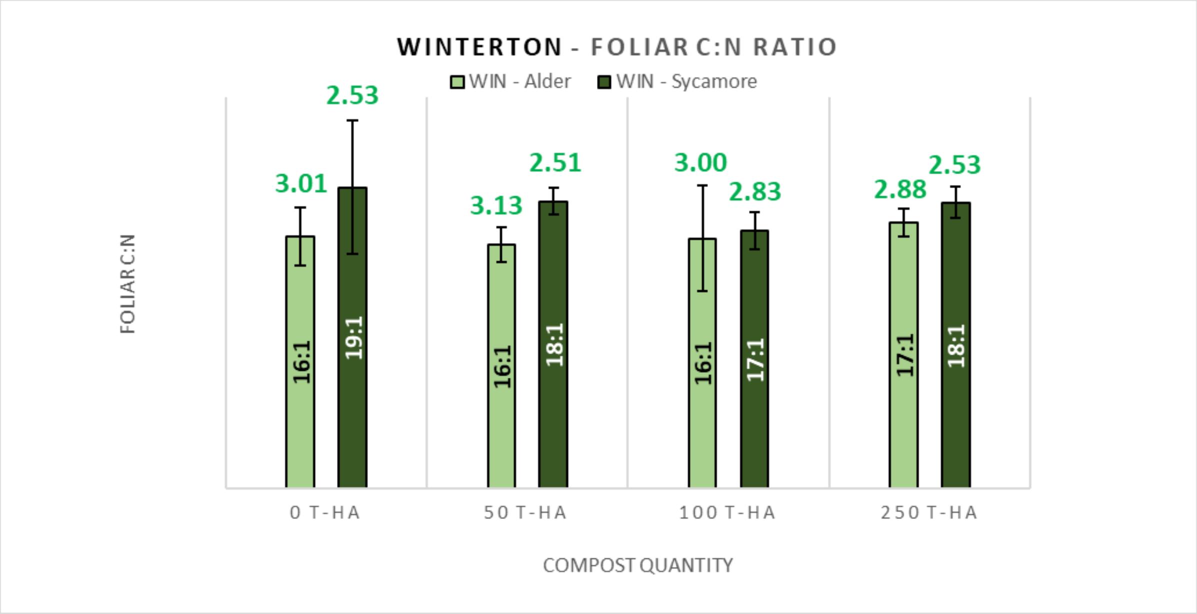


FIGURE 7.33 *– Impact of CGW quantity on foliar C:N ratio. No significant differences were found between treatments. In fact, foliar C:N was similar for all tree species and quantities of compost, especially ALDER. The highest foliar C:N ratio recorded was SYCAMORE-CONTROL (19:1). Comparing foliar C:N with soil C:N in the figure below, C:N ratios of foliage were consistently higher than those in soil. This suggests tree foliage alone did not lower soil C:N, as soil values are already lower than foliage. Values above bars represent (%) foliar-N.* ***GREEN =*** *above FC guidelines of (2.3 % for SYC; 2.8% for ALD).* ***PINK =*** *below FC guidelines of (2.3 % for SYC; 2.8% for ALD). No trees were Foliar-N deficient at Winterton.*

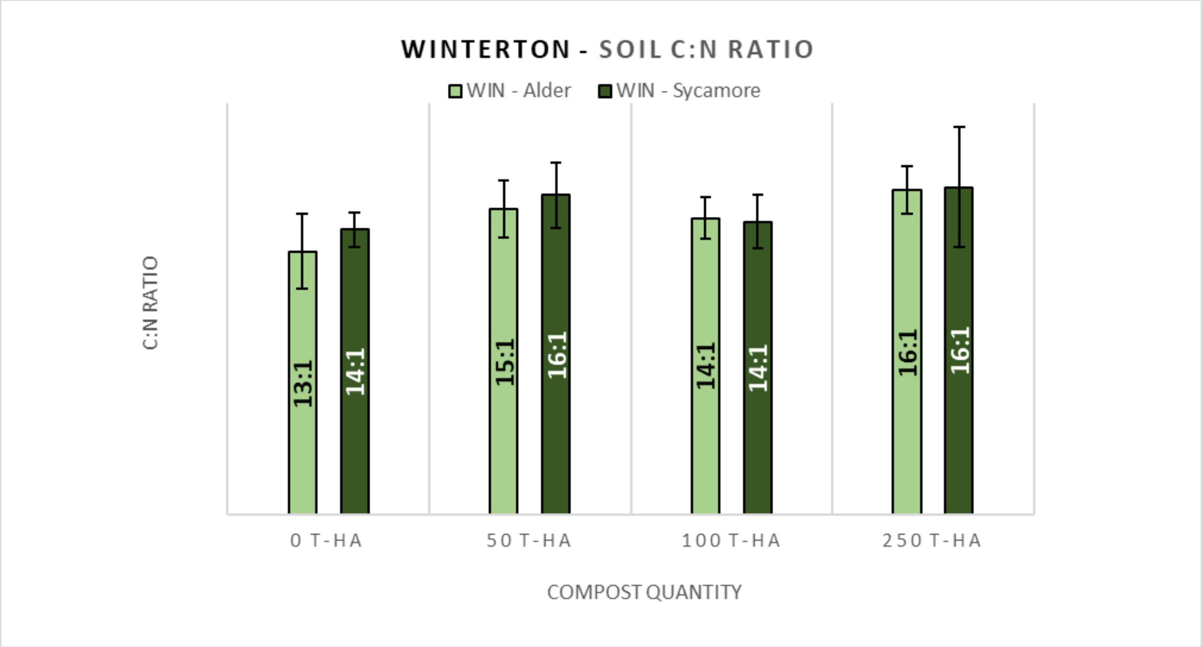


FIGURE 7.34 *– Impact of CGW quantity on soil C:N ratio. No significant differences were found between treatments. Soil C:N ratios did increase slightly with increasing CGW rate. The lowest soil C:N ratio recorded (13:1) was ALDER-CONTROL. Comparing soil values to foliar C:N in the previous figure, it is evident soil values do not correspond directly to foliar values. For example, SYCAMORE-CONTROL soils have a C:N ratio of (14:1), but foliar C:N is higher at (19:1). Thus, if foliar C:N is higher than that of soil, factors other than foliage must be altering soil chemistry.*

7.7.2 Impact of compost addition on soil physical quality

After 20-years of soil development at Winterton, soil physical parameters were assessed. Soil bulk density (SBD) served as a proxy indicator for soil formation, and some significant differences were found. Soil bulk density was significantly lower at 0-15 cm^{-1} sampling depth, compared with cores taken deeper down the profile i.e. 15-30 and 30-50 cm^{-1} (Kruskal-Wallis non-parametric ANOVA: H (5) = 109.4, p <0.001) **(FIGURE 7.35)**. Compared with Sycamore, SBD was also consistently lower under Italian alder, however Dunn's post-hoc test with bonferroni correction found differences between tree species were not significant. Altering compost quantity had no effect on bulk density, which suggests neither tree species nor compost alone altered topsoil density. Consequently, lower density topsoil at 0-15 cm^{-1} may be caused by multiple physical, chemical, and biological processes interacting over a period of 20-years.

Examining soil moisture (% water), similar patterns were found to those observed for bulk density. Soil moisture was significantly higher at 0-15 cm^{-1} sampling depth compared with cores taken deeper down the profile; Kruskal-Wallis non-parametric ANOVA: H (5) = 80.63, p <0.001. Indeed, at 0-15 cm^{-1} soil moisture was 6 – 7 % higher than at 15-30 and 30-50 cm^{-1} **(FIGURE 7.36).** In addition, soil moisture tended to increase alongside increasing quantities of compost, with both 100 $t\text{-}ha^{-1}$ and 250 $t\text{-}ha^{-1}$ treatments retaining 1 – 4 % more moisture (% water) than 50 $t\text{-}ha^{-1}$ treatments and controls: H (7) = 19.20, p 0.008 **(FIGURE 7.37)**. Dunn's post-hoc test revealed the increase in moisture recorded was also stronger under Italian Alder. It should be stated that the tendency of both Italian alder and compost to increase soil moisture after 20-years was only significant when bonferroni corrections were *not* applied to results **(FIGURE 7.37).**

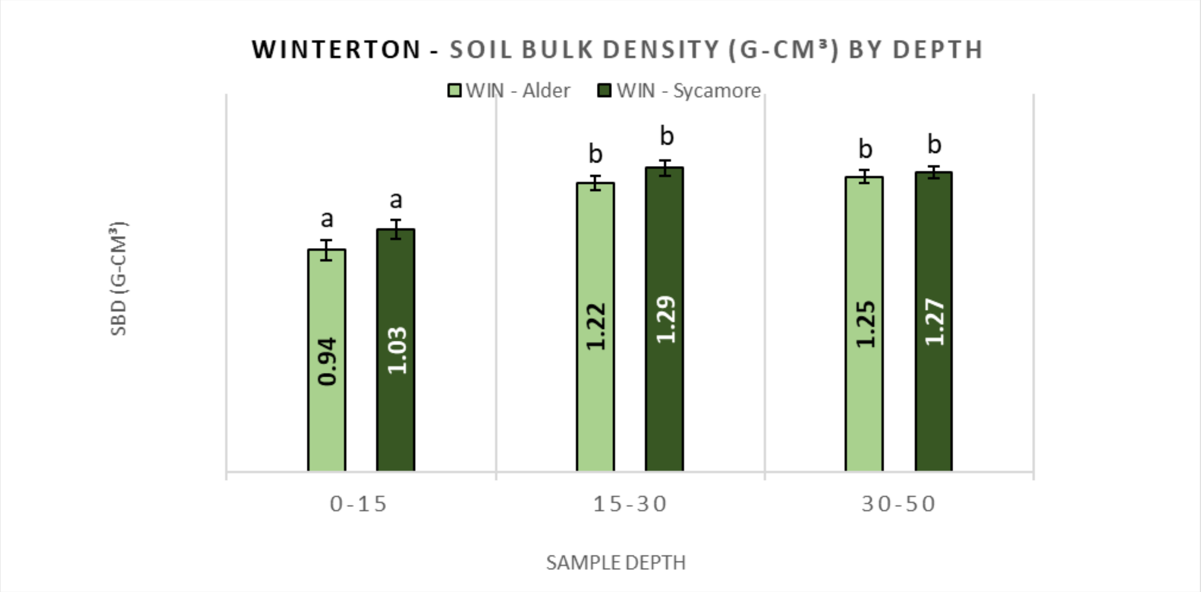


FIGURE 7.35 *– Changes in soil bulk density (SBD) at different sampling depth's. Where letters are not shared a significant difference exists (p<0.05). Kruskal-Wallis H test with bonferroni correction found SBD was significantly lower at 0-15 cm*$^{-1}$ *compared with all other sampling depths: H (5) = 109.4, p <.001. The effect was slightly stronger under Italian alder although differences between tree species were not significant.*

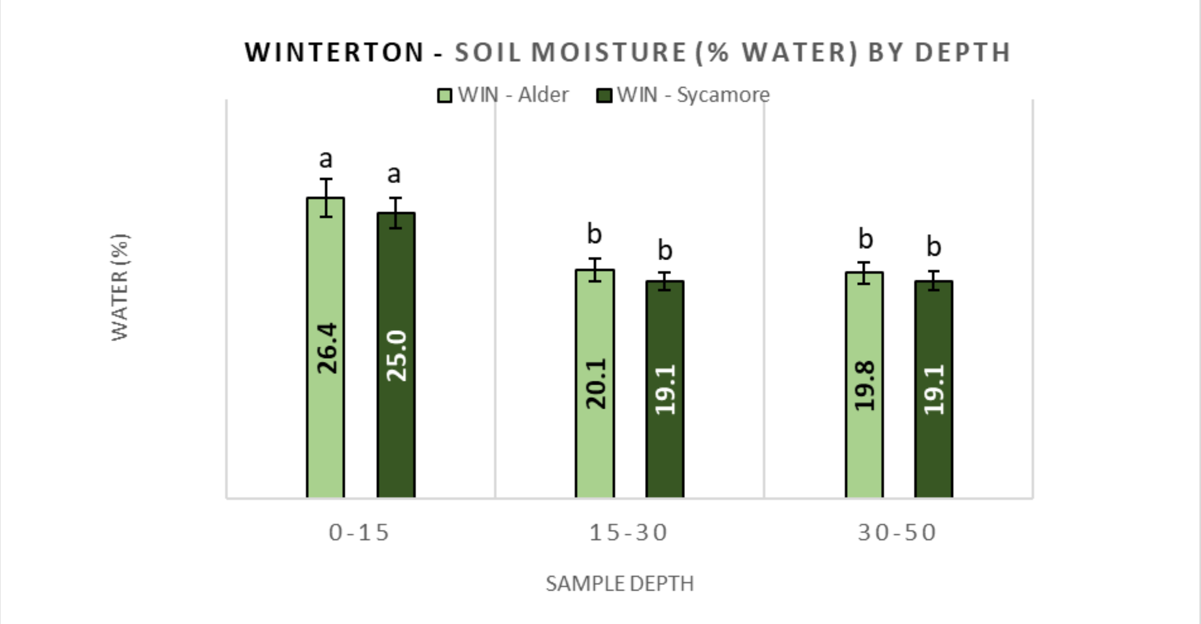


FIGURE 7.36 *– Changes in soil moisture (% water) at different sampling depth's. Where letters are not shared a significant difference exists (p<0.05). Kruskal-Wallis H test with bonferroni correction found soil moisture was significantly higher at 0-15 cm*$^{-1}$ *compared with all other sampling depths H (5) = 80.63, p <.001. Again, the effect was marginally stronger under Italian alder, although differences between tree species were not significant.*

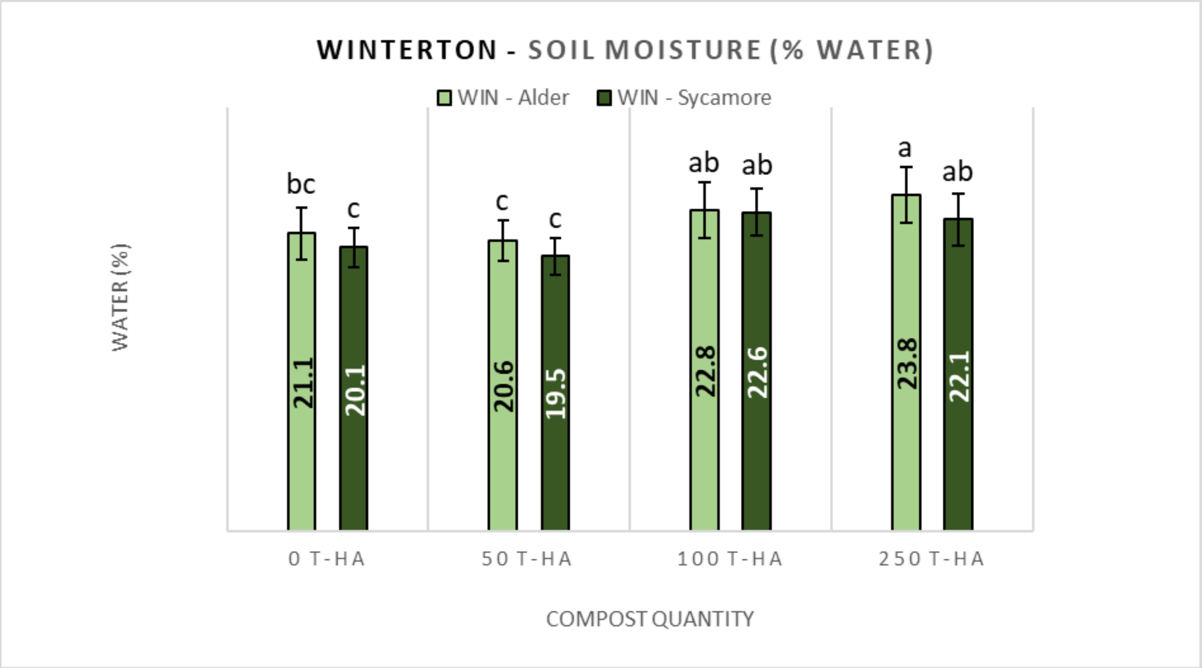


***FIGURE 7.37** – Changes in soil moisture (% water) with increasing quantities of compost. Where letters are not shared a significant difference exists ($p<0.05$). A Kruskal-Wallis H test found soil moisture increased significantly when more compost was applied H (7) = 19.20, p 0.008. However, Dunn's post-hoc test showed the effect was only significant when bonferroni correction was not applied. Nevertheless, for both ALDER and SYCAMORE the 100 t-ha^{-1} and 250 t-ha^{-1} treatments tended to retain significantly more moisture than 50 t-ha^{-1} applications and controls (p 0.004 – 0.05). The effect was also slightly stronger under ALDER on a consistent basis.*

7.7.3 Impact of compost addition on soil organic matter, soil nitrogen, and carbon storage

In November 2019, soil cores were collected from experimental plots at Winterton to evaluate CGW impacts on soil chemistry after 20-years. Cores provided the sample material from which data on soil organic matter (SOM), soil carbon, and soil nitrogen were generated. Regarding SOM a tree species effect was found, with Italian alder significantly increasing SOM content (+ 0.7 %) throughout Winterton: independent one-way ANOVA, F (1, 126) = 39.24, p <0.001.

Composts impact on SOM content was similarly investigated. One-way ANOVA followed by post-hoc test with bonferroni correction revealed significant treatment effects F (7, 120) = 6.706, p <0.001 **(FIGURE 7.38)**. The 'Alder-effect' was largely responsible, with ALDER-CONTROL having the highest SOM (6.0 %) and SYCAMORE-CONTROL the lowest (5.0 %). However, the exception was SYCAMORE 250 t-ha^{-1}, where SOM reached 5.5 %. Indeed, at this higher rate of compost, SOM in Sycamore plots was comparable to Alder **(FIGURE 7.38)**. In summary, the 'Alder-effect' raised SOM content by 0.7 % throughout Winterton. However, when 250 t-ha^{-1} was incorporated the 'compost-effect' diminished this gap, producing comparable SOM under either tree species.

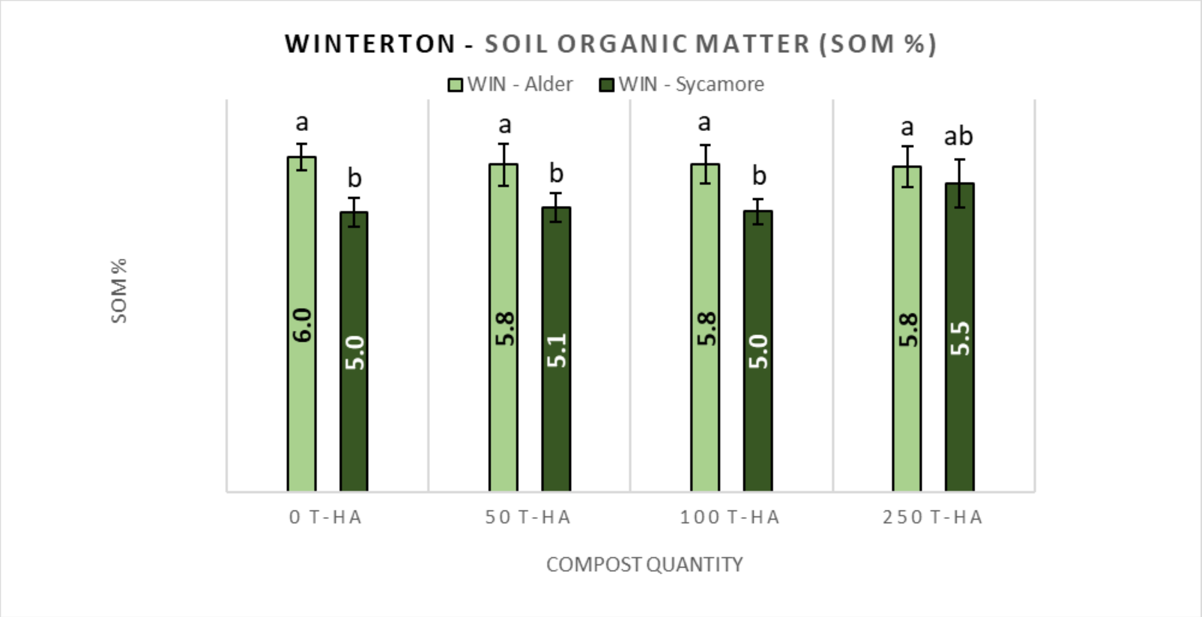


FIGURE 7.38 *– Impact of tree species and CGW quantity on soil organic matter content (SOM) at WINTERTON. Where letters are not shared a significant difference exists (p<0.05). An independent one-way ANOVA followed by post-hoc test with bonferroni correction found SOM was significantly higher under ALDER compared with SYCAMORE. However, when 250 t-ha^{-1} compost was applied differences in SOM between species were no longer significant F (7, 120) = 6.706, p <0.001.*

Total carbon (t-ha^{-1}) was assessed next, with independent one-way ANOVA used to check for differences in soil carbon between treatments. The result was marginally outside significance, F (7, 120) = 1.964, p <0.07. However, because the result was close a post-hoc test with bonferroni correction was still performed. This revealed just one significant difference. Namely, SYCAMORE 250 t-ha^{-1} treatments contained significantly more total soil carbon (+ 6.3 t-ha^{-1}) than SYCAMORE-CONTROL (p <0.04) **(FIGURE 7.39)**. Indeed, in reaching a soil total carbon value of 28.7 t-ha^{-1}, SYCAMORE 250 t-ha^{-1} achieved the highest soil carbon recorded throughout Winterton. Indeed, under Sycamore total carbon tended to increase with increasing compost rate whereas under Alder total carbon decreased with increasing compost rate **(FIGURE 7.39)**.

Regarding nitrogen, CGW is known to contain significant quantities of this vital element. Hence, compost addition is important for reclaimed soils especially when nitrogen capital is low. Thus, In 2019 total nitrogen (t-ha^{-1}) and available nitrogen (kg-ha^{-1}) were evaluated to understand composts ongoing influence after 20-years. For total nitrogen (t-ha^{-1}), similar trends to those observed for SOM and carbon were found. Namely, an 'Alder-effect' was present in control treatments. Indeed, ALDER-CONTROL had the highest total-N (2.3 t-ha^{-1}) recorded at Winterton, whilst SYCAMORE-CONTROL (1.6 t-ha^{-1}) had the lowest. These differences between controls were the only significant treatment effects identified for total nitrogen: one-way ANOVA followed by post-hoc test with bonferroni correction F (7, 120) = 2.114, p <0.05 **(FIGURE 7.40)**. However, a general pattern was evident under Sycamore, wherein total nitrogen increased with increasing compost rate. Conversely under Alder, total nitrogen decreased with increasing compost rate. A possible two-fold characterisation of these results would be that because ALDER-CONTROL achieved the highest total-N recorded at Winterton, ALDER-CONTROL highlights the 'Alder-effect'. Conversely, because SYCAMORE 250 t-ha^{-1} achieved this species highest total-N of 2.0 t-ha^{-1} which was comparable Alder's highest (ALDER-CONTROL - 2.3 t-ha^{-1}), Sycamore 250 t-ha^{-1} highlights the 'compost-effect' **(FIGURE 7.40)**.

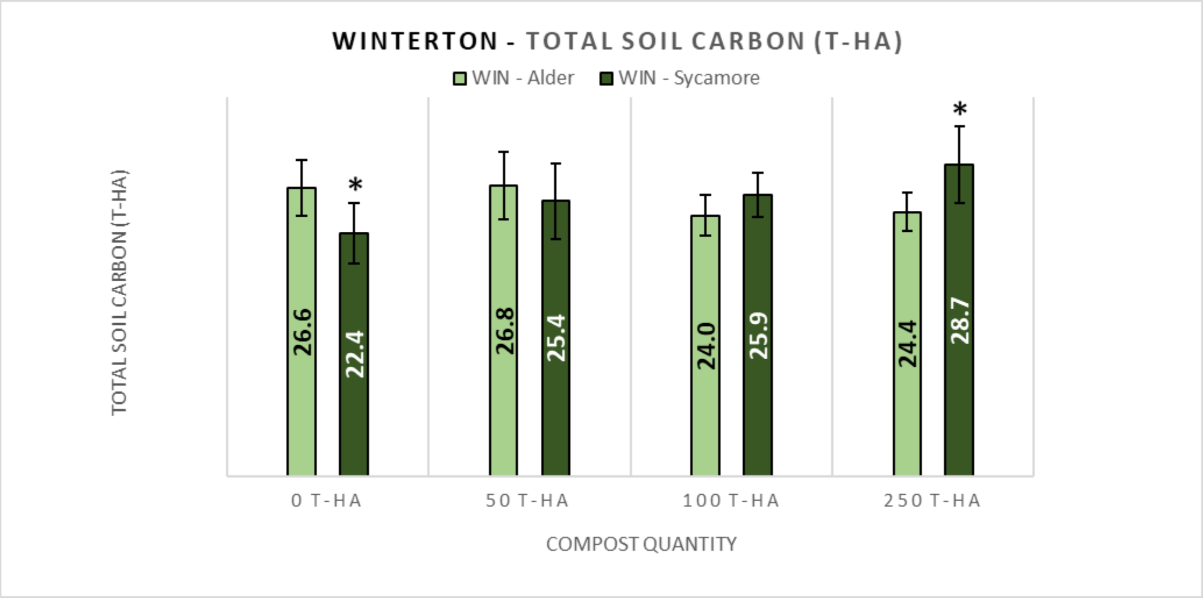


FIGURE 7.39 – *Impact of tree species and CGW quantity on soil total carbon ($t\text{-}ha^{-1}$) at WINTERTON. The only significant difference is marked by asterisks. Independent one-way ANOVA found differences between treatments were marginally outside statistical significance ($p<0.05$) F (7, 120) = 1.964, p <0.07. However, because the result was close, a post-hoc test with bonferroni correction was still performed. This revealed soils in SYCAMORE (250 $t\text{-}ha^{-1}$) treatments retained significantly more carbon than SYCAMORE-CONTROL ($p <0.04$), with the former achieving the highest soil carbon recorded at WINTERTON.*

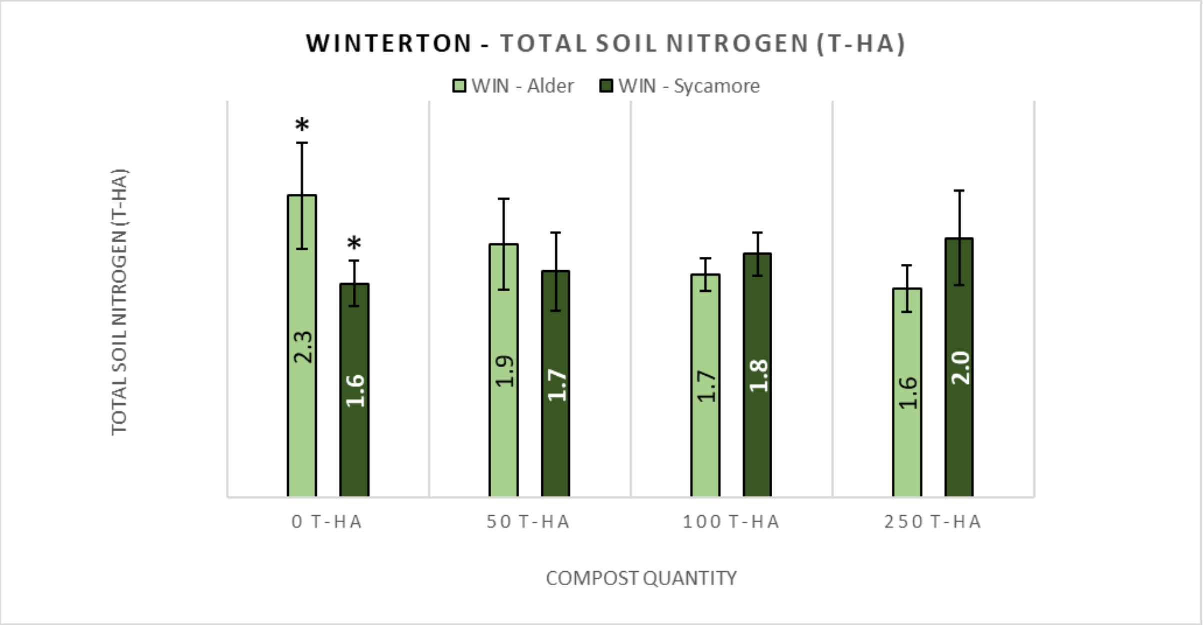


FIGURE 7.40 – *Impact of tree species and CGW quantity on soil total nitrogen (t-ha) at WINTERTON. The only significant difference is marked by asterisks. Independent one-way ANOVA followed by post-hoc test with tukey correction revealed one significant difference existed between treatments F (7, 120) = 2.114, p <0.047. Namely, ALDER-CONTROL treatments (0 $t\text{-}ha^{-1}$) contained significantly more total nitrogen than SYCAMORE-CONTROL TREATMENTS (0 $t\text{-}ha^{-1}$) ($p <0.046$). A general pattern was also evident, whereby changing the quantity of compost caused total nitrogen to decrease or increase incrementally. For SYCAMORE, increasing the quantity of compost caused a corresponding increase in total-N. Conversely for ALDER, increasing compost quantity caused a corresponding decrease in total-N.*

Finally, composts impact on available nitrogen (kg-ha^{-1}) shared a key similarity with data on total nitrogen. Namely, the difference in available nitrates (NO_3^-) between ALDER-CONTROL and SYCAMORE-CONTROL was the only significant difference recorded ($p<0.042$): one-way ANOVA followed by Dunn's post-hoc test with bonferroni correction F (7, 120) = 2.329, p <0.029) **(FIGURE 7.41)**. As with total-N, Winterton's highest available-N (0.42 kg-ha^{-1}) was found in ALDER-CONTROL. Conversely, Winterton's lowest available-N (0.26 kg-ha^{-1}) was recorded in SYCAMORE-CONTROL. Indeed, it was expected that ALDER 250 t-ha^{-1} treatments would have the highest available nitrates (NO_3^-). However, the 250 t-ha^{-1} treatments had Italian alders lowest recorded available-N, being equivalent to SYCAMORE-CONTROL and ALDER-50 t-ha^{-1} **(FIGURE 7.41)**.

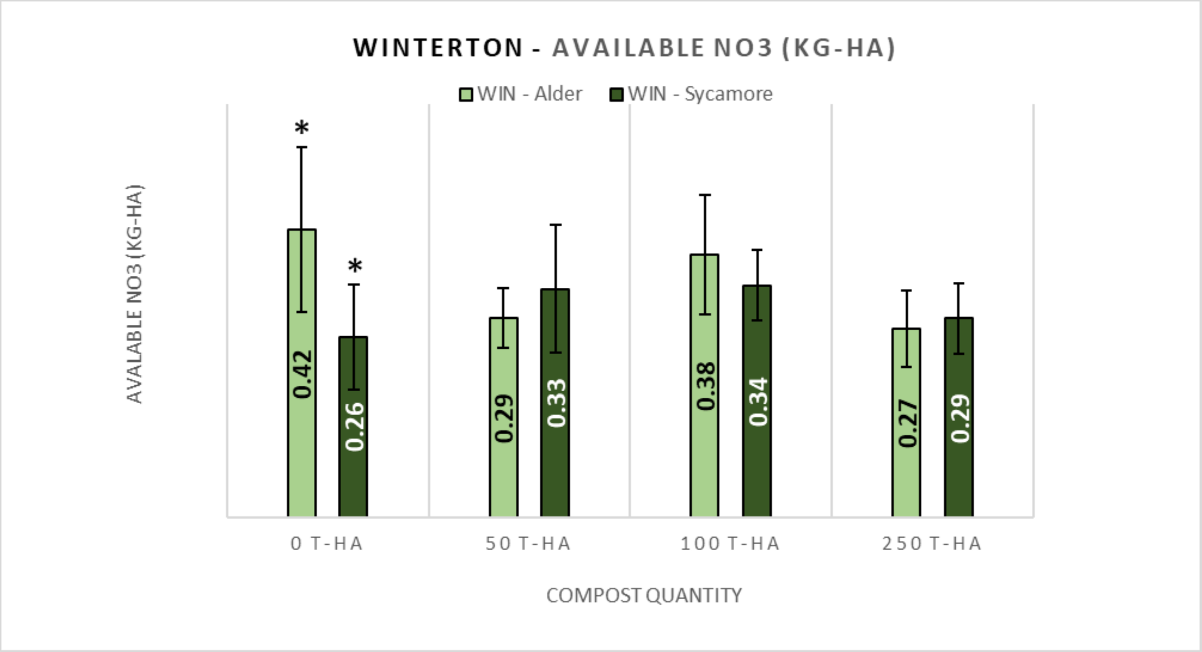


***FIGURE 7.41** – Impact of tree species and CGW quantity on soil available nitrogen (kg-ha^{-1}) at WINTERTON. The only significant difference is marked by asterisks ($p<0.042$). Independent one-way ANOVA followed by post-hoc test with bonferroni correction found soils in ALDER CONTROL contained significantly more available nitrogen than SYCAMORE CONTROL F (7, 120) = 2.329, p <0.029. This was also the case for total nitrogen (t-ha^{-1}).*

In summary, when compost was absent at Winterton, the ‘Alder-effect’ on soil chemistry was more pronounced. Indeed, Alder values for SOM (6.0 %), total nitrogen (2.3 t-ha^{-1}), and available nitrates (0.42 kg-ha^{-1}) reached their highest without compost addition. Conversely for Sycamore, 250 t-ha^{-1} treatments provided clear benefits. Sycamore grew taller (7.3 m^{-1}), achieved its highest survival (80 %), stored more SOM (5.5 %), more total carbon (28.7 t-ha^{-1}), and more total nitrogen (2.0 t-ha^{-1}) when 250 t-ha^{-1} compost was applied. Thus, different soil dynamics can manifest, depending on interactions between the compost and tree species effect.

7.7.4 Impact of compost addition on earthworm populations and soil development

Earthworms were sampled at Winterton in Sept-October 2018, 21-years after the experiment was established. Overall, monitoring found Italian alder's presence, plus increasing quantities of compost elevated abundance of epigeic and endogeic earthworms. Firstly, regarding earthworm community structure and diversity, eight species were found at Winterton, all at moderate to low levels. Mean densities for the eight species were: *Aporrectodea rosea* (15 m^{-2}), *Aporrectodea longa* (13 m^{-2}), *Lumbricus castaneus* (9 m^{-2}), *Aporrectodea caliginosa* (9 m^{-2}), *Allolobophora chlorotica* (8 m^{-2}), *Lumbricus rubellus* (7 m^{-2}), *Lumbricus terrestris* (4 m^{-2}), *Eiseniella tetraeda* (0.3 m^{-2}). In total, earthworm community density was (65 m^{-2}).

Examining the impact of tree species on earthworm abundance, **FIGURE 7.42** and **FIGURE 7.43** show endogeic (*A. rosea*, *A. caliginosa*) and epigeic (*L. castaneus*) abundance near doubled under ALDER. Indeed, Kruskal-Wallis H-test found this effect was significant, with Italian alder significantly increasing abundance of certain ecological groups: H (5) = 44.95, p <0.001 **(FIGURE 7.43)**. Specifically, endogeic density was greater under ALDER (40 m^{-2}) compared to SYCAMORE (24 m^{-2}), with Alder-endogeic greater than every other tree species x earthworm eco-group combination (p 0.001 – p 0.047). Evidently, the 'Alder-effect' was particularly strong, doubling endogeic density compared to SYCAMORE.

The impact of compost quantity on eco-group abundance was assessed next to ascertain whether higher rates of compost would produce a long-term increase in earthworm abundance **(FIGURE 7.44)**. Kruskal-Wallis H-test found compost quantity did have a significant impact on earthworm eco-group abundance: H (11) = 50.41, p <0.001. Specifically, by increasing compost quantity to 100 t-ha^{-1} or 250 t-ha^{-1}, epigeic abundance increased accordingly. Indeed, Dunn's post-hoc test with bonferroni correction showed compost increased epigeic abundance from 10 m^{-2} in CONTROL, to 23 m^{-2} when 250 t-ha^{-1} compost was applied (p 0.001 – p 0.03) **(FIGURE 7.44).** Endogeic abundance similarly responded when more compost was added, from (26 m^{-2}) in 50 t-ha^{-1}

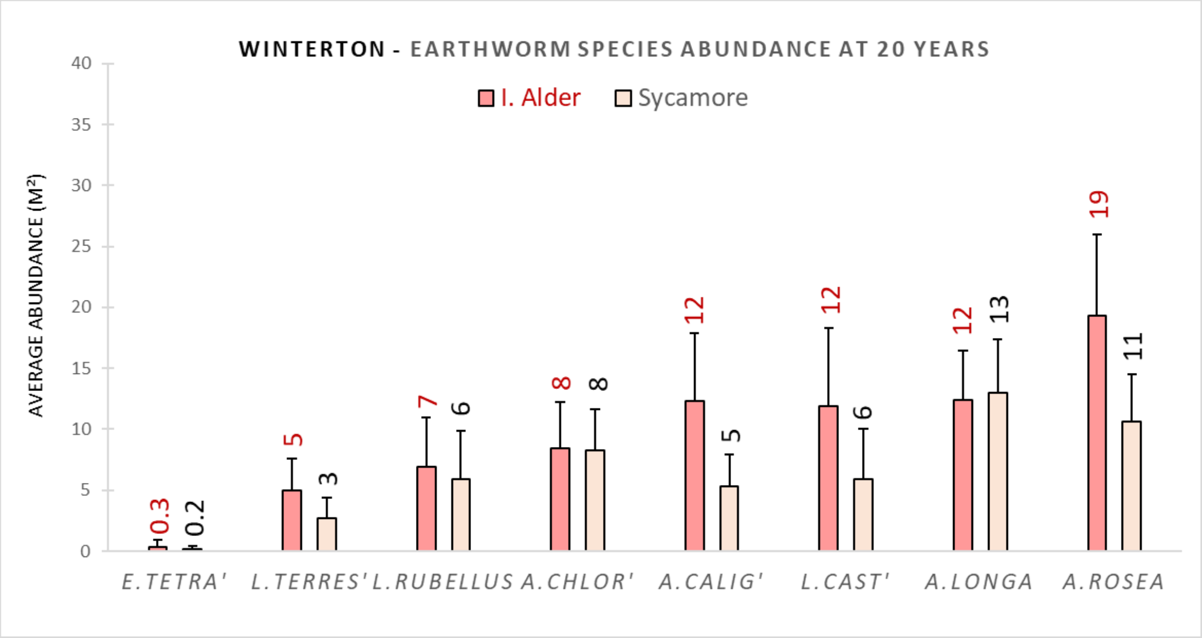


FIGURE 7.42 – Earthworm species abundance (m^{-2}) at WINTERTON following 21-years of site development. For many earthworm species, abundance was similar regardless of tree species effects. However under ALDER, the density of A. rosea, A. caliginosa, and L. castaneus practically doubled suggesting the presence of an 'Alder-effect'.

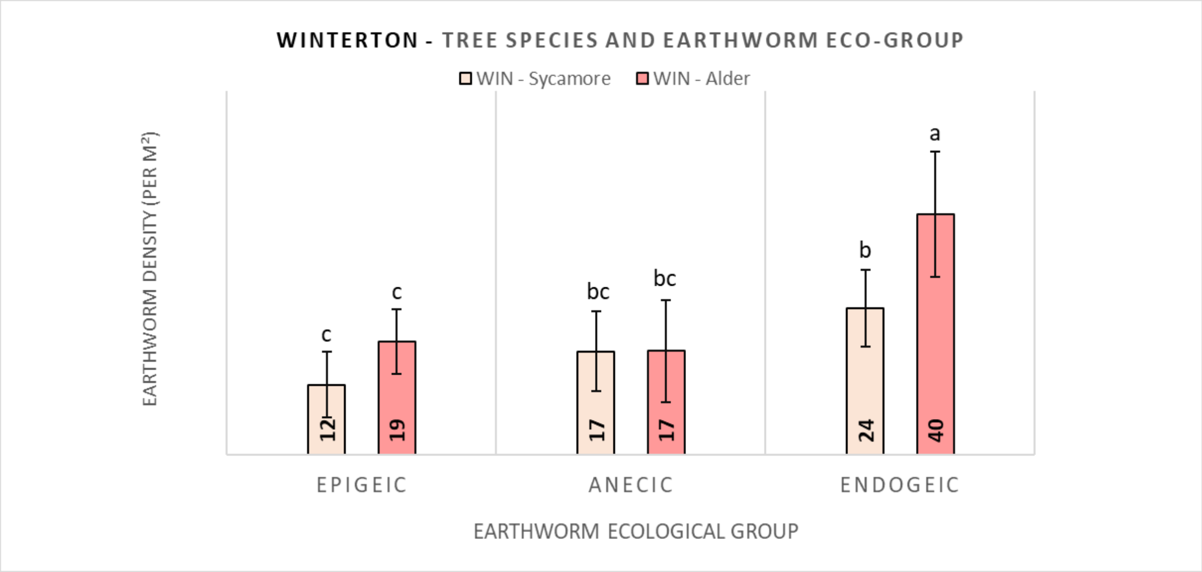


FIGURE 7.43 – Impact of tree species on abundance of earthworm ecological group (m^{-2}). Where letters are not shared a significant difference exists ($p<0.05$). Kruskal-Wallis H test found differences between tree species were significant H (5) = 44.95, p <0.001. Dunn's post-hoc test with bonferroni correction revealed endogeic densities were significantly higher under ALDER compared with SYCAMORE suggesting the presence of an 'Alder-effect' (p 0.001 – p 0.047).

treatments, to (36 m^{-2}) in the 250 t-ha^{-1} treatments, although the impact of CGW quantity on endogeic populations was not significant. Finally, anecic populations responded to compost quantity. However, unlike epigeic and endogeic populations, anecic species abundance *decreased* following compost addition **(FIGURE 7.44)**, likely in response to expansion of the other ecological groups. For example, anecic abundance was high in CONTROL (23 m^{-2}), being significantly greater than epigeic abundance (10 m^{-2}) and statistically comparable to the dominant endogeic populations (30 – 37 m^{-2}). Yet following compost addition anecic abundance declined, altering the levels of significance between anecics and other eco-groups **(FIGURE 7.44)**.

Finally, incorporation depth influenced earthworm abundance, with SHALLOW (0.1 m^{-1}) and DEEP (0.6 m^{-1}) treatments each producing different dynamics that were strong under ALDER, but weak under SYCAMORE. For example, Kruskal-Wallis H-test found incorporation depth significantly impacted earthworm eco-group abundance $\underline{H\ (11) = 48.52,\ p < 0.001.}$ Yet whereas ALDER-DEEP increased abundance of all three eco-groups, ALDER-SHALLOW increased abundance of endogeics only **(FIGURE 7.45)**. These differences become apparent when examining DEEP and SHALLOW incorporations impact on levels of statistical significance. In ALDER-DEEP, there are no significant differences between groups because all are similarly abundant. Conversely, in ALDER-SHALLOW there is a significant difference between groups because only endogeics are abundant **(FIGURE 7.45)**. The likely reason is a combination of ALDER plus DEEP compost incorporation provides abundant resources and spatial habitat to 0.6 m^{-1} depth thus accommodating all three eco-groups. Conversely, in ALDER-SHALLOW there is no compost below 0.1 m^{-1} meaning organic matter and spatial habitat are confined to the uppermost soil horizon and only endogeics proliferate. An overview of composts impact on tree growth, soil development, and earthworms is now presented using visual outputs from principal component analysis (PCA).

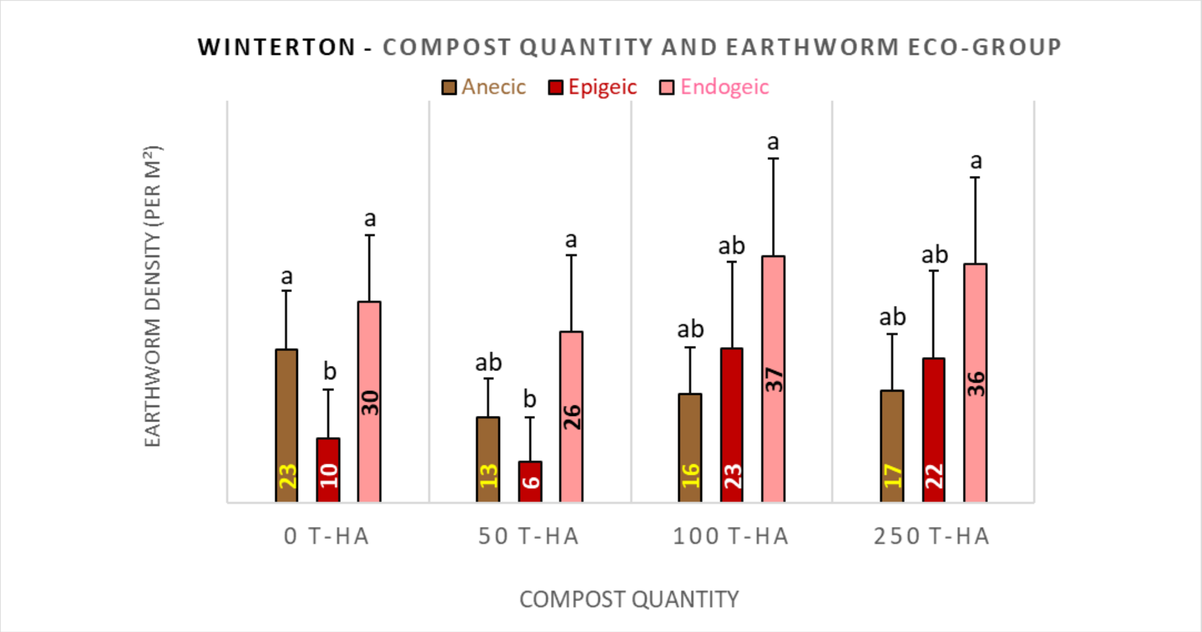


FIGURE 7.44 – *Impact of compost quantity on abundance of earthworm ecological group (m^{-2}). Where letters are not shared a significant difference exists ($p<0.05$). Kruskal-Wallis H-test found compost quantity had some significant impacts on eco-group density: H (11) = 50.41, $p<0.001$. When compost quantity rises from 0 and 50 t-ha^{-1}, up to 100 t-ha^{-1} or 250 t-ha^{-1} epigeic and endogeic abundance similarly rises. For epigeics the effect was significant. For endogeic the effect was not significant. In contrast, compost addition caused anecic abundance to decline.*

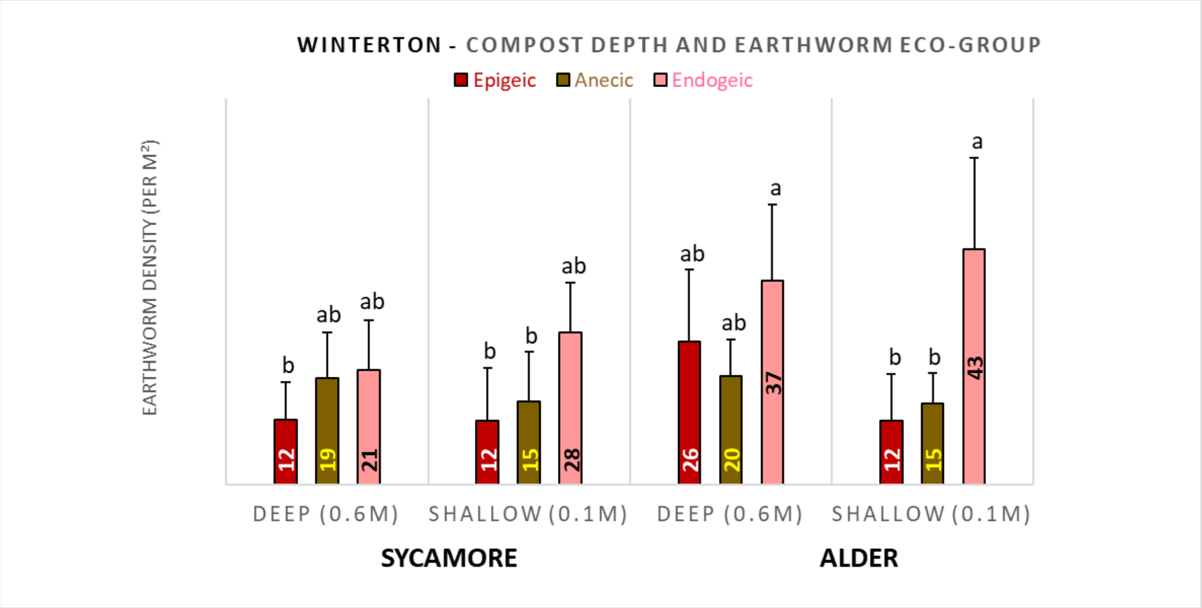


FIGURE 7.45 – *Impact of incorporation depth on abundance of earthworm ecological group (m^{-2}). Where letters are not shared a significant difference exists ($p<0.05$). Kruskal-Wallis H test found incorporation depth had a significant impact on earthworm abundance H (11) = 48.52, $p<0.001$. These impacts can be understood by examining changes in levels of statistical significance between ALDER-DEEP and ALDER-SHALLOW. For example, in ALDER-DEEP endogeic abundance is high and yet there are no significant differences between eco-groups because all are relatively abundant. Conversely, in ALDER-SHALLOW there are significant differences between eco-groups and only endogeics are abundant (p 0.001 – p 0.05). In essence, ALDER-DEEP increased the abundance of all three ecological groups, whereas ALDER-SHALLOW increased abundance of just one ecological group: endogeic earthworms.*

7.7.5 Overview of results

Principal component analysis (PCA) was used to evaluate the various categories of site data obtained between June 2018 and February 2019. The analysis produced graphical outputs known as 'biplots' and provided a visual summary of site dynamics. Using these, the interplay between compost, tree growth, earthworms, and soil conditions was evaluated by qualitatively analysing the distribution of data points throughout the biplots. For example, a clustering of data points indicated greater association among variables. Vector length indicated effect size and strength. Vector direction and angles between vectors indicated the nature of a relationship, e.g. when following similar trajectories variables are closely associated, but when at opposite ends of 180° angles variables have 'opposing' or 'polar' correlations, whereas vectors at right angles have no correlation or relationship. The PCA biplots in **FIGURE 7.46** and **FIGURE 7.47** show that, despite a degree of overlap among data points, data separates according to tree species at 0-15 cm^{-1} and 15-30 cm^{-1} sampling depths. Consequently a 'tree-species-effect' is present but is not powerful or prominent enough to completely separate Alder and Sycamore data points.

Firstly, regarding the 0-15 cm^{-1} sampling depth, 'Porosity', 'Water', 'SOM', 'Soil N', 'Endogeic', and 'Epigeic', associate more strongly with ALDER. Conversely, 'Foliar C:N', 'SBD', and 'Anecic' tend to associate more strongly with SYCAMORE. This means soils in ALDER plots are more porous, moist, contain more soil organic matter and nitrogen, and have greater abundance of endogeic and epigeic earthworms than those under SYCAMORE. Conversely, soils in SYCAMORE plots are denser, contain fewer earthworms, but despite this have a greater abundance of anecic earthworms **(FIGURE 7.46)**. Regarding overlap, 'Soil C' and 'NO3' tended towards ALDER although some values were comparable or greater under SYCAMORE. 'Height' and 'DBH' also require attention, as these tree growth variables disassociated from the core group of soil variables, being situated at right angles from soil variables and opposite 'Endogeic'. This means even though most soil and earthworm variables associate with ALDER, Italian Alder growth may not be wholly responsible.

Regarding Winterton's 15-30 cm^{-1} sampling depth, separation according to tree species remained prevalent with many of the same variables driving this trend. For example, 'Porosity', 'Water', 'SOM', 'Soil N', and 'Epigeic', continue to associate with ALDER at this lower depth. Similarly, 'Foliar C:N', 'SBD', and 'Anecic' continue to associate with SYCAMORE plots. Meanwhile, 'Height' and 'DBH' remain disconnected from the core group of soil related variables. Yet despite these similarities a key difference is evident. Namely, at 15-30 cm^{-1} depth 'Endogeic' earthworms no longer associate with ALDER, meaning the relationship between endogeics and ALDER exists chiefly in the uppermost 0-15 cm^{-1} of soil **(FIGURE 7.47)**. This is likely related to CGW incorporation depth, with endogeic density always greater in SHALLOW incorporations especially under ALDER. Results from Immingham are now presented.

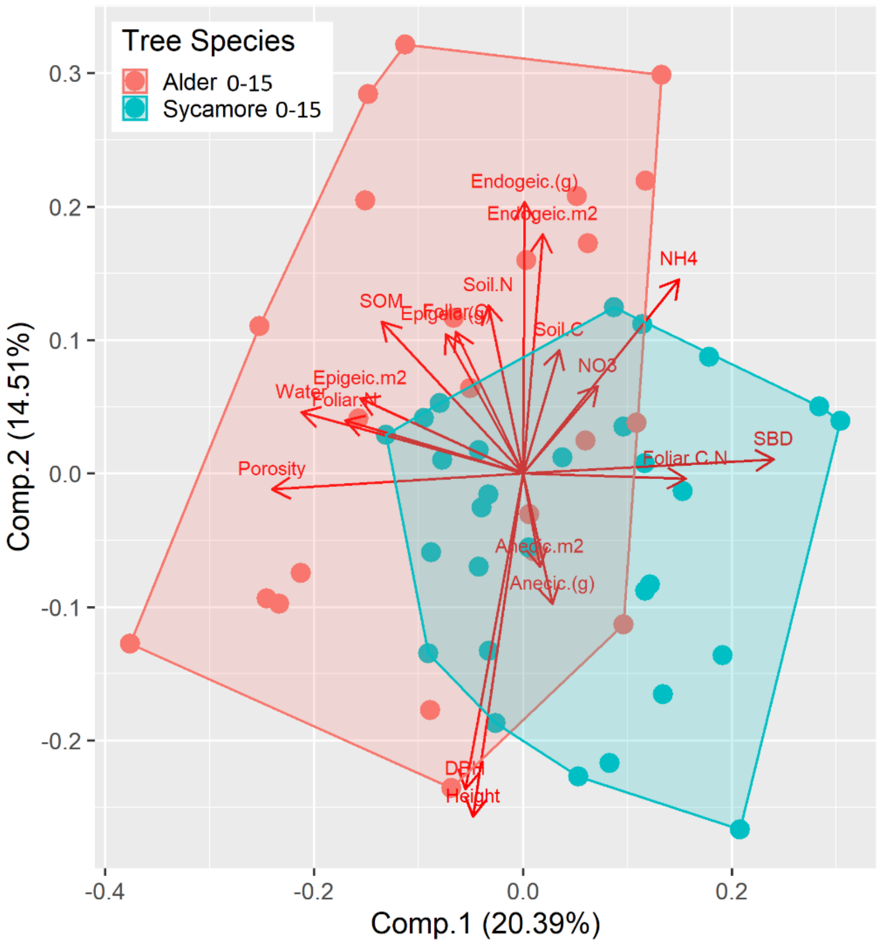


***FIGURE 7.46** – PCA examining soil physical, soil chemical, and tree growth related variables at Winterton (<u>0-15cm sampling depth</u>). Overall, the data shows evidence of separation according to tree species, with* PCA 1 versus PCA 2 accounting for around 35% of the total variation observed at this depth. *This indicates that a 'tree species effect' is present, although tree species do overlap. Increasing 'Porosity', Tree 'Height', 'DBH', 'SOM', and 'Endogeic' and 'Epigeic' earthworm numbers are associated with ALDER. Meanwhile, increasing 'SBD', 'Foliar C:N', and 'NH4' are more closely associated with SYCAMORE.*

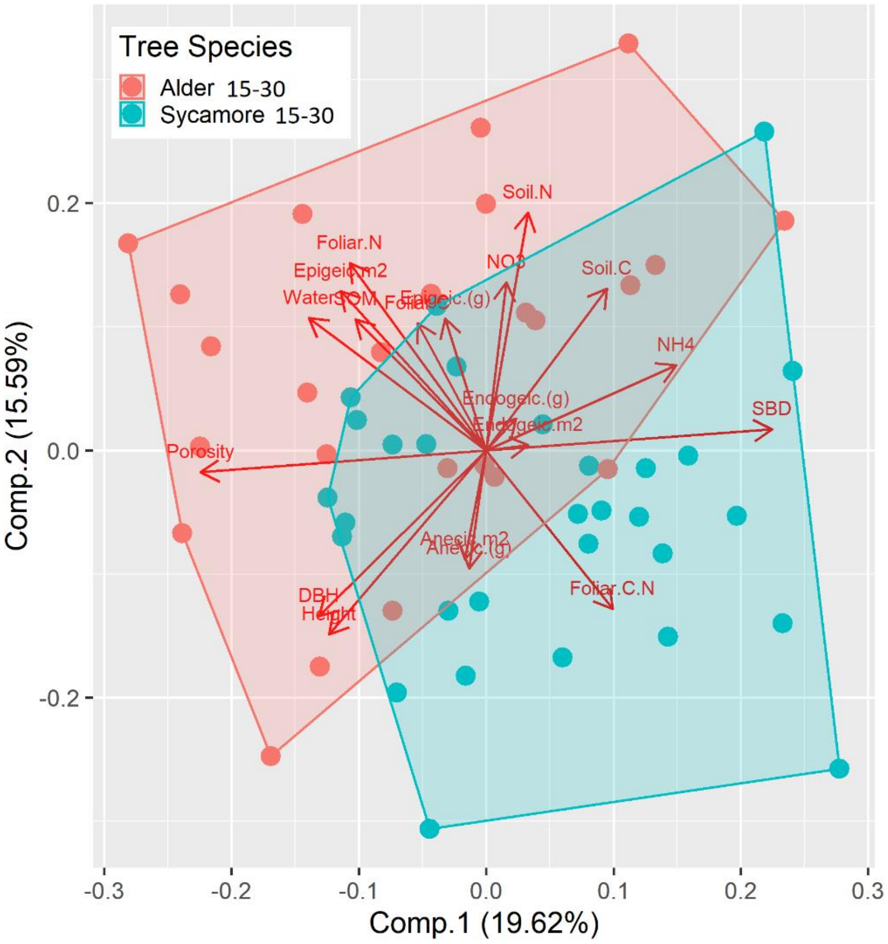


***FIGURE 7.47** – PCA examining soil physical, soil chemical, and tree growth related variables at Winterton (<u>15-30 cm sampling depth</u>). Once again, the data shows evidence of separation according to tree species, with PCA 1 versus PCA 2 accounting for around 35%* of the total variation observed. *Increasing 'Water' (i.e. soil moisture), Tree 'Height', 'DBH', 'SOM', and 'Epigeic' earthworm numbers associate with ALDER. Increasing 'SBD', 'Foliar C:N', and 'NH4' again associate with SYCAMORE.*

7.8 Results - Immingham

7.8.1 Impact of compost addition on tree survival, growth, and foliar nutrition

When the Humberside experiments began in 1998, survival rates at Immingham were poor for both tree species and most composted green waste (CGW) treatments. For example, Italian alder survival was 11 % in 1998, with only SYCAMORE-DEEP maintaining high survival at this time (+ 90 %). Consequently, dead trees were replaced and losses sustained in 1998 were not repeated in subsequent years. In fact, when the original experiment concluded in 2001, Immingham had the highest survival of all three experiments (Foot et al., 2003). Monitoring in 2019 found this had continued with 81 % of trees recorded in 2001 still present in 2019 **(FIGURE 7.48, FIGURE 7.49, FIGURE 7.50, FIGURE 7.51).** Consequently in 2019, survival rates at Immingham were far greater than at Winterton, particularly where Italian alder was concerned.

However, despite greater Italian alder survival at Immingham, compost did not have an obvious impact on tree survival. Nevertheless, a general pattern was observed with some statistically significant evidence in support. Namely, Italian alder and Sycamore survival were higher when: SHALLOW incorporations using moderate quantities of compost were applied (50, 100 t-ha^{-1}). This is outlined by a chi-squared (χ^2) test for independence which predicted 31 of 40 trees (79%) would survive in each treatment. However actual data differed significantly from this in two instances: $\underline{\chi^2 (15, 640) = 30.60, p < 0.01}$ **(TABLE 7.4).** The first was in the 50 t-ha^{-1} SYCAMORE-SHALLOW treatment where survival was significantly higher than expected. Indeed, 39 of 40 trees (97.5 %) remained in 2019 representing the highest survival recorded at Immingham **(FIGURE 7.51).** Conversely, in ALDER-SHALLOW (250 t-ha^{-1}) tree survival was significantly lower than expected, with only 23 of 40 trees (57.5 %) remaining in 2019 representing the lowest survival recorded at Immingham **(FIGURE 7.49).** Together, these examples show SHALLOW incorporations result in high tree survival provided a moderate amount of compost (50 t-ha^{-1}) is used, but when large quantities (250 t-ha^{-1}) are used survival is lower overall.

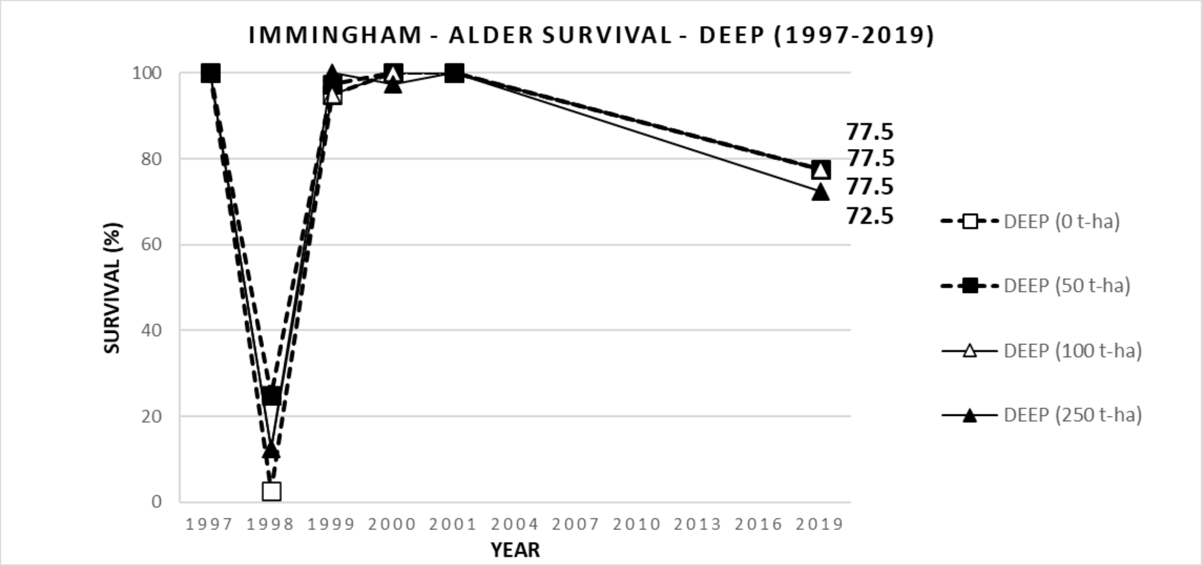


FIGURE 7.48 *– Italian Alder (%) survival at IMMINGHAM in DEEP (0.6 m) compost incorporations from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise throughout the early years. In 2019 survival was similar across all treatments. A chi-square (χ^2) test for independence found no significant differences between predicted and actual survival rates for any treatment.*

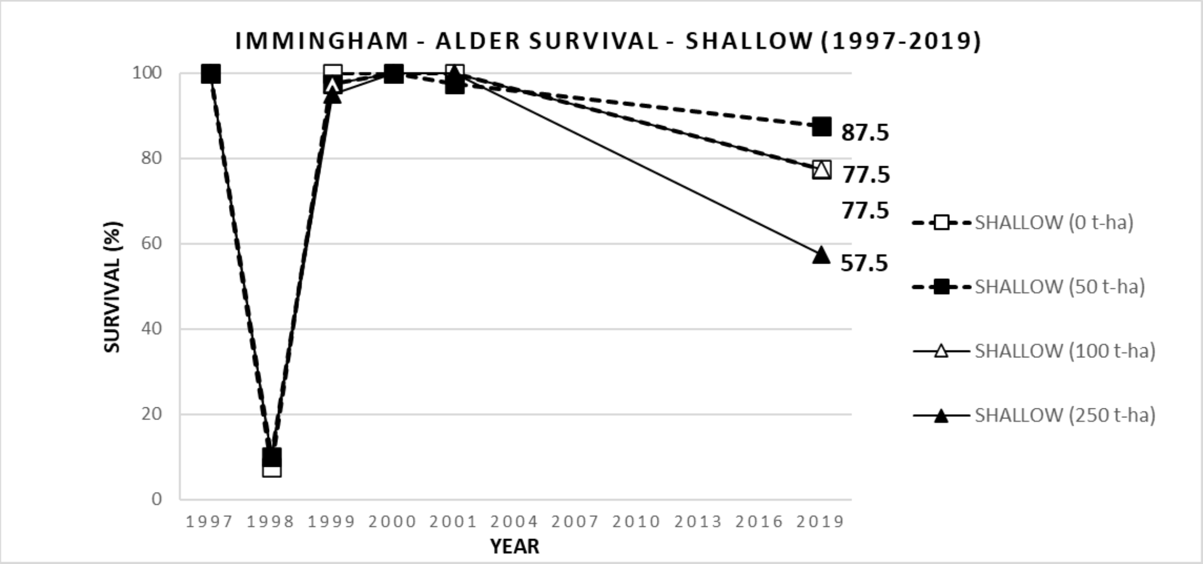


FIGURE 7.49 *– Italian Alder (%) survival at IMMINGHAM in the SHALLOW (0.1 m) incorporation from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise early on. In 2019, survival was highest in SHALLOW (50 t-ha^{-1}), and lowest in SHALLOW (250 t-ha^{-1}). Indeed, chi-square (χ^2) test for independence found survival in SHALLOW (250 t-ha^{-1}) was significantly lower than the predicted survival rate: χ^2 (15, 640) = 30.60, $p < 0.01$.*

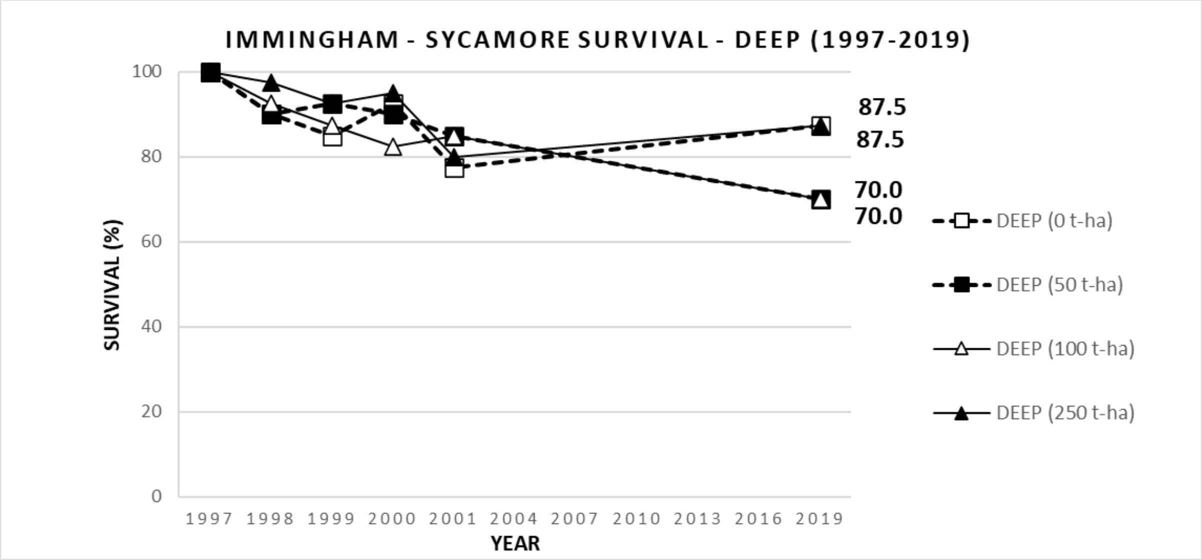


FIGURE 7.50 – Sycamore (%) survival at IMMINGHAM in the DEEP (0.6 m) compost incorporation from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise throughout the earliest years. In 2019 survival was similar across all treatments. A chi-square (χ^2) test for independence found no significant differences between predicted and observed survival rates for any treatment.

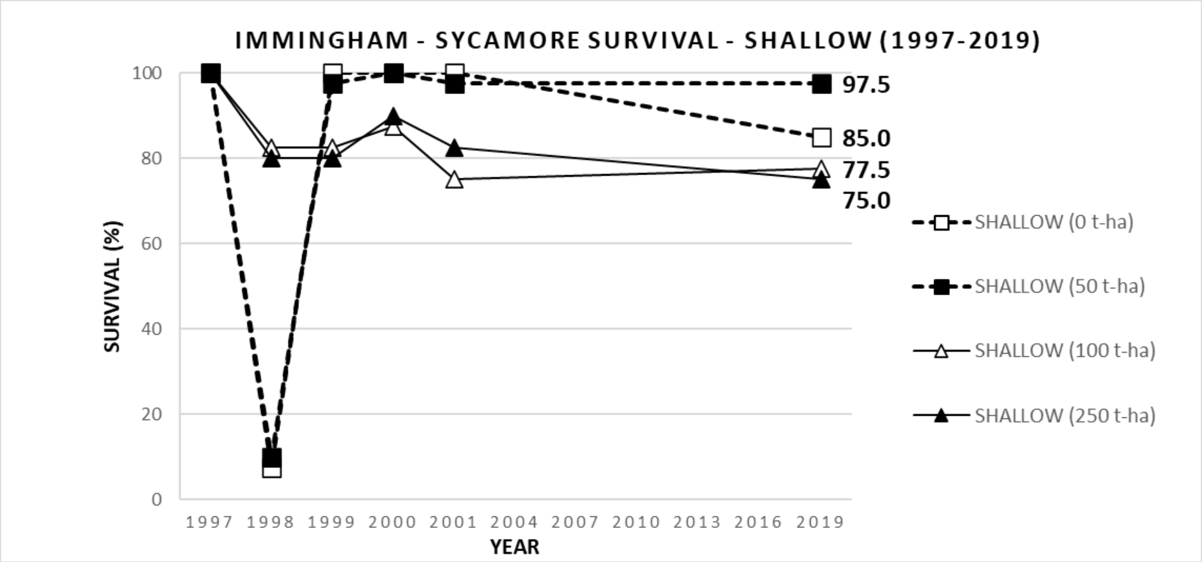


FIGURE 7.51 – Sycamore (%) survival at IMMINGHAM in the SHALLOW (0.1 m) incorporation from 1997 to 2019. Dead trees were replaced annually from 1998 to 2000 causing survival to fall and rise early on. In 2019, survival was highest in SHALLOW (50 t-ha^{-1}) treatment, and lowest in SHALLOW (250 t-ha^{-1}). Indeed, chi-square (χ^2) test for independence found survival in SHALLOW (250 t-ha^{-1}) was significantly lower than predicted: χ^2 (15, 640) = 30.60, $p < 0.01$.

IMMINGHAM - TREE SURVIVAL (20 years)

Tree / CGW-Depth	CGW (t-ha)	Predicted	Actual	% Survival	Higher	Lower	z-score	p = value
Alder Shallow	250	31.44	23	57.5		↓	2.88	0.01
Sycamore Shallow	100	31.44	39	97.5	↑		2.58	0.01

TABLE 7.4 *– Impact of treatment on predicted survival at IMMINGHAM. Chi-square (χ^2) test for independence found predicted survival was significantly different from observed survival in two instances: χ^2 (15, 640) = 30.60, $p < 0.01$. RED arrow indicates lower than expected survival. GREEN arrow indicates higher than expected survival. Survival tended to be higher when SHALLOW incorporations using moderate quantities of compost were applied.*

Tree height and stem diameter at breast height (DBH) were similarly recorded in February 2019. The main finding was Italian alder grew significantly taller than Sycamore throughout Immingham as outlined by a Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction: H (7) = 259.7, p 0.001. Further, Italian alder grew 0.9 – 1.9 m^{-1} taller at Immingham compared with Winterton **(FIGURE 7.52, FIGURE 7.54)**. However, despite significant differences in tree height and stem diameter between species, compost quantity had no influence on this with Alder and Sycamore growth relatively uniform regardless of compost application rate.

Although CGW quantity did not impact tree growth, incorporation depth did have a significant impact. Namely, trees in ALDER-DEEP grew significantly taller (+0.7 m) than trees in ALDER-SHALLOW: (Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction H (3) = 263.2, p 0.001) **(FIGURE 7.53).** Furthermore, according to **TABLE 7.5** many growth increments occurred after 2001, suggesting compost had some legacy effects on tree growth. For example, when original measurements concluded in 2001, trees planted in ALDER-DEEP were only 0.08 m taller than those planted in ALDER-SHALLOW. To create the +0.7 m difference recorded 18-years later, the vast majority of growth (0.62 m^{-1}) must have occurred post 2001. In summary, tree growth results from Immingham show Italian alder continues to dominate the overstory after 20-years. Compared with Winterton, at Immingham Alder survival was high (+ 70 %) and growth taller (+ 1 – 2 m^{-1}). DEEP incorporation also improved Alder growth (but not Sycamore). Indeed, despite high Sycamore survival (78 %), Alder's continued dominance confined Sycamore to the understory restricting its growth, resulting in Sycamore being 0.7 – 1.1 m^{-1} shorter at Immingham.

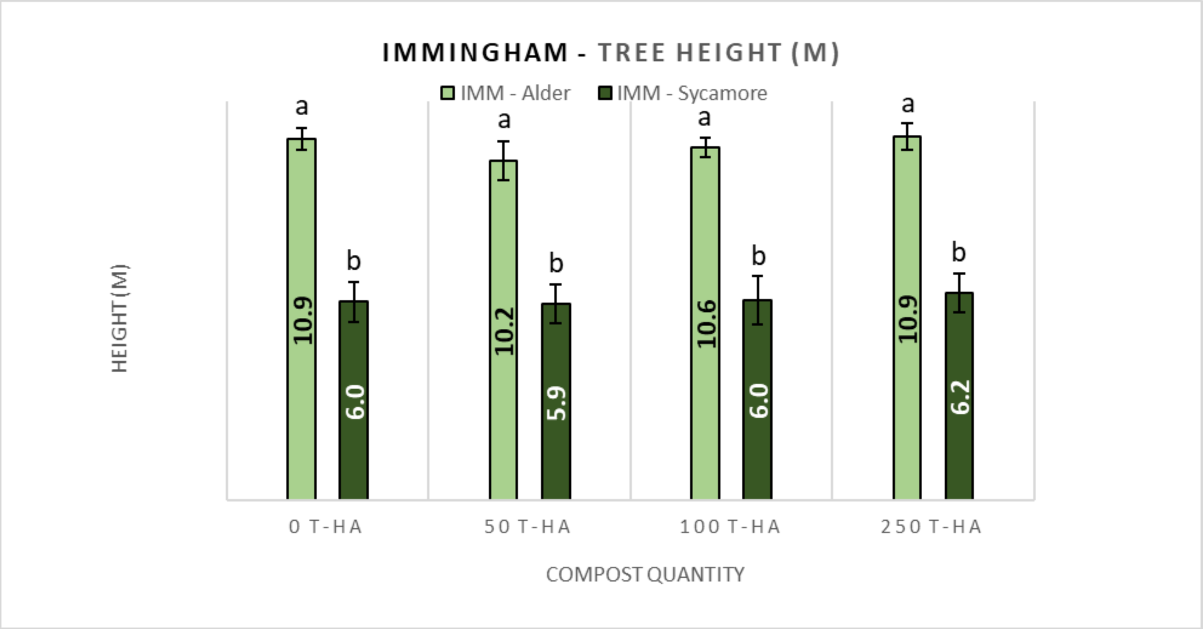


FIGURE 7.52 – *Impact of CGW quantity on tree height. Where letters are not shared a significant difference exists (p<0.05). CGW quantity had no significant impact on the height of either species. Nevertheless, a Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction confirmed that all ALDER treatments were significantly taller than all SYCAMORE treatments* *H (7) = 259.7, p 0.001. ALDERS fast growth habit meant it reached a greater height than SYCAMORE over the experiments duration.*

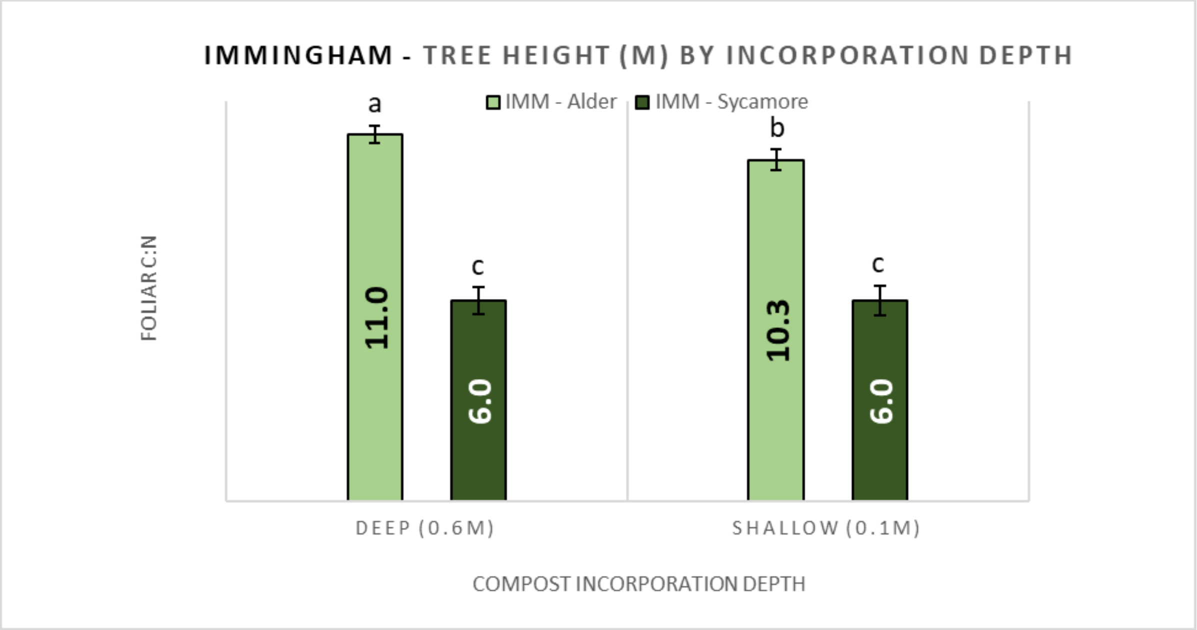


FIGURE 7.53 – *Impact of compost incorporation depth on tree height. Where letters are not shared a significant difference exists (p<0.05). Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction revealed ALDER DEEP grew significantly taller than ALDER SHALLOW, and taller than SYCAMORE throughout the experiment H (3) = 263.2, p 0.001.*

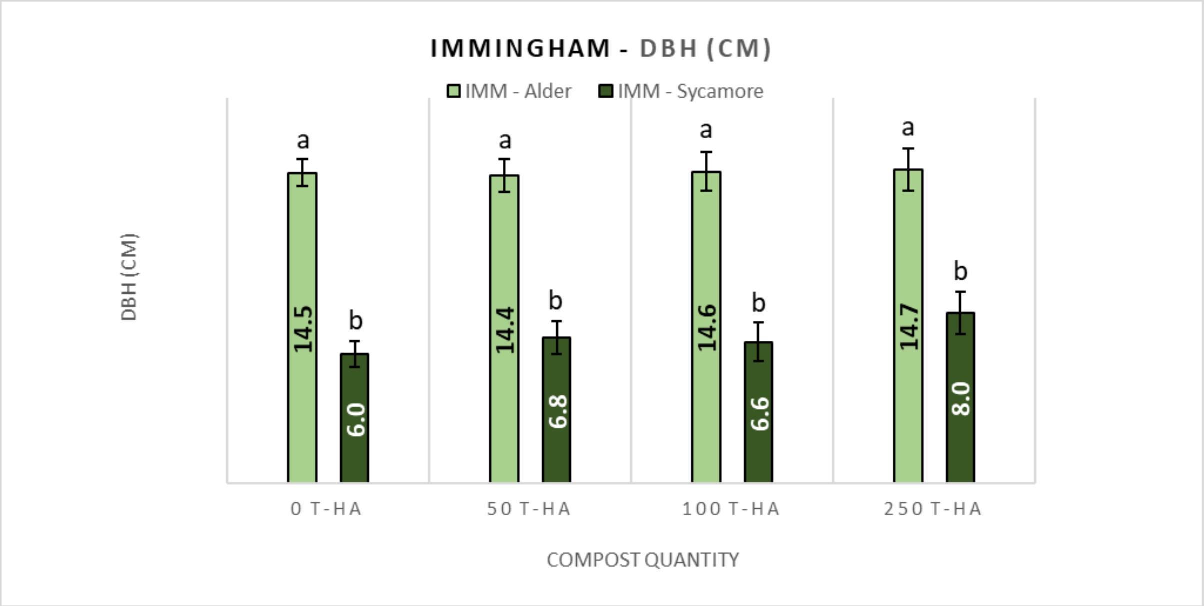


FIGURE 7.54 – *Impact of compost quantity on stem diameter. Where letters are not shared a significant difference exists (p<0.05). Compost quantity had no significant impact on stem diameter for either tree species. Despite this Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction found ALDER stem diameter was significantly greater than SYCAMORE H (7) = 286.6, p 0.001. Thus, ALDERS fast growth habit resulted in greater stem diameter over the period observed to date.*

IMMINGHAM - ANNUAL (%) GROWTH RATES (1997 - 2019)

Site	Treatment	**ALDER-DEEP**		**ALDER-SHALLOW**		**SYCAMORE-DEEP**		**SYCAMORE-SHALLOW**	
	Year	Height (m)	Growth (%)	Height (m)	Growth (%)	Height (m)	Growth (%)	Height (m)	Growth (%)
Immingham	1997	0.52		0.52		0.36		0.35	
	1998	0.35	-33	0.32	-38	0.24	-33	0.20	-42.86
	1999	0.66	89	0.61	91	0.21	-13	0.24	20.00
	2000	1.29	95	1.26	107	0.24	14	0.28	16.67
	2001	2.25	74	2.17	72	0.32	33	0.37	32.14
	2002 - 2019	11.00	9	10.30	9	6.00	18	6.00	17

TABLE 7.5 – *Annual tree height and (%) growth increments at IMMINGHAM from 1997 to 2019. In DEEP treatments Italian alder grew 0.7 m taller than those planted in SHALLOW incorporations. However, in 2001 the difference between ALDER-DEEP and ALDER-SHALLOW was only 0.08 m. Thus, the majority (0.62 m) of increased height recorded in 2019 emerged post-2001. Concerning SYCAMORE, incorporation depth had no influence on its growth.*

To evaluate whether composted green waste (CGW) improved tree growth via its influence on above and below-ground nitrogen cycling, foliar and soil C:N ratios were measured throughout Immingham after 20-years. Italian alder and Sycamore were expected to have differing effects on foliar and soil nitrogen cycling, due to Alders ability to fix atmospheric-N through root-bacteria symbiosis. However, in similar fashion to Winterton, CGW appeared to have minimal impact on foliar and soil C:N at Immingham. Indeed, both foliar and soil C:N ratios were almost identical for both tree species regardless of CGW application rate **(FIGURE 7.55** and **FIGURE 7.56)**. The homogeneity present in soil and foliar C:N values throughout Immingham, all being below 20:1, suggests variables additional to compost influenced nitrogen cycling throughout the site.

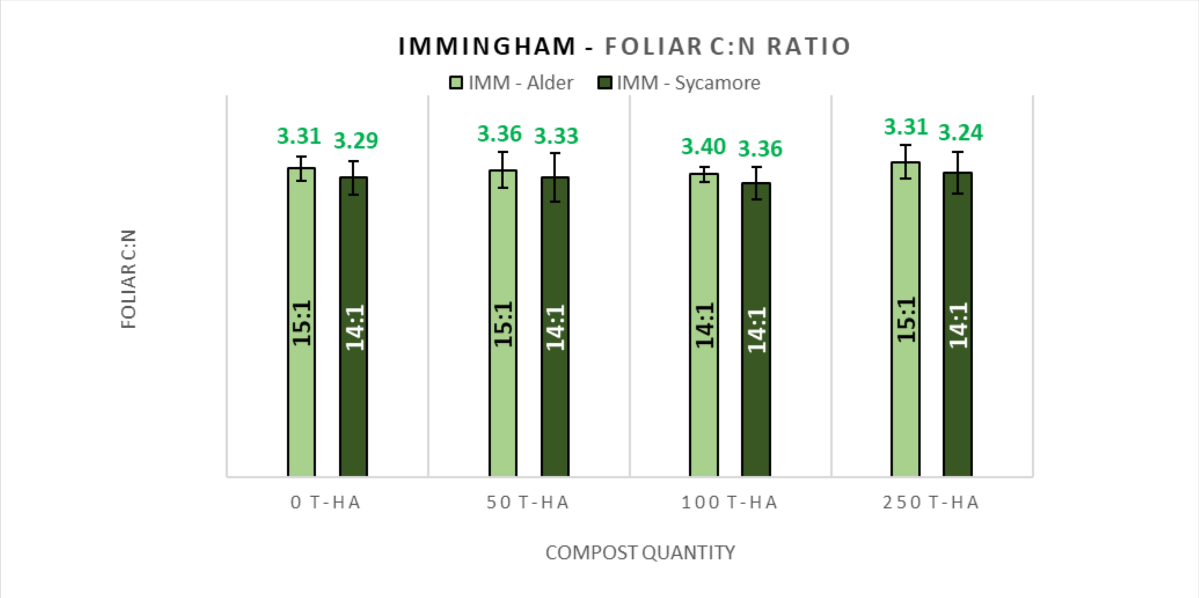


***FIGURE 7.55** – Impact of CGW quantity on foliar C:N at IMMINGHAM. No significant differences were found between treatments. Foliar C:N was similar for all tree species and quantities of compost. Values above bars represent (%) foliar-N. **GREEN =** above FC guidelines of (2.3 % for SYC; 2.8% for ALD). No trees were Foliar-N deficient at Immingham, however SYCAMORE values were similar to ALDER and much higher than Winterton.*

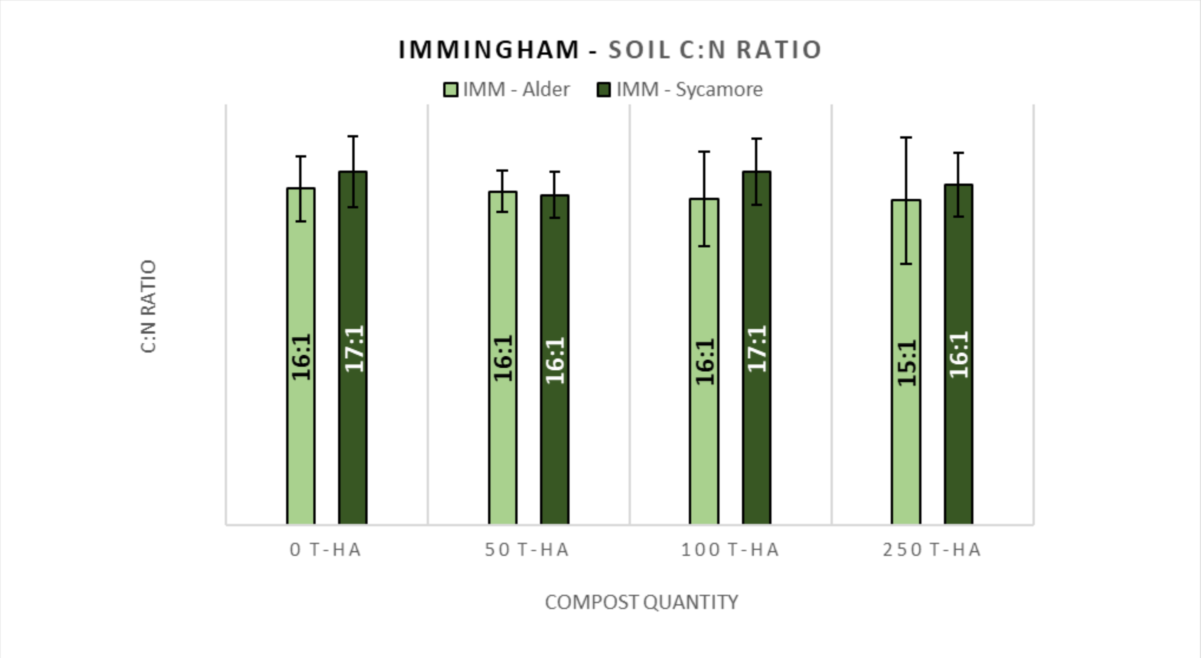


***FIGURE 7.56** – Impact of CGW quantity on soil C:N at IMMINGHAM. No significant differences were found between treatments. Although soil C:N was slightly higher under SYCAMORE, soil C:N was similar under all tree species and quantities of compost.*

7.8.2 Impact of compost addition on soil physical quality

Following 20-years of soil development at Immingham soil physical parameters were assessed, with soil bulk density (SBD) used as a proxy for soil formation. Overall, similar patterns were found at Immingham to those observed at Winterton. Namely, topsoils (0 – 15 cm^{-1}) had significantly lower bulk density and significantly higher moisture content than subsoils (15 – 50 cm^{-1}). However, a key difference was apparent between sites. Namely, Immingham subsoils were less dense (-0.10 – 0.15 g cm^{-3}) and contained more moisture (+1.5 – 3.0 % water) than Winterton subsoils. Consequently, Immingham subsoils were more porous, and better at regulating moisture availability and infiltration throughout the profile.

Indeed, independent one-way ANOVA followed by post-hoc test with bonferroni correction found that under both tree species SBD was significantly lower at 0-15 cm^{-1} sampling depth compared with cores extracted from deeper down the profile (i.e. 15-30 and 30-50 cm^{-1}): F (5, 186) = 16.85, p 0.001 **(FIGURE 7.57)**. However, despite differences in soil bulk density between sampling depths, neither compost quantity nor compost incorporation depth influenced SBD.

Moving on, at Immingham differences in soil moisture (% water) between treatments were compared using one-way ANOVA. As with Winterton, significant differences between topsoil and subsoil were found F (5, 186) = 4.782, p 0.001 **(FIGURE 7.58)**. Generally, soil moisture (% water) was significantly higher at 0-15 cm^{-1} sampling depth compared with cores taken from deeper down the profile. However, post-hoc test with bonferroni correction highlighted exceptions to this rule. At Immingham only Italian alder topsoils contained significantly more moisture than subsoils, whereas at Winterton both Alder and Sycamore topsoils contained more moisture than subsoils **(FIGURE 7.36).** In summary, like Winterton, Immingham topsoils had significantly lower bulk density and significantly higher moisture content than subsoils. However, under Sycamore the difference between topsoil and subsoil was not always statistically significant.

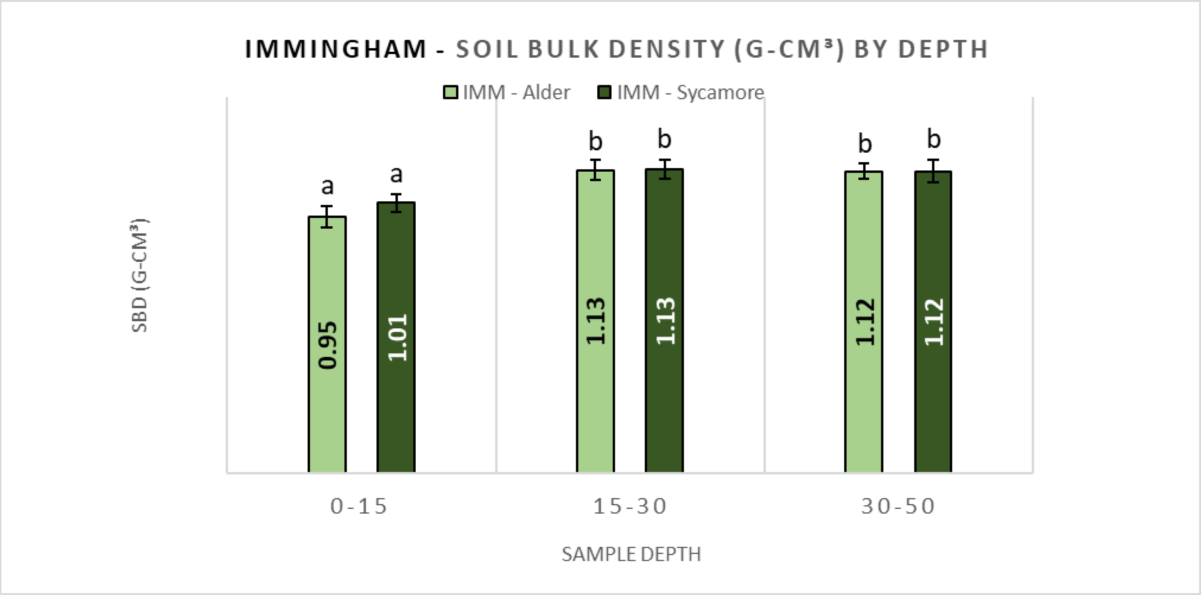


FIGURE 7.57 – *Changes in soil bulk density (SBD) at different sampling depth's. Where letters are not shared a significant difference exists ($p<0.05$). Independent one-way ANOVA with bonferroni correction found SBD was significantly lower at 0-15 cm^{-1} sampling depth for both tree species* *F (5, 186) = 16.85, p <0.001*.

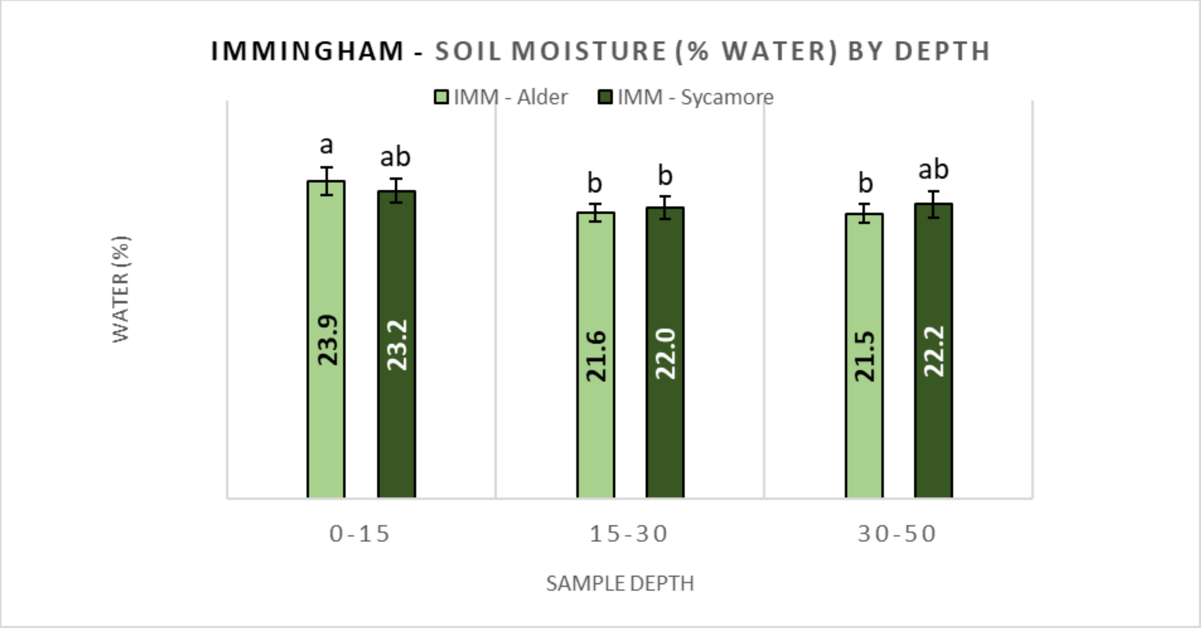


FIGURE 7.58 – *Changes in soil moisture (% water) at different sampling depth's. Where letters are not shared a significant difference exists ($p<0.05$). Independent one-way ANOVA with bonferroni correction found soil moisture was significantly higher only under ALDER 0-15 cm^{-1} sampling depth* *F (5, 186) = 4.782, p <0.001*. *This was not the case for SYCAMORE suggesting ALDER can have a marginally stronger influence on moisture retention in reclaimed topsoils.*

7.8.3 Impact of compost addition on soil organic matter, soil nitrogen, and carbon storage

To evaluate CGW impacts on soil chemistry at Immingham after 20-years, soil cores were collected from all experimental plots. Cores provided the sample material from which data on soil organic matter (SOM), soil carbon, and soil nitrogen were generated. Firstly, regarding the effect of compost quantity on SOM content, an independent one-way ANOVA found differences between treatments were marginally outside significance F (7, 120) = 1.964, p <0.07. However, because the result was close to the threshold ($p < 0.05$), a post-hoc test with tukey correction was still performed. This revealed one significant difference ($p < 0.05$). Namely, SYCAMORE (50 t-ha^{-1}) treatments contained more soil organic matter than SYCAMORE CONTROL (0 t-ha^{-1}) **(FIGURE 7.59)**. Indeed, SYCAMORE (50 t-ha^{-1}) had the highest SOM content throughout Immingham (5.8 %), slightly below the highest value recorded at Winterton (ALDER-CONTROL – SOM content – 6.0 %).

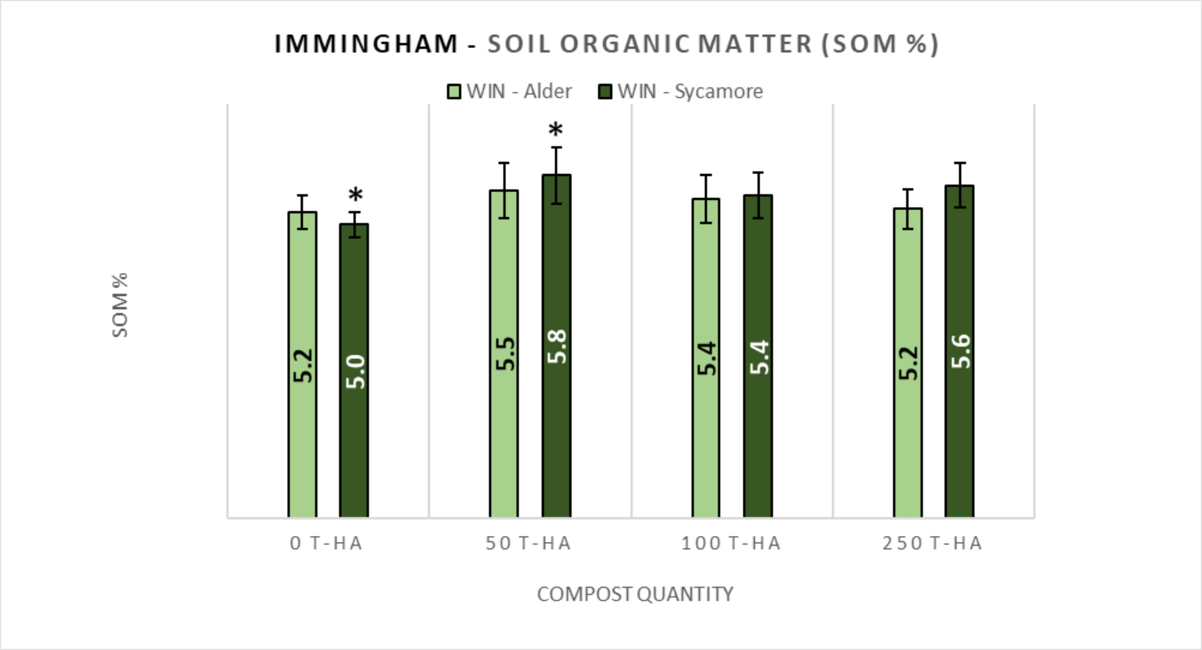


FIGURE 7.59 – *Impact of tree species and CGW quantity on soil organic matter content (SOM) at IMMINGHAM. Significant differences are marked by asterisks. Independent one-way ANOVA found differences in SOM between treatments were marginally outside significance F (7, 120) = 1.964, p <0.07. However, because the result was close, a post-hoc test with tukey correction was still performed. This identified one significant difference, namely SYCAMORE (50 t-ha^{-1}) treatments contained more SOM than SYCAMORE-CONTROL (0 t-ha^{-1}) ($p < 0.05$) as indicated by asterisks.*

Total soil carbon (t-ha^{-1}) was assessed next, revealing a pattern that appeared repeatedly in soil chemistry data. Namely, soil carbon and nitrogen tended to be higher under SYCAMORE rather than ALDER, with greater quantities found in SHALLOW rather than DEEP incorporations. This may be surprising, as Italian alder dominated the Immingham canopy, growing much larger at a faster rate than Sycamore, thus contributing additional biomass. Nevertheless, a trend of SYCAMORE and SHALLOW plots containing more carbon and nitrogen was first identified when comparing total carbon (t-ha^{-1}) in ALL ALDER versus ALL SYCAMORE plots. Indeed, Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction found SYCAMORE soils contained significantly more soil carbon (34.78 t-ha^{-1}) than ALDER soils (30.96 t-ha^{-1}): H (1) = 8.674, p 0.003.

Investigating this further, one-way ANOVA was used to examine how each tree species influenced soil carbon when different quantities of compost were applied. Again, soil carbon was consistently higher under SYCAMORE compared with ALDER, except in the 250 t-ha^{-1} treatments where soil carbon was similar for both F (7, 120) = 2.155, p <0.043 **(FIGURE 7.60)**. Post-hoc test with tukey correction revealed the greatest difference existed between SYCAMORE 100 t-ha^{-1} (37 t-ha^{-1} total carbon) and ALDER-CONTROL (28.5 t-ha^{-1} total carbon). Although the effect size was small and just outside statistical significance (p <0.07), SYCAMORE 100 t-ha^{-1} still contained substantially more soil carbon (8.5 t-ha^{-1}) than ALDER-CONTROL.

Finally, the impact of compost incorporation depth on carbon storage for both species was examined to understand whether SHALLOW (0.1 m^{-1}) or DEEP (0.6 m^{-1}) incorporations stored more soil carbon. Overall, at sampling depths between 0 and 30 cm^{-1}, SHALLOW incorporations stored more carbon than DEEP, whilst SYCAMORE soils outperformed ALDER **(FIGURE 7.61)**. Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction outlined this, finding a significant difference between treatments H (3) = 9.410, p 0.02. Specifically, SYCAMORE-SHALLOW (35.6 t-ha^{-1}) contained significantly more soil carbon (+ 5.2 t-ha^{-1}) than ALDER-DEEP (30.4 t-ha^{-1}) (p. < 0.02), outperforming all other 'tree species x incorporation depth' combinations **(FIGURE 7.61)**.

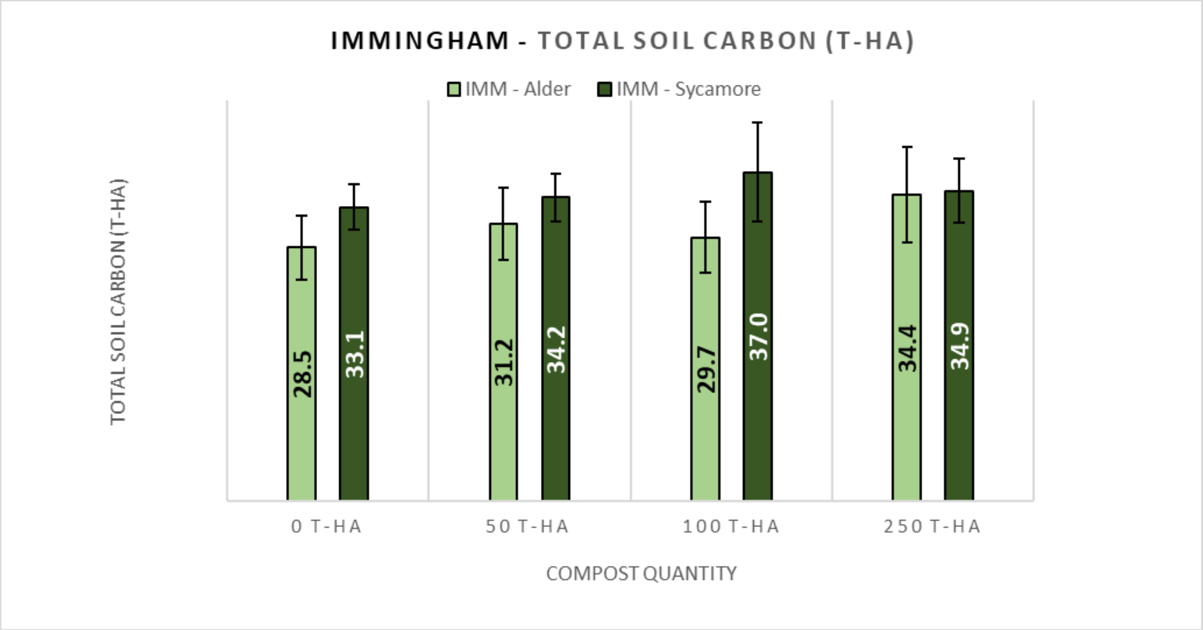


FIGURE 7.60 – *Impact of tree species and CGW quantity on soil total carbon (t-ha^{-1}) at IMMINGHAM. Independent one-way ANOVA found soil total carbon differed significantly among treatments F (7, 120) = 2.155, p <0.043. However, post-hoc test revealed differences between treatments were outside the threshold of statistical significance (p <0.07). Nevertheless, soil carbon was consistently higher under Sycamore except in the 250 t-ha^{-1} treatments where soil carbon was similar for both tree species.*

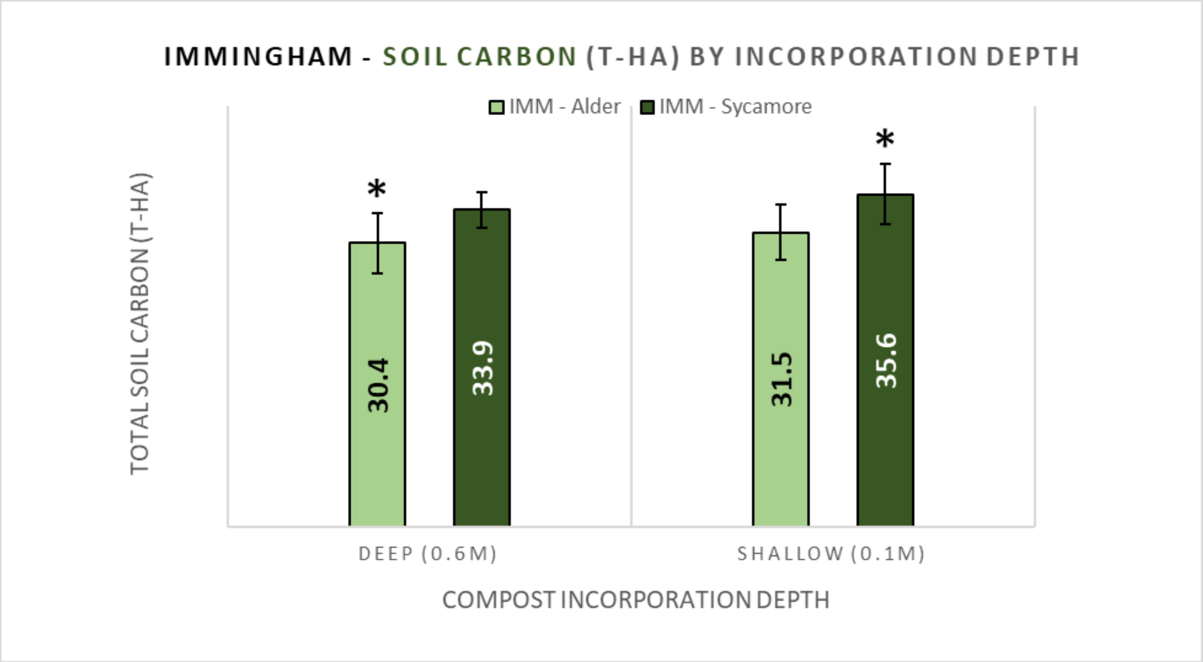


FIGURE 7.61 – *Impact of tree species and incorporation depth on soil total carbon (t-ha^{-1}) at IMMINGHAM. Significant differences are marked by asterisks. Kruskal-Wallis H-test compared DEEP versus SHALLOW incorporations. A significant difference was found H (3) = 9.410, p <0.02. Indeed, SYCAMORE-SHALLOW contained significantly more soil carbon (+ 5 t-ha^{-1}) than ALDER DEEP as indicated by asterisks. SYCAMORE-SHALLOW also contained more soil carbon than any other 'tree species x incorporation depth' combination.*

Moving on, soil total nitrogen (t-ha^{-1}) and available nitrogen (kg-ha^{-1}) were analysed by taking sub-samples from soil material used previously to generate soil organic matter and soil carbon data. Overall, similar patterns to those observed for carbon were found, with both total nitrogen (t-ha^{-1}), and available nitrogen (kg-ha^{-1}) highest under SYCAMORE and in SHALLOW incorporations.

Firstly, with regard to compost quantity the amount of compost applied had *no significant* impact on total nitrogen (t-ha^{-1}). Nevertheless, under both tree species total nitrogen rose linearly with increasing compost quantity, reaching its highest (2.5 t-ha^{-1}) in ALDER 250 t-ha^{-1}, which was 0.7 t-ha^{-1} more than ALDER-CONTROL (1.8 t-ha^{-1}) **(FIGURE 7.62)**. In contrast under SYCAMORE, total nitrogen levelled out at 50 t-ha^{-1}. Thus, interactions between ALDER and compost appear to have pushed total nitrogen to the highest rate recorded (2.5 t-ha^{-1}). Despite this, SYCAMORE stored more total nitrogen than ALDER in all other treatments (0, 50, 100 t-ha^{-1}). Furthermore, SYCAMORE accumulated its highest total nitrogen (2.3 t-ha^{-1}) in the 100 t-ha^{-1} treatments, almost achieving parity with ALDER 250 t-ha^{-1}.

Finally, the impact of compost incorporation depth on total nitrogen was examined to understand whether SHALLOW (0.1 m^{-1}) or DEEP (0.6 m^{-1}) incorporations would store more nitrogen. As with carbon, total soil nitrogen was higher in SHALLOW incorporations rather than DEEP, and greater under SYCAMORE rather than ALDER **(FIGURE 7.63)**. Indeed, a Kruskal-Wallis H-test followed by Dunn's post-hoc with bonferroni correction found the same significant difference as with carbon H (3) = 7.959, p 0.05. Namely, SYCAMORE-SHALLOW (2.3 t-ha^{-1}) contained significantly more total nitrogen than ALDER-DEEP (2.0 t-ha^{-1}) (p. < 0.02), with the former storing more total nitrogen than any other tree species x incorporation depth combination **(FIGURE 7.63)**. As such, higher C and N in SHALLOW incorporations appears related to one-off incorporation of compost performed in 1997. What's more, the effect remains detectable in 2019 following two decades of site development.

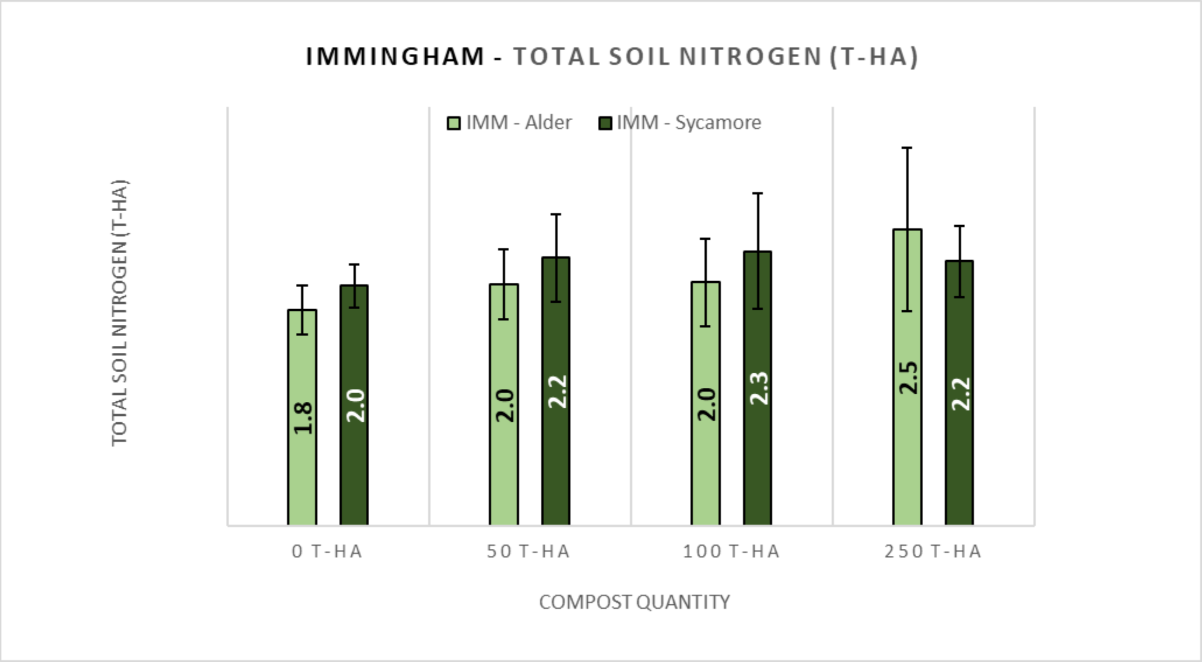


FIGURE 7.62 – Impact of tree species and CGW quantity on soil total nitrogen (t-ha^{-1}) at IMMINGHAM. No significant difference in was found between treatments F (7, 120) = 1.190, p <0.314. Nevertheless, under both tree species soil total nitrogen rises along a linear gradient with increasing quantities of CGW. The highest total nitrogen was achieved by ALDER (250 t-ha^{-1}), reaching (0.7 t-ha^{-1}) more than the respective ALDER-CONTROL. In all other treatments SYCAMORE contained more total nitrogen than ALDER, suggesting different dynamics are playing out in SYCAMORE plots.

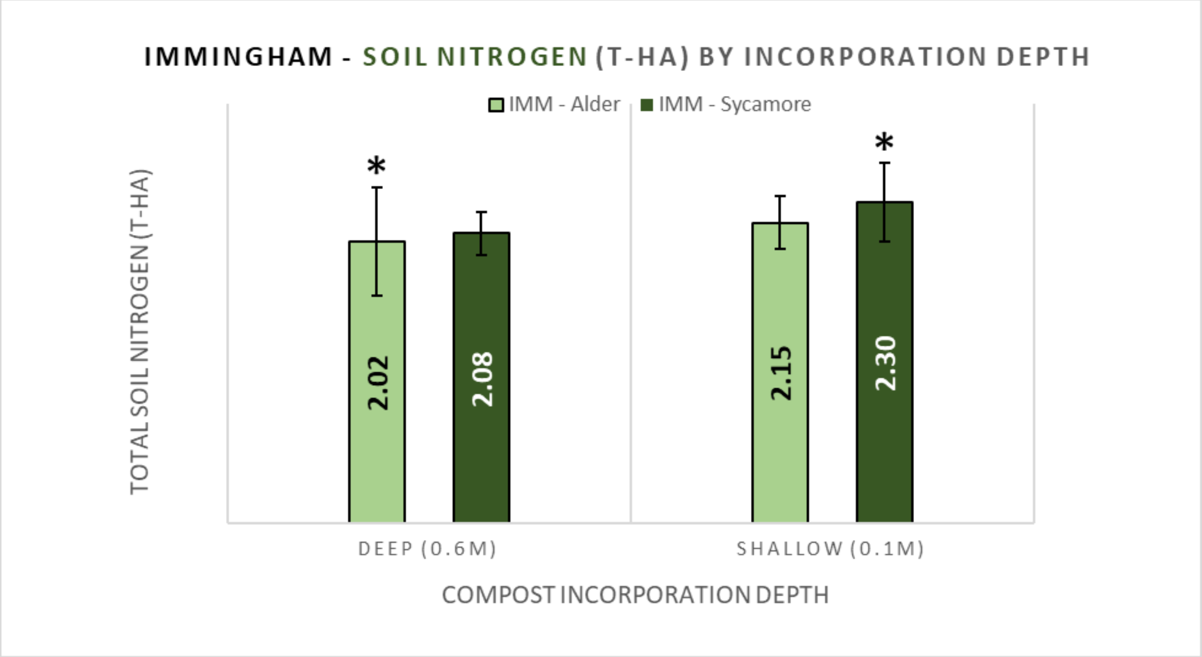


FIGURE 7.63 – Impact of tree species and incorporation depth on soil total nitrogen (t-ha^{-1}) at IMMINGHAM. Significant differences are marked by asterisks. Kruskal-Wallis H-test compared DEEP versus SHALLOW incorporations. A significant difference was found H (3) = 7.959, p 0.05. Namely, SYCAMORE-SHALLOW contained significantly more total nitrogen (+ 0.3 t-ha^{-1}) than ALDER DEEP as indicated by asterisks. Indeed, SYCAMORE-SHALLOW contained more soil nitrogen than any other 'tree species x incorporation depth' combination.

The final soil chemical variable assessed was available nitrogen (kg-ha^{-1}). Unlike total nitrogen (t-ha^{-1}) which measures all nitrogen present in soil, whether it be organic, inorganic, or physically, chemically, or biologically bound to the soil matrix; available nitrogen (kg-ha^{-1}) measures the fraction that is mobile and available for uptake by plants. Thus, by measuring available-N the efficacy of different tree species x compost combinations could be more thoroughly evaluated.

Overall, the patterns previously observed for total carbon and total nitrogen (t-ha^{-1}) were repeated for available-N (kg-ha^{-1}). Namely, available-N was higher under SYCAMORE and in SHALLOW incorporations **(FIGURE 7.64, FIGURE 7.65)**. Yet by examining the impact of compost quantity, Kruskal-Wallis H-test found nitrogen availability was significantly higher when certain quantities of compost were applied: <u>H (7) = 20.14, p 0.005</u>. Specifically, Dunn's post-hoc test with bonferroni correction revealed SYCAMORE 100 t-ha^{-1} provided significantly more available-N (kg-ha^{-1}) than SYCAMORE 250 t-ha^{-1} (p <0.01), and all ALDER treatments except the corresponding ALDER 100 t-ha^{-1} (p 0.004 – 0.05) **(FIGURE 7.64).**

Incorporation depth similarly impacted soil available-N (kg-ha^{-1}). Independent one-way ANOVA identified significant differences between tree species and incorporation depth <u>F (3, 124) = 3.140, p <0.03</u>. Overall, soil available-N was generally higher under SYCAMORE than ALDER. However, the difference between SYCAMORE-SHALLOW (1.59 kg-ha^{-1}) and ALDER DEEP (1.09 kg-ha^{-1}) was statistically significant (p. 0.03) **(FIGURE 7.65)**. This suggests dynamics at play in SYCAMORE plots alter soil chemistry to a greater extent than those associated with ALDER. Ultimately, results from soil chemical analysis at Immingham show that despite ITALIAN ALDER's vigorous growth, strong survival, and canopy dominance; levels of soil organic matter (%), total carbon (t-ha^{-1}), total nitrogen (t-ha^{-1}), and available nitrogen (kg-ha^{-1}) are comparable or greater under SYCAMORE. This may be somewhat surprising, however given the effect is soil-based with significant differences occurring between treatments, a compost-effect from one-off application in 1997 appears partly responsible.

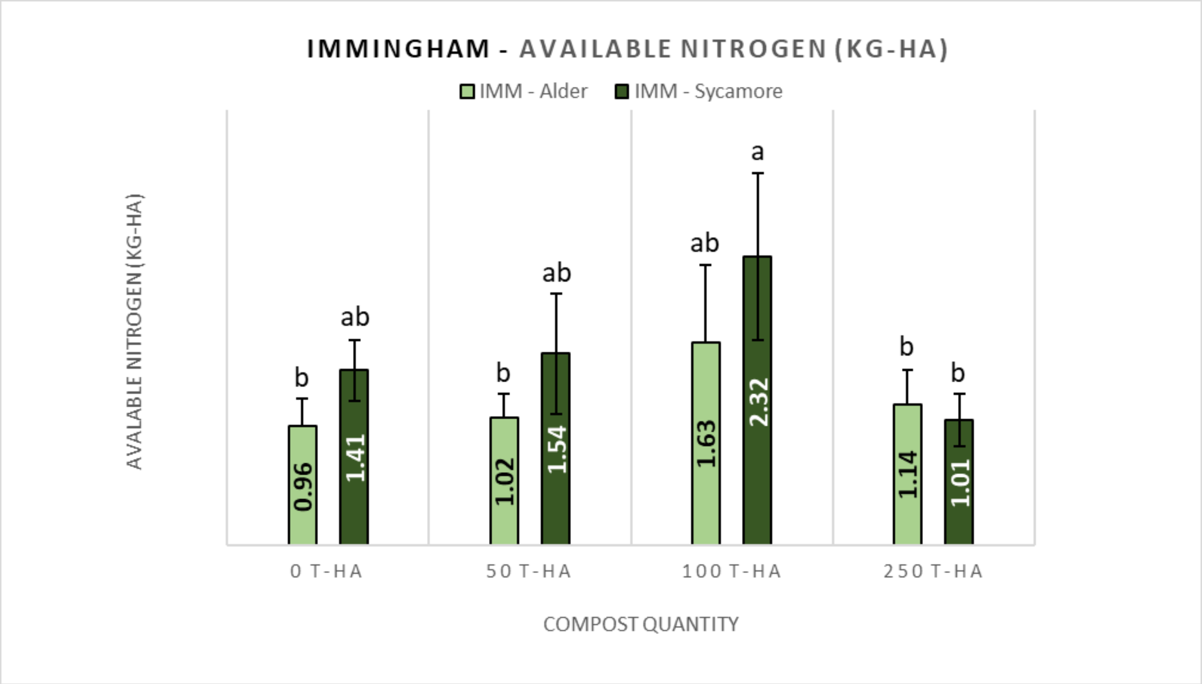


FIGURE 7.64 – *Impact of tree species and CGW quantity on soil available nitrogen (kg-ha^{-1}) at IMMINGHAM. Where letters are not shared a significant difference exists ($p<0.05$). Kruskal-Wallis H test followed by Dunn's post-hoc with bonferroni correction revealed a statistically significant differences among treatments H (7) = 20.14, p 0.005. Post-hoc test revealed available-N was significantly higher in SYCAMORE 100 t-ha^{-1} compared to SYCAMORE 250 t-ha^{-1} (p <0.01), and all ALDER treatments except the corresponding ALDER 100 t-ha^{-1} (p 0.004 – 0.05). Available nitrates (NO_3^-) account for 75 % of the values illustrated with the remainder being ammonium (NH_4^+).*

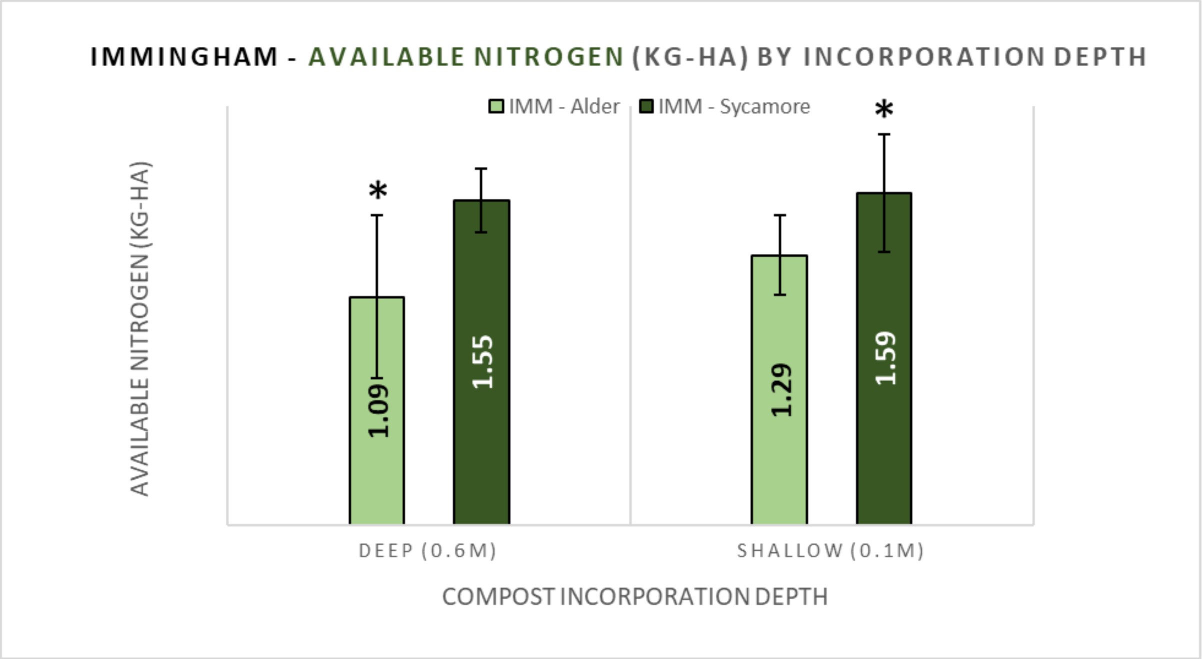


FIGURE 7.65 – *Impact of tree species and incorporation depth on soil available nitrogen (kg-ha^{-1}) at IMMINGHAM. Significant differences are marked by asterisks. Independent one-way ANOVA found significant differences between treatments F (3, 124) = 3.140, p <0.03. Post-hoc test with bonferroni correction revealed available-N was significantly higher in SYCAMORE-SHALLOW compared with ALDER DEEP (p. 0.03).*

7.8.4 Impact of compost addition on earthworm populations and soil development

Earthworms were sampled at Immingham in April-May 2019, more than 21-years after the experiment was first established. Overall, a series of dynamics similar to those found at Winterton were identified. Namely, Italian alder's presence, plus increasing quantities of composted green waste elevated endogeic and epigeic abundance. However, although community structure was similar at both sites, Immingham's population was twice as abundant (117 m^{-2}) as Winterton's (65 m^{-2}) and more diverse, being composed of ten species compared to Winterton's eight. Mean densities for Immingham's ten species were as follows: *Aporrectodea longa* (34 m^{-2}), *Allolobophora chlorotica* (27 m^{-2}), *Lumbricus castaneus* (21 m^{-2}), *Aporrectodea caliginosa* (19 m^{-2}), *Aporrectodea rosea* (6 m^{-2}), *Lumbricus rubellus* (4 m^{-2}), *Lumbricus terrestris* (4 m^{-2}), *Lumbricus festivus* (1 m^{-2}), *Dendrobaena octaedra* (0.3 m^{-2}), *Octolasion cyaneum* (0.2 m^{-2}) **(FIGURE 7.66)**.

Examining the impact of tree species on earthworm eco-group, ALDER increased abundance of endogeic (*A. caliginosa, A. chlorotica*) and epigeic (*L. castaneus*) earthworms as was the case at Winterton. However, Kruskal-Wallis H-test found the impact of tree species on eco-group abundance was only significant in one instance: H (5) = 64.95, p <0.001 **(FIGURE 7.67)**. Namely, epigeic abundance was significantly greater under ALDER (+ 14 m^{-2}) compared with SYCAMORE. Endogeics were also notably greater under ALDER (+ 21 m^{-2}), although the difference was not statistically significant. Thus, community structure under ALDER was dominated by endogeics and epigeics. Conversely under SYCAMORE, anecic earthworms were more prominent (40 m^{-2}) being equal to endogeic (40 m^{-2}) and significantly greater than epigeic (18 m^{-2}) populations (p <0.01). Indeed, at Immingham anecic species (*A. longa*) accounted for (32 %) of the total population compared to (26 %) at Winterton, further outlining anecics more prominent association with SYCAMORE at Immingham **(FIGURE 7.66, FIGURE 7.67)**.

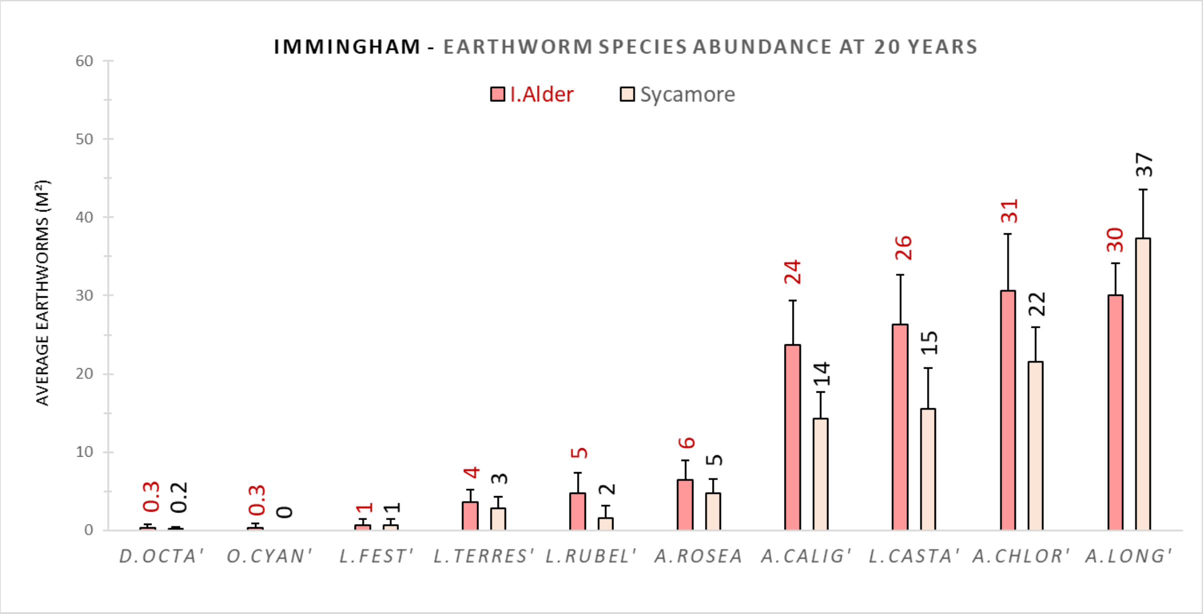


FIGURE 7.66 – *Earthworm species abundance (m^{-2}) at IMMINGHAM following 21-years of site development. For some earthworm species, abundance was similar regardless of any tree related effects. However under ALDER, the density of A. chlorotica, A. caliginosa, and L. castaneus was greater suggesting the presence of an 'Alder-effect'. Indeed, earthworm abundance was significantly greater under ALDER, with an average of 10 m^{-2} more earthworms associated with ALDER trees.*

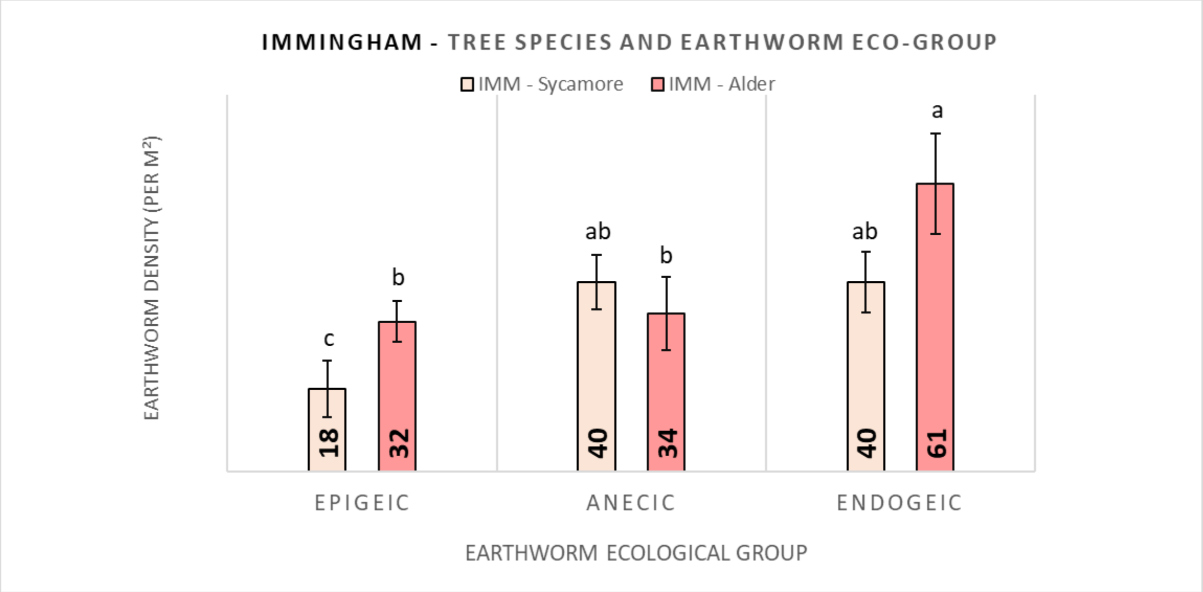


FIGURE 7.67 – *Impact of tree species on abundance of earthworm ecological group (m^{-2}). Where letters are not shared a significant difference exists ($p<0.05$). Overall, ALDER increases abundance of certain endogeic and epigeic species. Indeed, Kruskal-Wallis H test found significant differences in eco-group between tree species <u>H (5) = 64.95, p <0.001</u>. Dunn's post-hoc test with bonferroni correction revealed that ALDER increased epigeic and endogeic abundance relative to SYCAMORE. Conversely, SYCAMORE supported fewer epigeic and endogeic earthworms than ALDER, allowing anecic A. longa to adopt a more prominent role.*

The impact of compost quantity on eco-group abundance was assessed next, to ascertain whether increasing quantities of compost would produce a long-term increase in earthworm abundance **(FIGURE 7.68)**. Overall, compost raised abundance of endogeic earthworms, with effects greater at higher application rates. Indeed, Kruskal-Wallis H-test confirmed composts impact was significant, although the nature of its impact alternated depending on the quantity of compost applied: <u>H (11) = 55.93, p <0.001</u>. Adding 50 t-ha^{-1} compost into CONTROL plots significantly increased endogeic (+ 21 m^{-2}) and anecic (+ 8 m^{-2}) abundance. Epigeics also increased, but only marginally (+ 5 m^{-2}). Clearly, 50 t-ha^{-1} benefitted all three eco-groups. However, by increasing compost to 250 t-ha^{-1} anecic and epigeic abundance fell to levels observed in CONTROL's benefitting endogeic earthworms only. In summary, small quantities of compost increased abundance of all three eco-groups, whereas larger concentrations amplified endogeic populations only.

Finally, incorporation depth influenced earthworm abundance, with ALDER-DEEP increasing endogeic and epigeic populations. Yet, closer examination found this was chiefly a tree-species effect, with ALDER increasing epigeic density wherever it was present: (Kruskal-Wallis H-test <u>H (11) = 71.10, p <0.001)</u>. Indeed, ALDER's presence increased epigeic abundance regardless of incorporation depth to levels comparable with other eco-groups **(FIGURE 7.69)**. Nevertheless, another general non-statistical trend remained apparent and may have broader implications. Namely, ALDER increased endogeic abundance again, whilst ALDER-DEEP maintained high abundance for all three eco-groups **(FIGURE 7.69)**. Yet even though ALDER-DEEP had the highest abundance for all eco-groups, it had lower carbon and nitrogen stocks than any 'tree species x incorporation depth' treatment. Conversely, SYCAMORE-SHALLOW (0.1 m^{-1}) had the highest carbon and nitrogen stocks but supported only one or two eco-groups. Thus, the greatest differences in carbon and nitrogen stocks were among ALDER-DEEP and SYCAMORE-SHALLOW, which incidentally had the greatest contrasts in earthworm eco-group abundance. A summary of composts impact on tree growth, soil development, and earthworms is now presented using visual outputs produced via principal component analysis (PCA).

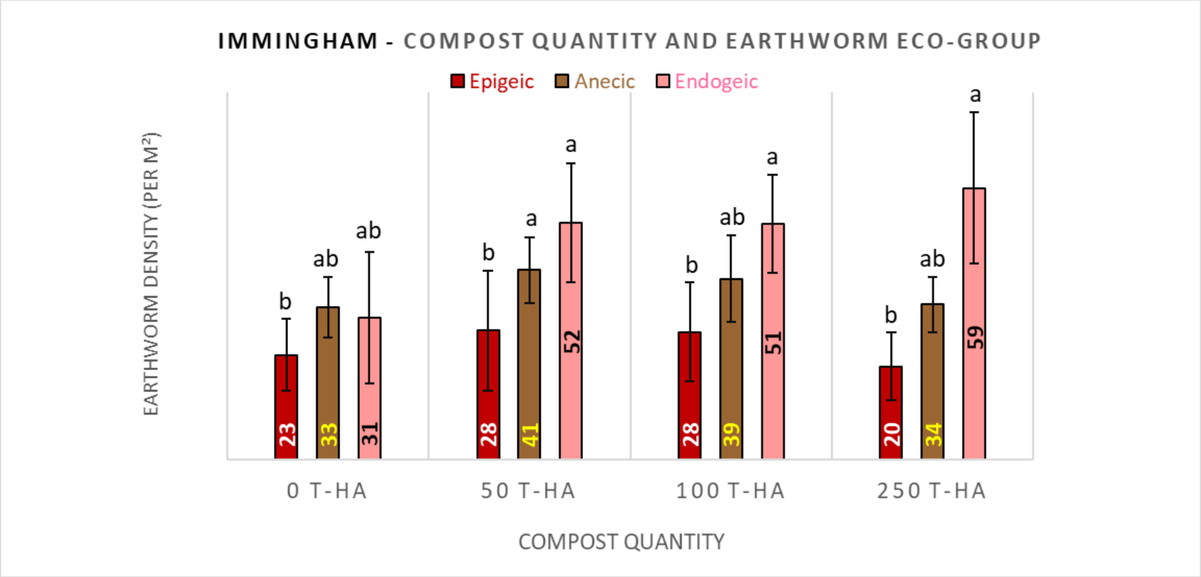


FIGURE 7.68 – Impact of compost quantity on abundance of earthworm ecological group (m^{-2}). Where letters are not shared a significant difference exists ($p<0.05$). Kruskal-Wallis H-test found compost quantity had a significant impact on eco-group density: H (11) = 55.93, p <0.001. In effect, a small quantity of compost (50 t-ha^{-1}) increased abundance of all eco-groups relative to CONTROL. However, as compost quantity gradually increases, larger concentrations (250 t-ha^{-1}) continue to amplify endogeic populations at the expense of other eco-groups.

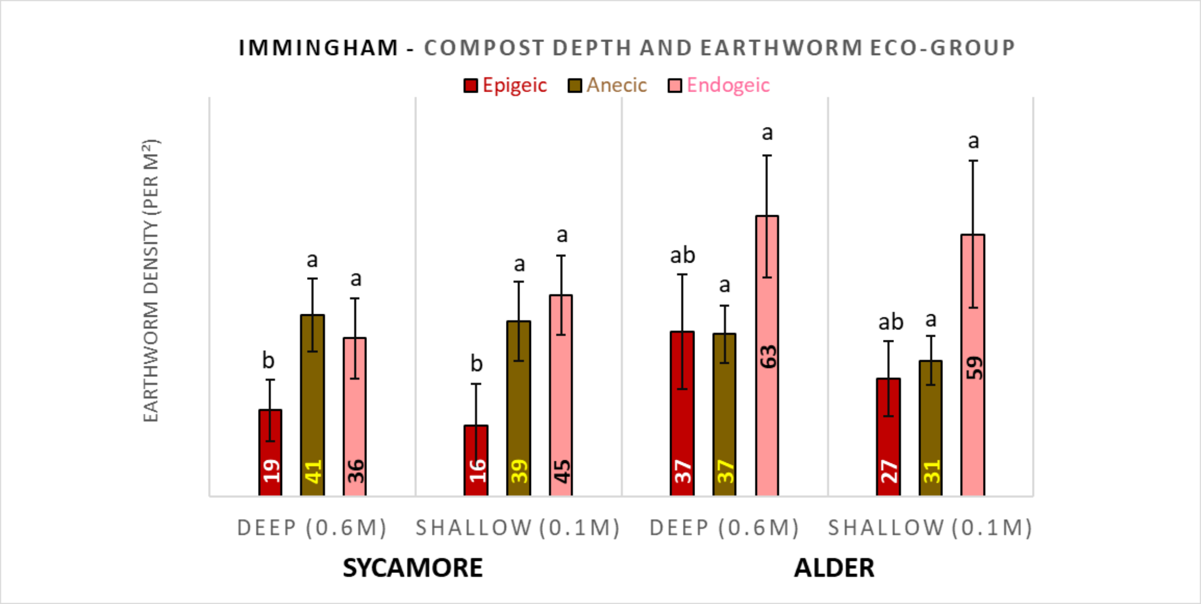


FIGURE 7.69 – Impact of incorporation depth on abundance of earthworm ecological group (m^{-2}). Where letters are not shared a significant difference exists ($p<0.05$). Kruskal-Wallis H test found incorporation depth had a significant impact on earthworm density H (11) = 71.10, p <0.001. However, Dunn's post-hoc test with bonferroni correction found this was a tree species effect, with ALDER increasing epigeic abundance regardless of incorporation depth, to levels comparable with other eco-groups. Nevertheless, compared with other treatments ALDER-DEEP noticeably increases both endogeic and epigeic populations, maintaining high abundance for all three ecological groups. This suggests interactions from ALDER above and compost below when distributed DEEP throughout the soil profile, supports greater earthworm abundance than any other treatment.

7.8.5 Overview of results

Principal component analysis (PCA) was used to evaluate the various categories of site data obtained between November 2018 and July 2019. The analysis produced graphical outputs known as 'biplots' which provided a visual summary of site dynamics. Using these, the interplay between compost, tree growth, earthworms, and soil conditions was evaluated by qualitatively analysing data distribution throughout the biplots. For example, a clustering of data points indicates greater association among variables. Vector length indicates effect size and strength. Vector direction and angles between vectors indicates the nature of a relationship, i.e. vectors following similar trajectories are more closely associated, vectors at opposite ends of 180° angles have 'opposing' or 'polar' correlations, whereas vectors at right angles have no correlation or relationship. Overall, the PCA biplots in **FIGURE 7.70** and **FIGURE 7.71** show data points separate according to tree species, with separation clearer at 0-15 cm^{-1} than at 15-30 cm^{-1}. Indeed, the effects observed are not exclusive to ALDER, as several key variables (vectors) associate with SYCAMORE data points. Thus, 'tree-species-effects' seem to be apparent, or rather, contrasting dynamics between ALDER and SYCAMORE plots, especially at 0-15 cm^{-1}.

Regarding the 0-15 cm^{-1} sampling depth, 'Height', 'DBH', 'Foliar C', 'Porosity', 'Water', 'Epigeic', 'Endogeic', 'Anecic', associate with ALDER. Conversely, 'SBD', 'Soil C', 'Soil N', and 'NO3' associate more strongly with SYCAMORE. This means ALDER growth was greater than SYCAMORE, ALDER foliage contained more carbon, whilst soils in ALDER plots were more porous, more moist, with greater abundance of epigeic, endogeic, and anecic earthworms. In contrast, SYCAMORE soils were denser and contained fewer earthworms. Yet despite this, soil carbon, soil nitrogen, and plant available nitrates were higher under SYCAMORE at 0-15 cm^{-1} **(FIGURE 7.70)**. This may be unexpected given ALDER's superior growth and greater earthworm activity. Indeed, accumulation of soil resources under any tree canopy is associated with production of tree biomass, hence superior ALDER growth might elevate soil carbon and nitrogen stocks. Given this occurred under SYCAMORE instead, presumably factors other than ALDER and including SYCAMORE generated additional biomass, resulting in higher carbon and nitrogen stocks under SYCAMORE at 0-15 cm^{-1} depth. Alternatively, dynamics present under ALDER may have been absent or weaker under SYCAMORE, meaning resource expenditure in SYCAMORE plots was lower leaving more C and N. Notably, even though ‘Soil C’, ‘Soil N’, and ‘NO3’ do associate with SYCAMORE, these variables are at near right angles to tree growth vectors for both tree species. This suggests dynamics un-related to tree growth may have increased soil carbon and nitrogen **(FIGURE 7.70)**.

Regarding Immingham’s 15-30 cm^{-1} sampling depth, separation still occurs according to tree species, however effects are less clear. Indeed, there is greater cross over of data points between tree-species, whilst grouping of soil carbon and nitrogen under SYCAMORE no longer occurs. Consequently, rather than vectors / variables separating according to tree species per se, they simply associate strongly or weakly with dynamics occurring in ALDER plots **(FIGURE 7.71)**. For example, ‘Anecic’ earthworms previously associated strongly with ALDER at 0-15 cm^{-1} depth, but at 15-30 cm^{-1} now have a weaker correlation and are surrounded by several SYCAMORE data points suggesting a less definitive ‘Alder-effect’. Conversely, ‘Soil C’, ‘Soil N’, and ‘NO3’ no longer group together under SYCAMORE, and are closer in proximity to several ALDER data points, outlining a weakening of a possible ‘Sycamore-effect’ **(FIGURE 7.71)**. Hence to summarise, data does separate according to tree species. Separation is also more apparent at 0-15 cm^{-1} where soil carbon and nitrogen associate with SYCAMORE plots. However, the increase in carbon and nitrogen under SYCAMORE may not arise exclusively via tree species effects. Indeed, they may be partially ascribed to non-tree related dynamics occurring in or less prevalent in SYCAMORE plots but driven by a broader range of factors than tree growth alone.

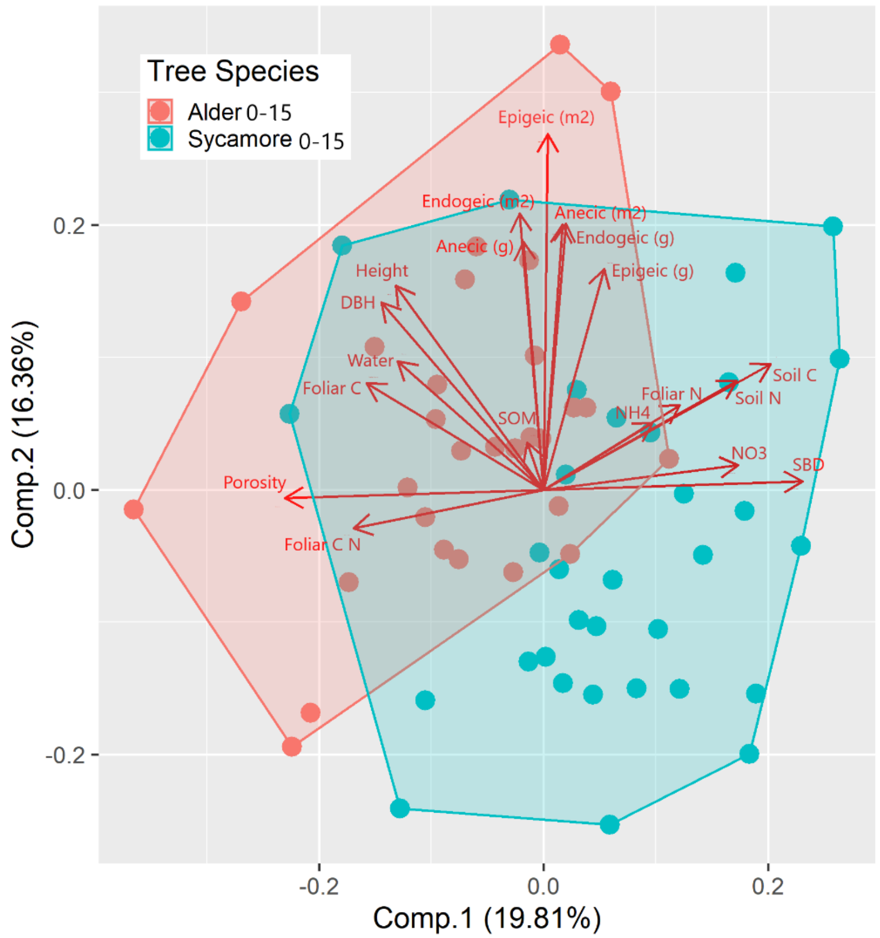


FIGURE 7.70 – *PCA examining soil physical, soil chemical, and tree growth related variables at Immingham (0-15cm sampling depth). Overall, the data shows evidence of separation according to tree species, with* PCA 1 versus PCA 2 *accounting for around 26% of the total* variation observed at *0-15cm sampling depth. This indicates that a weak 'tree species effect' or 'Alder effect' is present. Increasing 'Porosity', Tree 'Height', 'DBH', and earthworm numbers, especially the number of 'Epigeic (m2)' tend to associate with Alder. Conversely, Sycamore is more closely associated with increasing 'SBD', 'Soil C', and 'NO3'.*

FIGURE 7.71 – *PCA examining soil physical, soil chemical, and tree growth related variables at Immingham (15-30 cm sampling depth). Overall, the data shows evidence of separation according to tree species. The separation is along* PCA 1 versus PCA 2, and accounts for around 37% of the total variation observed at *15-30 cm depth. A slightly stronger 'Alder effect' is present at 15-30cm with similar variables driving the separation. Increasing Tree 'Height', 'DBH', 'Endogeic (m2)' and 'Epigeic (m2)' associate strongly with Alder. Conversely Sycamore is not strongly associated with any variables, although increasing 'Soil C', 'SBD', and available 'NO3' tend towards Sycamore as with the previous sampling depth.*

7.9 Discussion

The Humberside experiments are among the first to investigate composted green waste's (CGW) early impact on tree establishment and growth. Returning more than 20-years later has revealed a range of legacy effects with CGW increasing tree growth, nutrient availability, carbon storage, and earthworm activity throughout this period. Overall, three core dynamics are responsible for these legacy effects: 1). The Alder-effect, 2). The compost-effect, 3). The multi-layer system. To elaborate, the first two are descriptive terms. The 'Alder-effect' describes how Alder's fast-growth, N-fixing capabilities, and labile leaf-litter has wide-ranging impacts that alter plant and earthworm community composition, carbon storage, and nitrogen cycling. Similarly the 'compost-effect' describes how compost adds energetic substrates and nutrients to soils, boosting tree growth, earthworm activity, soil formation, and nutrient cycling. Both the 'Alder-effect' and 'compost-effect' are part of the experiments original factorial design. They are primary factors driving experimental results and in fact amplify one another through synergistic interactions.

In contrast, the 'multi-layer system' *characterises* how the experiments function. Rather than conforming to boundaries created at inception, the experiments on Humberside increasingly transgress plot boundaries, behaving as *interactive* systems instead. Consider, the trees and compost which make-up the experimental treatments have been joined by colonising earthworms and plants. Over time colonising organisms have moved progressively throughout the site whilst Alder and Sycamore have grown larger thus enhancing edge-effects. Hence, from within a framework created by the original boundaries, a system that interacts across boundaries, both above-ground and below has begun to develop. Consequently, the present study conceptualises this interactive system as 'multi-layer', i.e. being composed of distinct strata of soils and plants, stacked over one another in successive layers. There are five layers in total with: 1). 'DEEP incorporation' of compost being the lowest layer then 2). 'SHALLOW incorporation' of compost above, each dominated by different earthworm eco-groups. 3). A 'ground layer' of either mixed leaf-litter or herbaceous plants then covers the compost infused soil. Finally, a dual layered tree canopy of 4). 'SYCAMORE understory' and 5). 'ALDER overstory' provides the systems overarching structure.

Changes to any layer can modify ecosystem function and ecosystem service provision, however changes to the 'ALDER overstory' or 'compost layers' have the greatest impact. Indeed, data from both sites demonstrates that reducing the number of ALDER or adjusting compost depth alters earthworm community structure. In turn, this shift in earthworm community structure alters plant growth, soil development, nutrient cycling, and carbon storage, with differences between ALDER-DEEP and SYCAMORE-SHALLOW demonstrating this. For example, ALDER-DEEP supports the greatest abundance of all three earthworm eco-groups, and produces the highest recorded tree growth, but has the lowest stocks of soil carbon and nitrogen. In contrast, SYCAMORE-SHALLOW has the highest stocks of soil carbon and nitrogen but has the lowest earthworm abundance and lowest tree growth. In effect, these two treatments reside at opposite ends of the Humberside 'functional spectrum'. Indeed, ALDER-DEEP is more effective at promoting rapid soil formation and nutrient cycling mediated by earthworms. Conversely, SYCAMORE-SHALLOW is more effective at carbon storage and soil resource accumulation.

Yet rather than functioning as isolated treatments, effects observed in ALDER-DEEP and SYCAMORE-SHALLOW arise via interactions between these treatments, wherein plots behave as small components within a larger woodland mosaic. This 'mosaic effect' arises from the experiments overall proportions and layout, as plots are small and intimately mixed together with each plot containing just 10 trees producing significant edge-effects. Consequently, a monoculture of SYCAMORE in SHALLOW compost would be unlikely to elevate soil carbon beyond levels presently recorded as the 'Alder-effect' would be lacking from the system. Indeed, results from Immingham demonstrate ALDER interactions with compost influences SYCAMORE plots, by increasing SYCAMORE foliar-N, lowering soil C:N under SYCAMORE, altering herb-layer composition, and increasing earthworm abundance throughout. These dynamics are now discussed in greater detail.

7.9.1 Impact of compost addition on tree survival, growth, and foliar nutrition

Overall, CGW improved performance of Alder and Sycamore at both sites, yet composts wider impact differed according to tree species and site conditions. Large concentrations of CGW increased tree growth, soil carbon storage, and Sycamore survival, but accelerated Alder death. Indeed, CGW impacts on tree performance (i.e. growth / survival) depends on interactions among multiple factors. Specifically site conditions, tree species, compost quantity, and compost depth.

Concerning the effects of individual tree species, ALDER grew significantly taller than SYCAMORE at both sites. This is expected given ALDER's superior rate of growth (Foot et al., 2003; Foot and Moffat, 2008). From this position of canopy dominance, ALDER influenced interactions between compost and earthworms at both sites, allowing nitrogen enriched compounds to cycle through multiple system layers. Prior to 2001, compost alone increased ALDER and SYCAMORE height compared with controls (Foot et al., 2003). However, as time progressed Alder increasingly interacted with compost, ensuring trees on Humberside outperformed similar species planted on un-amended sites.

Concerning growth observed within Humberside only, the greatest improvements in tree growth occurred when larger quantities of compost (250 t-ha^{-1}) were incorporated to 0.6 m^{-1} depth. Indeed, DEEP incorporation improved ALDER and SYCAMORE growth by 0.7 – 1.1 m^{-1} compared to SHALLOW incorporation. A significant proportion of this additional growth occurred after Foot et al. (2003) concluded their observations, meaning improvements in tree growth continue to accrue beyond the first 5-years. It is likely that DEEP incorporation improves soil structure at depth, benefitting roots as they expand over time. Earthworm interactions with compost and ALDER provide an important mechanism for this, given their burrowing and bioturbation activities are known to build and maintain soil structure. Increased abundance of all three eco-groups plus improved tree growth in ALDER-DEEP demonstrates this link between earthworm activity, soil formation, and enhanced tree performance across Humberside. Thus, the recommended practice of incorporating compost at depth demonstrably improves soil structure and tree growth and is mediated by earthworms.

Comparing the Humberside experiments to other studies that did not utilise organic amendments provides further evidence of composts beneficial impact on tree height. For example, a 10-year study of woodland establishment on UK landfill by Foot and Moffat (2008) examined ITALIAN ALDER and SYCAMORE growth at three different landfills on soils of similar depth and construction to Humberside. Tree growth was lower at these unamended landfills, with ITALIAN ALDER height ranging 5.5 – 8.3 m^{-1} and SYCAMORE ranging 1.0 – 4.0 m^{-1}. Both species reached maximum height within 5 – 10 years, outlining that growth limitations were quickly realised. In contrast on Humberside ITALIAN ALDER height ranged 9.0 – 11.0 m^{-1} whilst SYCAMORE height ranged 5.9 – 7.3 m^{-1}. Thus, compared with unamended landfills, ALDER on Humberside grew + 2.7 – 3.5 m^{-1} taller whilst SYCAMORE grew + 3.0 – 4.9 m^{-1} taller (Foot and Moffat, 2008). This suggests CGW facilitated superior growth on Humberside over a longer period.

Yet, despite improved tree growth in composts presence, both tree species reached half the height recorded for natural systems. For example, in natural systems Caudullo and Mauri (2016) found Italian Alder achieved a maximum height of 25 – 30 m throughout its lifetime. Bobinac et al. (2020) found Italian alder reached heights of 13.4 m and 19.5 m in 11 and 16 years respectively, this being greater than Italian alder achieved on Humberside over a longer timeframe. Similarly, Hein et al. (2009) found Sycamore reached heights of 7.0 m over 10 years and 15.0 m over 20 years, again, both being greater than heights achieved on Humberside. Such clear height differences among natural and reclaimed systems suggest a hierarchy exists among different site types. This follows a trend from lowest to highest growth of unamended sites -> amended sites -> natural systems. Considering this there is a need to further improve soil conditions on reclaimed land beyond those provided by CGW. Nevertheless, ALDER and SYCAMORE grew substantially taller when CGW was present, compared with site controls and unamended landfills.

Turning attention to tree survival allows us to explore how interactions between site conditions, compost, and particularly ITALIAN ALDER, caused two identical experiments to diverge. This divergence arose because relatively high ALDER survival at one site (Immingham 76 %) but relatively lower ALDER survival at another (Winterton 38 %), resulted in stronger and weaker 'Alder-effects'. Indeed, differences in 'Alder-effect' intensity between sites had downstream impacts, causing the sites to diverge from one another as evidenced by experimental results.

Firstly, two main factors reduced Alder survival at Winterton, site conditions and the *concentration* of compost. The *concentration* of compost refers to large quantities of compost being present in small volumes of soil, with SHALLOW-250 t-ha^{-1} providing the clearest example. This treatment, (but also others like it), place large quantities of compost in a narrow band of soil. As a result, ALDER survival declines by 10 – 20 % regardless of site conditions. Compost was not the primary driver of ALDER decline, it merely exacerbated pre-existing issues presented by poor soils. Indeed, Winterton topsoils were wetter (+ 1.8 – 2.5 %), and subsoils denser (+ 0.10 – 0.15 g cm^{-3}) and drier (- 1.5 – 3.0 %) than those at Immingham, suggesting water was unevenly distributed throughout Winterton's soil profile. Twenty years ago, Foot et al. (2003) found landfill soils on Humberside were prone to water-logging in winter and drought in summer. More recently, when soil sampling was conducted in 2019 a localised area of Winterton was found to be waterlogged and anaerobic leading to substantial tree cover loss.

In addition to drought and water-logging, ALDER also suffered from fungal infection at Winterton, displaying symptoms consistent with *Phytophthora alni* (Webber et al., 2004; McKay, 2011) **(FIGURE 7.72)**. Interestingly, Cech and Hendry (2003) suggest *Phytophthora alni* is unlikely to cause significant damage to Alder unless trees are weakened by additional factors like compaction and fluctuating water tables. Further, Webber et al. (2004) found flooding and high nitrate environments increase occurrence of this disease. Evidently, poor soil conditions stressed Italian alder making it vulnerable to infection, whilst the act of concentrating compost around alder roots

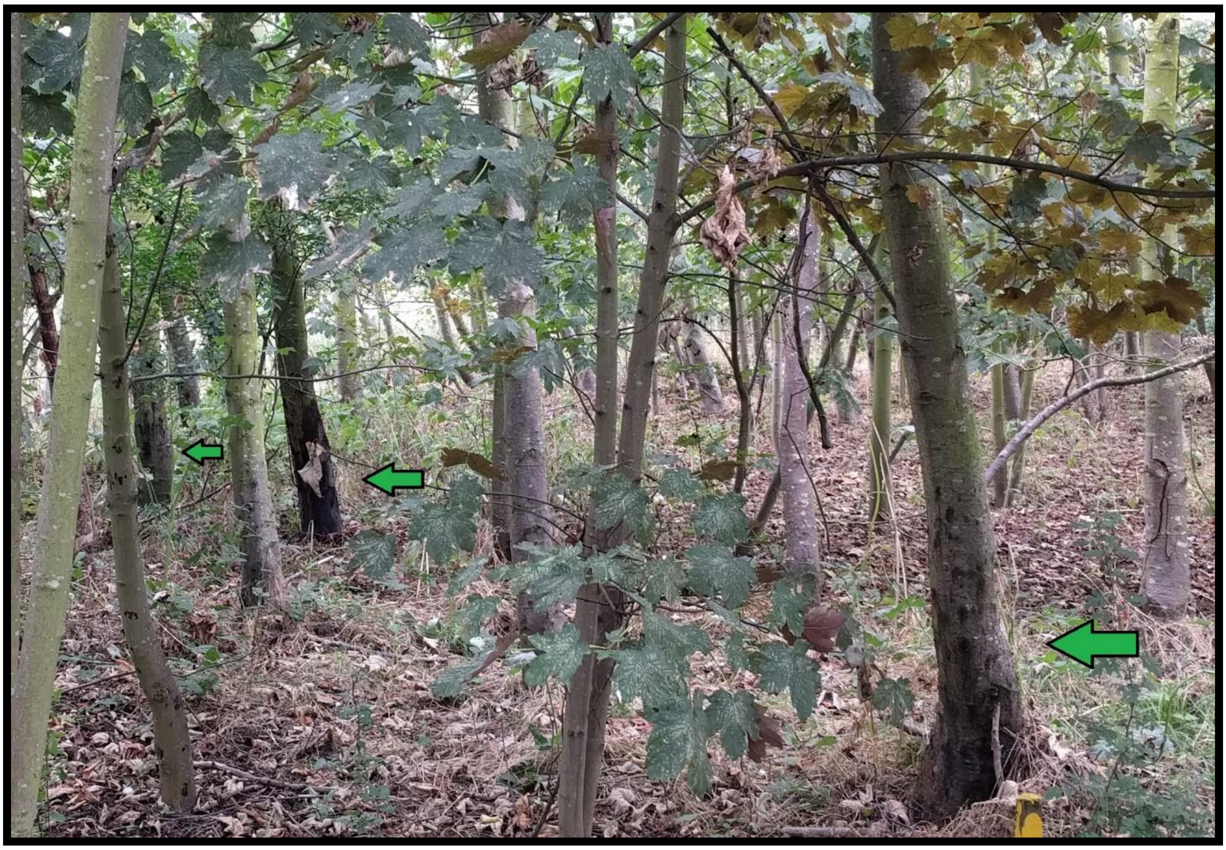


FIGURE 7.72 – Italian Alder trees infected by pathogen. Symptoms are consistent with Phytophthora alni, an increasingly common disease associated with multiple species of Alder. The 'tarry spots' visible on the lower stem characterise this disease. Black exudates emanating from the spots Indicate underlying bark is necrotic or dead. Over the coming years complete dieback can ensue, especially on sites where water levels and flows are inherently volatile (Webber et al., 2004).

In SHALLOW treatments created nutrient rich habitat for the pathogen. This pathology outlines the risks involved with combining N-fixing tree species with large *concentrations* of compost. Namely, highly enriched soils stimulate pathogen growth, which combined with poor soil structure and un-favourable moisture regimes reduce ALDER survival thus diminishing the 'Alder-effect'.

Interestingly, Italian alder survival is frequently low in landfill soils. Studies by Rawlinson et al. (2004), Foot and Moffat (2008), and now the present study identify a re-occurring pattern. Namely, Italian alder consistently grows faster and taller than most species but in doing so experiences some of the greatest losses. Considering this, Rawlinson et al. (2004) classified Italian alder as a 'high risk species', meaning large-scale losses are likely. However, results from Immingham suggest ITALIAN ALDER also offers 'high reward'. Indeed, whereas ALDER survival is low at Winterton, ALDER continues to dominate Immingham, resulting in a doubling of earthworm abundance, a quadrupling of soil mineral nitrogen, and a 25% increase in carbon storage. Accordingly, the present study attributes key differences between sites to the contrasting fortunes of ALDER. Thus, relatively high ALDER survival at Immingham (76 %), but low ALDER survival at Winterton (38 %) was the single greatest difference between sites causing them to diverge.

Paradoxically, further evidence for ALDER's central role in site divergence can be found by comparing SYCAMORE growth and foliar nitrogen between sites. At Winterton, SYCAMORE height significantly improved, growing 0.7 – 1.1 m taller than at Immingham especially in DEEP incorporations. Indeed, in the 250 t-ha^{-1} treatments at Winterton SYCAMORE achieved a mean height of 7.3 m^{-1}, this being its greatest growth on Humberside and the only occasion where SYCAMORE height was comparable to ALDER throughout Humberside. In contrast, at Immingham SYCAMORE growth was much lower and was homogenous throughout the site (6.0 m), with no improvement in any treatment. SYCAMORE growth was lower and homogenous at Immingham due to ALDER's higher survival (76 %), with ALDER's canopy dominance maintaining shading pressure over SYCAMORE thus restricting its growth. Conversely at Winterton, low ALDER survival (38 %) left an open canopy allowing SYCAMORE to exploit openings and increase its overall height aided by compost. These dynamics provide insight into the extent of ALDER's influence, with ALDER's contrasting fortunes at either site altering SYCAMORE growth. In turn, the contrasting fortunes of each tree species had downstream impacts on soil formation, nutrient cycling, and carbon storage.

For example, SYCAMORE foliar-N demonstrates ALDER's profound influence on site dynamics. Consider, where ALDER survival was high (76 % at Immingham) SYCAMORE foliar-N was similarly high (3.3 %), and near identical to ALDER foliar-N (3.4 %). According to values published by Foot and Moffat (2008) a foliar-N of (3.3 %) is a percentage point above SYCAMORE's recommended minimum (>2.3 %), and half a percentage point above the recommended minimum for ITALIAN ALDER (>2.8 %). In contrast, at Winterton where ALDER survival was low (38 %) SYCAMORE foliar-N was similarly lower (2.6 %) reflecting the reduced 'Alder-effect'. Evidently the 'Alder-effect' was prevalent at Immingham and elevated SYCAMORE foliar-N. With foliage of both ALDER and SYCAMORE enriched with nitrogen, a double layered N-rich tree canopy developed at Immingham, with downstream consequences for soil development at this site.

The dual layer N-enriched SYCAMORE and ALDER tree canopy that emerged at Immingham had a range of impacts but especially influenced soils. Indeed, SYCAMORE-SHALLOW treatments at Immingham recorded the highest C and N stocks throughout Humberside. This developed even though ALDER suppressed SYCAMORE growth, outlining the prevalence of systemic effects. For example, if SYCAMORE was solely responsible for high C and N stocks in SYCAMORE-SHALLOW at Immingham, we might expect superior SYCAMORE growth to coincide with this. Yet evidenced demonstrates SYCAMORE growth was lower at Immingham (– 0.7 to 1.1 m^{-1}) in comparison to Winterton. Consequently, the present study argues SYCAMORE was not solely responsible for increased C and N stocks in SYCAMORE-SHALLOW. Instead, greater ALDER survival at Immingham produced a stronger 'Alder-effect', which *interacted* with SYCAMORE to increase C and N stocks overall. Indeed, equal proportions of ALDER (76 %) and SYCAMORE (77 %) survived at Immingham, meaning SYCAMORE's contribution was important to increased C and N stocks. This dynamic only arose at Immingham where Alder and Sycamore survival was equal. Hence, the *balance* of interactions at Immingham appears prominent, with both tree species contributing to peak C and N stocks in SYCAMORE-SHALLOW.

In summary, the 'Alder-effect' was stronger at Immingham increasing ALDER growth and SYCAMORE foliar-N. The presence of N-enriched ALDER *and* SYCAMORE foliage then facilitated interactions between tree species and compost. Indeed, equal survival of ALDER and SYCAMORE created a favourable balance of N-enriched foliage, each with contrasting properties. Adding these two types of foliage to high *concentrations* of compost then helped to produce peak C and N stocks in SYCAMORE-SHALLOW plots. Thus, the 'compost-effect' was an important factor, with evidence of composts impacts found in soil chemistry data and now, further discussed.

7.9.2 Impact of compost addition on soil nitrogen, organic matter, and carbon storage

At both experiments CGW contributed to changes in soil development compared with controls, including increased soil organic matter (+ 0.5 – 0.8 %), total carbon (+ 6.3 – 8.5 t-ha^{-1}), total nitrogen (+ 0.4 – 0.7 t-ha^{-1}), and available nitrogen (+ 0.5 – 0.9 kg-ha^{-1}). These increases were detected after 20-years, outlining the existence of CGW induced soil-based legacy effects.

The extent of composts influence in soil depends on multi-layer interactions, the magnitude of which is modulated by site conditions. At Immingham site conditions supported a virtuous circle of soil development, driven by interactions between compost, ALDER, SYCAMORE, and earthworms. At Immingham these interactions peaked in SYCAMORE-SHALLOW, where soil chemistry was significantly different from ALDER-DEEP in several regards. Namely, soil carbon (+ 5.0 t-ha^{-1}), total nitrogen (+ 0.3 t-ha^{-1}), and available nitrogen (+ 0.5 kg-ha^{-1}), were all greater in SYCAMORE-SHALLOW. The present study argues SYCAMORE-SHALLOW reveals the factors driving site development on Humberside, and what happens when they converge. Indeed, peak C and N stocks recorded in SYCAMORE-SHALLOW at Immingham were produced by multiple factors interacting synergistically in soil. These factors were: ALDER overstory – (76 % survival, 10 – 11 m height), SYCAMORE understory – (77 % survival, 6 m height), ground-layer – (nitrophilous herbaceous plants), SHALLOW compost – (50, 100 t-ha^{-1}), and specific earthworm communities – (endogeic - 45 m^{-2}, anecic – 39 m^{-2}, epigeic – 16 m^{-2}). Although these factors were present at both experiments, the balance between them produced different dynamics at each site. Specifically, the ability of site conditions to increase or decrease ALDER survival appears to have strengthened or weakened synergies occurring across multiple layers. At Immingham site conditions maintained high ALDER survival, strengthening feedbacks and synergies occurring between ALDER and other layers resulting in a range of positive dynamics. Conversely at Winterton poor soil conditions and pathogens severely reduced ALDER survival disrupting these synergies, resulting in a breakdown between the 'compost-effect' and 'Alder-effect'. These dynamics are apparent in soil chemistry data and are now explored.

At Immingham, synergies among experimental factors were stronger than at Winterton, peaking in SYCAMORE-SHALLOW. The ‘Alder-effect’ remained crucial to these synergies, even though soils receiving direct SYCAMORE inputs produced the highest C and N stocks. Indeed, at Immingham SOM, total carbon, total nitrogen, and available nitrogen were comparable or greater under SYCAMORE than ALDER even though the latter dominated the site. However, ALDER’s superiority ensured it contributed to SYCAMORE soil chemistry through a variety of mechanisms. Namely 1). by increasing SYCAMORE foliar-N, 2). depositing ALDER foliage throughout the experiment, 3). promoting growth of herbaceous ground layer plants.

The herbaceous layer was indicative of ALDER’s widespread impact at Immingham and may have increased C and N storage via interactions with SHALLOW compost. For example, a study by Rawlik et al. (2018) undertaken in reclaimed mine spoils found *Alnus glutinosa* (common Alder) promoted growth of nitrophilous weeds, producing higher above-ground herb biomass and diversity than any other tree species studied. Woś et al. (2018) similarly found *A. glutinosa* promoted nitrophilous weed growth resulting in high herb biomass, but on this occasion, herb diversity was extremely low. Nevertheless, both studies show Alder’s influence on herb biomass is profound (Rawlik et al., 2018). Photographic evidence obtained during site monitoring shows similar effects occurred at Immingham wherein high ALDER survival coincided with vigorous herb layer growth **(FIGURE 7.73)**. Thus, ALDER’s impact on herb-layer, its ability to enrich SYCAMORE foliage, and subsequent interactions between litterfall, herb-layer, and SHALLOW compost demonstrate how multi-layer interactions increase soil C and N in SYCAMORE-SHALLOW.

Indeed, concerning the role of SHALLOW compost Biederman and Whisenant (2011) studied how amendment placement impacted soil restoration on landfills. They found surface application generated more carbon and nitrogen than incorporating amendments into the soil matrix. They concluded surface amendment promoted interactions between grasses, forbs, and soil microbes, thus increasing above-ground plant biomass and soil C and N. Though their study lasted just 2-years

FIGURE 7.73 *– Comparison of ground-layers at Humberside sites. At IMMINGHAM (top) ALDER survival was high and herbaceous plants dominated the ground-layer (photo taken May 2019). At WINTERTON (bottom) ALDER survival was low, SYCAMORE was more dominant and leaf-litter covered the ground-layer (photo taken August 2018).*

and examined wood-based amendments, it demonstrates the mechanisms and interactions underlying increased soil C and N. Namely, the placement of amendments into aerobic topsoils where biological activity is greatest promotes interactions among plants, amendments, and soil-organisms causing soil carbon and nitrogen accumulate in topsoil.

Crucially, at Immingham SYCAMORE-SHALLOW plots received a greater proportion of SYCAMORE foliage. This influx of recalcitrant litter likely regulated synergies occurring between ALDER, compost, and earthworms. According to Grossman et al. (2020), functionally diverse litter mixtures lose labile carbon more slowly than those containing functionally similar litter. This means SYCAMORE foliage provided a functional contrast to ALDER, slowing down resource turnover allowing C and N to accumulate. Overall, the SHALLOW layer of compost at Immingham was the meeting point for multiple variables, interacting in a highly concentrated manner. The 'Alder-effect' enriched SYCAMORE foliage and increased herbaceous ground cover, adding to the high concentration of organic matter already provided by SHALLOW compost. The combination of SHALLOW compost, ALDER, SYCAMORE, herbaceous ground cover, and earthworms appear responsible for peak C and N stocks in SYCAMORE-SHALLOW at Immingham. However, without ALDER or compost these dynamics would not have occurred as evidenced by experimental data from Winterton, outlining the importance of both effects in Humberside's multi-layer system.

Indeed, soil chemistry data from Winterton further illustrates how the 'Alder-effect' and 'compost-effect' drive results obtained from the Humberside experiments. However, in contrast to Immingham site conditions at Winterton *disrupted* synergistic interactions, causing them to collapse into isolated factors. This is demonstrated by CONTROL and 250 t-ha^{-1} treatments which respectively highlight the 'Alder-effect' and 'compost-effect' operating separately. The 'Alder-effect' manifests in ALDER-CONTROL treatments only, where compost was absent. Conversely the 'compost-effect' was only present in SYACMORE 250 t-ha^{-1}, where ALDER was absent. Soil total-N at Winterton outlines the 'Alder-effect' at Winterton, being highest in ALDER-CONTROL (2.3 t-ha^{-1}) where ALDER survival was incidentally highest (53 %). Then, when ALDER survival declined so did total-N, reaching its lowest (1.6 t-ha^{-1}) in ALDER-250 t-ha^{-1} treatments. This shows soil-N increased and decreased in response to ALDER's presence or absence, proving soil-N is a proxy for the 'Alder-effect'. Further, as compost quantity increased at Winterton, soil total-N under ALDER declined highlighting the breakdown of synergies among compost and Alder. In ALDER-CONTROL plots the breakdown of synergy between ALDER and compost repeated for multiple soil variables. For example, ALDER-CONTROL had Winterton's highest SOM (6.0 %), total-N (2.3 t-ha^{-1}), available nitrogen (0.42 kg-ha^{-1}), and lowest soil C:N (13:1) illustrating a clear and consistent 'Alder-effect' in ALDER-CONTROL without compost. At Winterton this 'Alder-effect' was absent from all other treatments and high concentrations of compost even *reduced* ALDER survival, outlined by ALDER-250 t-ha^{-1} at Winterton which had the lowest Alder survival and lowest soil C and N stocks. Clearly, for peak C and N stocks to arise, synergy between compost and Alder is required.

SYCAMORE plots at Winterton provide further evidence of this breakdown in synergy, wherein the 'compost-effect' manifests only under SYCAMORE and never under ALDER. Indeed, whilst multiple soil chemical values were highest under ALDER-CONTROL, the only treatment capable of matching this was SYCAMORE 250 t-ha^{-1}. Indeed, under SYCAMORE 250 t-ha^{-1} soil organic matter (5.5 %), total-C (28.7 t-ha^{-1}), and total-N (2.0 t-ha^{-1}) were statistically comparable or greater than ALDER-CONTROL, whereas ALDER-250 t-ha^{-1} had the lowest soil C and N stocks of all. This means the 'compost-effect' manifested only when ALDER was absent, whilst the Alder-effect' manifested only when compost was absent. Thus, after 20-years at Winterton the Humberside experiments most important dynamics have disassociated, and the site increasingly shifts towards dominance by SYCAMORE.

Developments at Winterton demonstrate what can happen when the 'Alder-effect' is removed from a restored landfill site. Soil C and N stocks are lower and understory species proliferate, whilst the transition from one dominant tree-species to another expending carbon.

Nevertheless, despite low ALDER survival at Winterton after 20-years, the 'Alder-effect' likely influenced site and soil development presently observed. This is apparent because dead ALDER (visible standing and laid out across the ground) are comparable in size to surviving ALDER trees. This suggests many Italian alder died *after* reaching a substantial height, giving ALDER time to influence site development. Indeed, after the first 4-years of site observations Foot et al. (2003) found Winterton had the highest ALDER survival (74 %), and joint highest ALDER growth (2.12 m) on Humberside. In all likelihood, ALDER losses occurred after Foot et al. (2003) study concluded, giving ALDER ample time to interact with compost. Hence, at Winterton ALDER likely improved ground conditions for SYCAMORE, facilitating woodland succession. ALDER can assist in the creation of moist, nutrient-rich soils SYCAMORE prefers (Caudullo and Mauri, 2016; Pasta et al., 2016). This is evidenced by higher-than-expected SYCAMORE survival throughout Winterton, demonstrating ALDER's value as a nurse species prior to eventual demise by widespread losses.

Yet ultimately, despite ALDER's positive influence at Winterton over time, poor ALDER survival resulted in lower soil C and N storage. Total carbon (– 7.5 t-ha^{-1}), total nitrogen (– 0.3 t-ha^{-1}), available nitrogen (– 0.8 kg-ha^{-1}), and earthworm abundance (– 52 m^{-2}), were lower at Winterton, despite SYCAMORE's growing influence. The peak C and N stocks recorded in SYCAMORE-SHALLOW at Immingham failed to materialize at Winterton despite greater SYCAMORE growth (+ 0.65 m) and higher than expected survival (+ 75 %) at the Winterton site. This clearly illustrates that greater SYCAMORE dominance alone does not increase soil C and N. Indeed, photographic evidence comparing Winterton with Immingham show that SYCAMORE has different impacts on ground layer dynamics to ALDER, with likely implications for C and N storage. When SYCAMORE was dominant its foliar litter covered Winterton's soils, limiting herbaceous plant growth and reducing earthworm abundance **(FIGURE 7.73)**. Indeed, van der Plas et al. (2016) associated SYCAMORE with reduced understory plant diversity and reduced earthworm biomass. Further, Rawlik et al. (2018) found SYCAMORE significantly reduced nitrophilous weeds causing woody shrubs to dominate the ground layer. These very effects are observed at Winterton and likely impact C and N storage explaining lower soil stocks at Winterton. Thus, multi-layer dynamics guided by ‘Alder-effect’ interactions with compost, are essential for delivering peak C and N stocks observed at Immingham. SYCAMORE undoubtedly remains important, but without the ‘Alder-effect’ peak C and N stocks are not possible. Interestingly earthworm populations respond to dynamics created by tree-species and compost, suggesting they provide a mechanism and mediate multi-layer synergies occurring above-ground and below. Earthworm community responses to dynamics driven by Alder and compost are now discussed in further detail.

7.9.3 Impact of compost addition on earthworm populations and soil development

After 20-years, experimental treatments established in 1998 continue to influence earthworm activity on Humberside. In particular, tree species and compost incorporation depth shaped earthworm community structure and abundance, with wider impacts on soil formation, nitrogen cycling, and carbon storage.

Tree species strongly influenced earthworm abundance, as evidenced by differences under ALDER and SYCAMORE both within and between sites. Within sites, abundance of endogeic (+ 16 – 21 m^{-2}) and epigeic (+ 7 – 14 m^{-2}) earthworms was always greater under ALDER, showing the 'Alder-effect' consistently elevated earthworm numbers regardless of site conditions. Comparing sites, at Winterton ALDER survival was low (38 %) and incidentally so was earthworm abundance (65 m^{-2}). The improved performance of SYCAMORE at Winterton may have contributed to this by lowering earthworm abundance further. In contrast, at Immingham ALDER survival was twice that of Winterton (76 %), with earthworm abundance near doubling (117 m^{-2}) in response to ALDERS greater survival. Thus, a widespread 'Alder-effect' existed at both sites but was most prevalent at Immingham as it persisted after 20-years. Here, its constant presence sustained multilayer interactions including deposition of N-rich Sycamore and Alder foliage, and proliferation of herbaceous understory plants thus maintaining high earthworm abundance at Immingham. This outlines the 'Alder-effects' systemic nature and its positive impacts on certain species of earthworm. Indeed, ALDER's presence consistently increased earthworm abundance, but when ALDER declined so too did the number of earthworms.

Differences in abundance of certain earthworm species in response to Alder and Sycamore can be explained by their feeding preferences. Rajapaksha et al. (2013) examined which leaf-litter earthworms preferred from six different tree species and found common alder *(A. glutinosa)* leaf-litter was one of the most favoured foliar foods for all four earthworm species studied (*A. caliginosa, A. chlorotica, A. longa, and L. terrestris*). In contrast Sycamore foliage was one of the least preferred.

Three of the four earthworm species studied by Rajapaksha et al. (2013) were among the most abundant earthworms on Humberside (*A. caliginosa, A. chlorotica*, *A. longa*). This suggests the 'Alder-effect' elevated earthworm abundance by providing a favoured edible foliar food resource.

Indeed, the feeding behaviours documented by Rajapaksha et al. (2013) explain numerous dynamics observed on Humberside. Namely, SYCAMORE's tendency to have lower endogeic (*A. caliginosa, A. chlorotica*) and epigeic (*L. castaneus*) abundance than ALDER, and the fact anecic abundance (*A. longa*) remained similar under both trees. Concerning the latter, Rajapaksha et al. (2013) found when presented with a variety of foliage, the only earthworm to consume almost all SYCAMORE foliage (92 %) inside a period of 28-days was *A. longa*. Incidentally, this endo-anecic species was highly abundant under both tree species at both experiments on Humberside, outlining *A. longa's* ability to take advantage of both ALDER and SYCAMORE foliage on restored landfills regardless of which tree dominates. In contrast, Rajapaksha et al. (2013) found *A. chlorotica* consumed the least SYCAMORE foliage (37 %) within a 28-day period of the earthworm species studied, suggesting *A. chlorotica* might be less abundant when feeding on Sycamore leaves. In fact, this very dynamic occurred on Humberside, with *A. chlorotica* abundance low (8 m^{-2}) at Winterton where SYCAMORE was prevalent. Conversely, at Immingham *A. chlorotica* was the second most abundant earthworm (38 m^{-2}), most likely because Alder dominated this site for 20-years. Evidently, earthworm population dynamics on Humberside closely match Rajapaksha et al. (2013) findings, meaning these lab-based findings have been repeated at field-scale in a restoration context, providing restoration practitioners with evidence useful for predicting outcomes in the field and their impact on ecosystem services. Indeed, given Alder increased the abundance of multiple earthworm species on Humberside (*A. caliginosa, A. chlorotica*, *A. longa,* and *L. castaneus*), the above earthworm species appear to provide a biological mechanism that can mediate multiple-dynamics observed on Humberside. When establishing woodland on former landfill, different proportions of Italian alder and Sycamore in a planting mixture could be combined with the earthworm species identified in this study to re-instate select ecological processes and services depending on the outcomes desired.

Another notable dynamic apparent at both Humberside sites was the way in which earthworms responded to the presence of CGW and incorporation depth which in turn mediated CGW impact on multiple ecosystem services. Earthworms are known to incorporate organic matter into the soil matrix, through a variety of feeding, burrowing, bioturbation, and aggregate formation activities (Blouin et al., 2013). Data from both sites regarding the impact of compost quantity on earthworm abundance shows that when compost is added earthworm abundance rises above levels observed in controls. The threshold is different at each experiment, being 50 t-ha^{-1} at Immingham, and 100 t-ha^{-1} at Winterton. The higher threshold at Winterton could be explained by a much weaker 'Alder-effect' at the time of sampling. With Alder deaths high at this site the 'Alder-effect' has likely begun to fade hence lack of synergy between compost and Alder leaves earthworms more reliant on compost. In short, with Alder's influence fading more CGW is required to create detectable legacy effects on earthworms.

Interestingly, incorporation depth data from both sites suggests earthworm communities shift with the change in compost depth. The community appears to respond directly to the distribution and concentration of compost throughout the profile, as evidenced by changes in abundance and community structure with impacts on ecosystem services. For example, in ALDER-DEEP all three eco-groups are abundant. Conversely, in ALDER-SHALLOW only endogeics maintain high abundance. It is likely that because ALDER-DEEP distributes and spreads organic resources throughout the soil profile, earthworms similarly spread out in response to the resource base. This broad distribution of resources allows earthworms to spread out and create burrow systems which come under less intense competition from other earthworms. Conversely, when organic resources are concentrated in the upper horizon (as with ALDER-SHALLOW), earthworm populations move up the soil profile towards these resources. With resources concentrated in a narrow band of soil competition is more intense. There is less space for a functionally diverse species composition resulting in a community dominated by endogeics. In general, earthworm abundance is greater under ALDER because litter is more labile and soils are N-rich, but surprisingly C and N stocks are lower. Conversely, in SYCAMORE-SHALLOW an increase in SYCAMORE foliage reduces the 'Alder-effect' lowering earthworm abundance, regulating feedbacks between Alder, compost, and earthworms resulting in peak C and N stocks in SYCAMORE-SHALLOW. The high concentration of compost in SYCAMORE-SHALLOW and ALDER-SHALLOW has similar impacts on earthworm community structure. Yet, with greater contributions from ALDER in the latter, the 'Alder-effect' causes a cascade of multi-layer interactions which stimulates concentrated endogeic activity but reduces earthworm community abundance and lowers soil C and N stocks.

This is important because if we imagine a cross section through the ecosystem, we begin to see how tree species, compost incorporation depth, and earthworm dynamics respond to one another, with above-ground and below-ground interactions operating together in a cycle tied together through the activity of earthworms. Understanding and envisioning this interactive multi-layer system is important because it shows restoration practitioners how choices made during restoration come together, play-out, and exert specific effects on ecosystem functions and services. The fact these changes remain detectable at 20-years demonstrates the existence of legacy effects throughout the entire reclamation system.

8.0 Final discussion

8.1 Collective discussion from all study sites

Results from five study sites demonstrate one-off application of CGW can improve tree growth, whilst providing a range of soil physical, chemical, and biological benefits. CGW impacts remain detectable at 5, 10, and 20 years, however composts degree of influence depends on application quantity and depth. Adjustments to either of these result in trade-offs and deliver contrasting benefits. Deep applications improve tree growth, whereas shallow applications promote growth of ground vegetation leading to increased carbon storage. Compost also has temporal effects which can dominate site development from 0 to 5 years. The period of dominance can depend on CGW quantity, with higher rates liable to last longer and have a greater impact. Indeed, from 0 to 10-years compost interactions with ground vegetation and earthworms govern primary production, soil formation, nutrient cycling, and carbon storage, and throughout this period the tree effect is generally small. However, as the canopy closes and natural processes establish, composts influence diminishes. Between 10 and 20-years, the tree effect grows in significance, suggesting an inverse relationship between 1). the effects of compost, and 2). The effect of trees over time. Pioneers such as Italian alder and Silver birch are highly compatible with compost and earthworms, benefitting from interactions between the two whilst suppressing growth of slower growing understory species. The subsequent decline of Italian alder at one site after 20-years allowed Sycamore to emerge from the understory and become co-dominant highlighting the potential importance of secondary understory species to long-term site development. Nevertheless, even though faster growing species confined Sycamore to the understory, Sycamore still benefitted from compost application between 0 and 20-years.

Overall, compost has a consistent impact, which becomes more obvious at higher application rates. Indeed, when used in sufficient quantities, the CGW effect is characterised as low C:N ratio (<20:1), high nitrogen (2.0 t-ha), neutral to base pH (7-8), increased organic matter and carbon storage, compatibility with pioneer trees, and proliferation of grasses, herbaceous plants, and burrowing earthworms. CGW seems best suited to fast growing, high turnover, fertile systems where organic matter is decomposed quickly, and nutrients are freely available (Wardle et al., 2004; Reich, 2014; Semchenko et al., 2018). Indeed, the greatest gains in tree growth and carbon storage occur under 600 t-ha^{-1} treatments, when earthworms and grasses are present. This suggests when large quantities of compost are available, a synergistic interaction occurs between understory vegetation, earthworms, trees, and CGW. Italian alder and Silver birch benefit most from this system because they are fast growing species with labile litter allowing them to integrate successfully into competitive, resource rich environments. Functional traits including tree canopy density may also benefit this system, with Silver birch in particular having an open and sparse canopy. This allows heat and light to reach the soil surface, benefitting understory plants. This probable influx of radiative and photosynthetic energy benefits ground vegetation and earthworms, further explaining why Silver birch is compatible with the standard reclamation practice of establishing herbaceous and grassland cover over newly manufactured soil (Bending et al., 1999; Bradshaw and Hüttl, 2001).

Interactions between plants and soil organisms are needed for CGW to become fully integrated into an aggregated soil structural matrix (Scullion and Malik, 2000; Pey et al., 2013; Deeb et al., 2017). The role of trees in this process changes over time. For example, when compost is incorporated into manufactured soil at project outset, newly planted saplings make a minimal contribution to soil pedogenesis. In the early stages even fast growing species like Italian alder and Silver birch need 5-years before meaningful interactions with site developmental processes can be seen (Bending and Moffat, 1997). Consequently, throughout this early period herbaceous plants and grasses dominate primary production, providing the bulk of plant inputs into the reclaimed soil, making a notable contribution to soil development early on. Within these first 5-years compost also plays a dominant role, promoting growth of all plants. CGW provides available nutrients and carbon rich compounds which support plant-microbial interactions, and recovery of ecological processes

(Beesley, 2012; Haynes et al., 2015; Lorenz et al., 2021). Beyond this initial 5-year period composts impacts on tree growth become increasingly indirect. Rather than supplying nutrients which stimulate tree growth benefits arise through interactions with soil biology and ground vegetation, as this stimulates ongoing soil structural development and turnover of resources. Indeed, even when application rates are doubled tree growth does not increase, highlighting the limitations of one-off compost application and the temporary nature of direct effects.

Overall, when compost accounts for 15 – 25 % total soil volume it provides its greatest ecosystem-wide benefits. For sites included in the present study, this was 250, 300, or 600 t-ha^{-1}. This agrees with guidance on application rates provided by WRAP (2010) and is a fairly substantial quantity of compost. Potential benefits from large application rates include additional gains in carbon (6 – 15 t-ha^{-1}), nitrogen (0.5 – 2.0 t-ha^{-1}), organic matter (0.5 – 3.0 %), and tree height (0.5 – 2.0 m^{-1}). However, beyond 30% compost volume, tree growth and soil development rarely receive further benefit. Indeed, increasing CGW quantity from 15%, to 20% or 25% does not provide linear improvements. Instead, the margin of benefit diminishes. In particular, when using a large one-off application of CGW, effects on tree growth level out at around 20% compost. It should be noted that recommended application volumes of 15 – 25% risk contamination of watercourses and encourage proliferation of ground vegetation. Yet, rather than perceiving the latter as detrimental, increased growth of ground cover plants can offer significant benefits in terms of soil formation, nutrient cycling, and carbon storage.

Application depth should be considered alongside CGW quantity, with different application depths providing different benefits. The present study found compost incorporation throughout the upper 0.5 m provided the greatest benefit to tree growth, which is in agreement with guidance from WRAP (2010). Indeed, deep incorporation increases tree height by 1.0 m, when compared with shallow application. However, shallow applications increase carbon and nitrogen stocks by concentrating resources in ground vegetation’s root-zone, and in biologically active aerobic soil layers. Indeed, wherever trees leave space for ground vegetation, either through slow growth, senescence, or a diffuse canopy; ground vegetation interactions with CGW and earthworms promotes soil regeneration, especially during the first 10-years of site establishment.

In continuation, altering application quantity and depth impacts carbon and nitrogen storage over a 20-year period. For example, compared with deep incorporation (0.5 – 0.6 m), a shallow application (to 0.1 m) concentrates CGW around the root-zone, stimulating plant-faunal-microbial interactions. This can significantly increase carbon storage and nitrogen cycling in the upper 15 cm of the soil, as Biederman and Whisenant (2011) also found. Indeed, carbon and nitrogen stocks in topsoil can be increased using moderate shallow applications (100 t-ha). Indeed, by incorporating 10 – 15 % CGW per total soil volume into the upper 15 cm topsoil, plant-faunal-microbial interactions integrate compost into the soil matrix. Consequently, when applying a recommended volume of compost (i.e. 15 – 30 %), one third of the chosen quantity could be concentrated in the upper 15 cm of the profile to promote ground vegetation, whilst the remaining material could be incorporated evenly to 0.6 m depth benefitting deeper rooting trees.

Regarding the compost-effect, this is strong in years 0 to 10, but during years 10 to 20 the tree-species effect grows. Earthworms mediate carbon and nitrogen increases associated with higher rates of compost along with Italian alders' presence, evidenced by significantly higher endogeic density under Italian alder and production of mull soils wherever earthworms are present. Indeed, the highest carbon (40.9 t-ha^{-1}) and nitrogen (2.7 t-ha^{-1}) stocks found by this study were recorded when 25% compost by soil volume was incorporated to 0.6 m depth. These gains likely accrued over 10-years and were only recorded in earthworms' presence. Conversely, when earthworms were absent pockets of compost were found intact beneath the surface and C and N stocks were lower, highlighting the risk of compost under-utilisation in earthworms' absence. Conflict with WIN and IMM. Presence of nitrogen rich Alder plus CGW, may stimulate greater earthworm activity resulting in lower carbon and nitrogen stocks.

For nitrogen vulnerable zones (NVZ), application of the recommended 15 – 30 % volume is not possible due to risk of watercourse contamination. According to Beesley (2012) this may be mitigated by incorporating compost to depth. Thus, for NVZs one obvious solution would be to incorporate a permitted volume of compost throughout the soil profile during the soil manufacturing phase, followed by one or more smaller shallow applications in subsequent years. Doing this would allow the recommended 15-30 % volume to be applied over an extended period. Indeed, compost does appear to be more stable below the soil surface as abiotic weathering is reduced. In contrast, surface application concentrates CGW where plants and soil organisms are more active and abiotic weathering is greater increasing the mobility of nutrients. To mitigate this, earthworm migration could be monitored, so that follow-up applications could be timed to coincide with faunal appearance on-site thus reducing the risk of hydrological contamination by poorly incorporated compost. This means migrating earthworms would also receive fresh food resources as they re-colonise. At this stage earthworm inoculation onto sites may be considered, especially where reclaimed sites are effectively islands with little chance of natural colonisation. Nevertheless, earthworm species/ecological categories must be matched to the given site and soil conditions, with natural colonisation generally preferable for this purpose (Butt, 2008; Butt and Putwain, 2017; Butt and Quigg, 2021).

Clearly, a degree of compatibility has been found between CGW, grasses, herbaceous plants, Italian alder, Silver birch, and earthworms. Collectively, the study sites characterise CGW combined impact on site dynamics from 0 to 20-years. In doing so, demonstration has been provided of suitable uses for PAS-100 CGW, in terms of different application strategies, and what the likely effects will be. It should be stressed that when earthworms are absent such as at GOH-SOUTH, contrasting site dynamics can be observed, with soil pedogenesis and nutrient cycling processes differing significantly and CGW remaining underutilised. Indeed, compost underutilisation, as indicated by intact compost remaining below the soil surface after 10 years at Greenoakhill South, alongside deficiencies in foliar-N at the same site, strongly suggest slower organic matter turnover and nutrient cycling when earthworm bioturbation is absent. Compost underutilisation coincided with lower tree growth, and lower carbon and nitrogen stocks, but an overall restoration trajectory comparable to sites colonised by earthworms. Consequently, an acceptable degree of restoration can be provided in earthworms absence using less CGW. Indeed, composts characteristic effect on N-cycling and other processes may be un-desirable when restoration goals diverge from fast growing, nutrient rich systems. Just as certain tree species, soil-fauna, and microbes favour certain soil conditions, CGW appears to possess characteristics suitable for certain aims and ecological contexts. Indeed, CGW properties reflect the composting processes, input materials, and microbial metabolism that created it. Thus, CGW seems ideal for systems which might naturally develop SOM with similar physical, chemical, and biological characteristics to that produced by PAS-100 composting techniques. Thus, matching organic amendments with plants and soil organisms that might naturally synthesise similar organic substrates, creates positive feedbacks which favour proliferation of CGW compatible organisms. The result would be ongoing production of organic material that is chemically and biologically similar to CGW (Hättenschwiler et al., 2005; Frouz et al., 2015; Schelfhout et al., 2017; Steidinger et al., 2019; Liang et al., 2017; Angst et al., 2021; Lorenz et al., 2021).

9.0 Conclusion

In conclusion, this study investigated composted green wastes (CGW) legacy impact on reclaimed ecosystem development. Data reveals the compost-effect is systemic, influencing multiple supporting and regulating services simultaneously. Soon after application, compost increases tree growth, and carbon and nitrogen stocks. The effect is short acting but remains detectable over timescales of decades. Over the timeframes observed, interactions between compost, soil organisms, and ground vegetation are the primary mechanism for soil formation, carbon storage, and nitrogen cycling. The tree-effect on ecosystem development is initially small but increases as woodland matures. Large *concentrations* of compost (15-30 %) are more effective at stimulating earthworm activity and ground vegetation growth, and together these factors accelerate formation of mull-soil and increase carbon and nitrogen stocks. Indeed, PAS-100 CGW has consistent and specific impacts, offering practitioners a predictable method for improving physical, chemical, and biological quality of reclaimed soils. Data suggests that when CGW is applied to reclaimed landfill it increases soil fertility and organic matter turnover, through positive feedback between ground covering vegetation, earthworms, and latterly trees. The compost-effect is also self-reinforcing, possessing properties that provide habitat conditions and resources desired by certain plant groups, including non-acidic grasses, early successional herbaceous, and fast-growing plants that produce labile biomass. Such plants appear to proliferate when fertility is high, and nutrients are freely available, with earthworm bioturbation promoting development of these very conditions. This so called 'CGW-effect' is characterised as low C:N ratio, high nitrogen, neutral to base pH, with increased organic matter and carbon storage. CGW is most compatible with pioneer trees, grasses, herbaceous plants, and burrowing earthworms. Silver birch and Italian alder perform well with CGW as traits including fast growth, high quality foliage, and short lifespans make them effective competitors in these fast growing high-turnover systems. Indeed, the plant and soil resources which restoration practitioners assemble may be conceptualised as 'reclamation systems', namely, collections of complimentary plants, soil forming materials, and restoration techniques, with organic amendments being key components of these. Where reclamation components are complimentary to one another and suited to site conditions, ecosystem integrity will likely improve. By adjusting amendment type, quantity, and depth, soil organisms, ground vegetation, and canopy and understorey tree species, it may be possible to create different ‘reclamation systems’ for different applications.

10.0 References

ABDUL-KAREEM, A. W. & MCRAE, S. G. 1984. The effects on topsoil of long-term storage in stockpiles. *Plant and Soil,* 76**,** 357-363.

AERTS, R. & HONNAY, O. 2011. Forest restoration, biodiversity and ecosystem functioning. *BMC Ecology,* 11**,** 1-10.

ALEXANDER, S., ARONSON, J., WHALEY, O. & LAMB, D. 2016. The relationship between ecological restoration and the ecosystem services concept. *Ecology and Society,* 21.

ALI, H., KHAN, E. & SAJAD, M. A. 2013. Phytoremediation of heavy metals--concepts and applications. *Chemosphere,* 91**,** 869-81.

ANGST, G., MUELLER, K. E., NIEROP, K. G. J. & SIMPSON, M. J. 2021. Plant- or microbial-derived? A review on the molecular composition of stabilized soil organic matter. *Soil Biology and Biochemistry,* 156**,** 108189.

APONTE, C., GARCÍA, L. V. & MARAÑÓN, T. 2013. Tree species effects on nutrient cycling and soil biota: A feedback mechanism favouring species coexistence. *Forest Ecology and Management,* 309**,** 36-46.

ARAUJO, Y., LUIZÃO, F. J. & BARROS, E. 2004. Effect of earthworm addition on soil nitrogen availability, microbial biomass and litter decomposition in mesocosms. *Biology and Fertility of Soils,* 39**,** 146-152.

ASHWOOD, F. 2014. Best Practice Guidance Note 18 - Lowland Calcareous Grassland - Creation and management in land regeneration. *In:* GROUP, T. L. R. A. U. G. R. (ed.). Farnham, Surrey: Forest Research.

ASHWOOD, F. 2016. *WOODLAND RESTORATION ON LANDFILL SITES: EARTHWORM ACTIVITY AND ECOSYSTEM SERVICE PROVISION.* PhD, University of Central Lancashire.

ASHWOOD, F., BUTT, K. R., DOICK, K. J. & VANGUELOVA, E. I. 2017. Interactive effects of composted green waste and earthworm activity on tree growth and reclaimed soil quality: A mesocosm experiment. *Applied Soil Ecology,* 119**,** 226-233.

ASHWOOD, F., BUTT, K. R., DOICK, K. J. & VANGUELOVA, E. I. 2018. Effects of composted green waste on soil quality and tree growth on a reclaimed landfill site. *European Journal of Soil Biology,* 87**,** 46-52.

ASHWOOD, F. E., DOICK, K. J., ATKINSON, G. E. & CHENOWETH, J. 2014. Under-utilisation of organic wastes during brownfield regeneration to community woodland: tackling the barriers. *Waste Manag Res,* 32**,** 49-55.

BASLER, A., DIPPOLD, M., HELFRICH, M. & DYCKMANS, J. 2015. Microbial carbon recycling – an underestimated process controlling soil carbon dynamics – Part 1: A long-term laboratory incubation experiment. *Biogeosciences,* 12**,** 5929-5940.

BECK, P., CAUDULLO, G., DE RIGO, D. & TINNER, W. 2016. Betula pendula, Betula pubescens and other birches in Europe: distribution, habitat, usage and threats. *In:* SAN-MIGUEL-AYANZ, J. (ed.) *European Atlas of Forest Tree Species.* Luxembourg.

BEESLEY, L. 2012. Carbon storage and fluxes in existing and newly created urban soils. *J Environ Manage,* 104**,** 158-65.

BENAYAS, J. M. R., NEWTON, A. C., DIAZ, A. & BULLOCK, J. M. 2009. Enhancement of Biodiversity and Ecosystem Services by Ecological Restoration: A Meta-Analysis. *Science,* 325**,** 1121-1124.

BENDING, N., MCRAE, S. & MOFFAT, A. 1999. *Soil-forming materials: their use in land reclamation,* London, The Stationary Office.

BENDING, N. & MOFFAT, A. 1997. *Tree establishment on landfill sites: Research and updated guidance*, Department of the Environment, Transport and the Regions.

BERG, B. 2000. Litter decomposition and organic matter turnover in northern forest soils. *Forest Ecology and Management,* 133**,** 13-22.

BIEDERMAN, L. A. & WHISENANT, S. G. 2011. Amendment Placement Directs Soil Carbon and Nitrogen Cycling in Severely Disturbed Soils. *Restoration Ecology,* 19**,** 360-370.

BIRKHOFER, K., SCHONING, I., ALT, F., HEROLD, N., KLARNER, B., MARAUN, M., MARHAN, S., OELMANN, Y., WUBET, T., YURKOV, A., BEGEROW, D., BERNER, D., BUSCOT, F., DANIEL, R., DIEKOTTER, T., EHNES, R. B., ERDMANN, G., FISCHER, C., FOESEL, B., GROH, J., GUTKNECHT, J., KANDELER, E., LANG, C., LOHAUS, G., MEYER, A., NACKE, H., NATHER, A., OVERMANN, J., POLLE, A., POLLIERER, M. M., SCHEU, S., SCHLOTER, M., SCHULZE, E. D., SCHULZE, W., WEINERT, J., WEISSER, W. W., WOLTERS, V. & SCHRUMPF, M. 2012. General relationships between abiotic soil properties and soil biota across spatial scales and different land-use types. *PLoS One,* 7**,** e43292.

BLANCHART, E., ALBRECHT, A., ALEGRE, J., DUBOISSET, A., GILOT, C., PASHANASI, B., LAVELLE, P. & BRUSSAARD, L. 1999. Effects of earthworms on soil structure and physical properties. *Earthworm management in tropical agroecosystems,* 5**,** 149-171.

BLOODWORTH, A. J., SCOTT, P. W. & MCEVOY, F. M. 2009. Digging the backyard: Mining and quarrying in the UK and their impact on future land use. *Land Use Policy,* 26**,** S317-S325.

BLOUIN, M., HODSON, M. E., DELGADO, E. A., BAKER, G., BRUSSAARD, L., BUTT, K. R., DAI, J., DENDOOVEN, L., PERES, G., TONDOH, J. E., CLUZEAU, D. & BRUN, J. J. 2013. A review of earthworm impact on soil function and ecosystem services. *European Journal of Soil Science,* 64**,** 161-182.

BOBINAC, M., ANDRAŠEV, S., ŠUŠIĆ, N., BAUER-ŽIVKOVIĆ, A. & I JORGIĆ, Đ. 2020. Growth and structure of Italian alder (Alnus cordata) linear plantation at age 11 and 16 years at Fruška gora (Serbia). *Šumarski list,* 144 (9-10)**,** 455-462

BOTTINELLI, N. & CAPOWIEZ, Y. 2021. Earthworm ecological categories are not functional groups. *Biology and Fertility of Soils,* 57**,** 329-331.

BOTTINELLI, N., HEDDE, M., JOUQUET, P. & CAPOWIEZ, Y. 2020. An explicit definition of earthworm ecological categories – Marcel Bouché's triangle revisited. *Geoderma,* 372**,** 114361.

BOTTINELLI, N., JOUQUET, P., CAPOWIEZ, Y., PODWOJEWSKI, P., GRIMALDI, M. & PENG, X. 2015. Why is the influence of soil macrofauna on soil structure only considered by soil ecologists? *Soil and Tillage Research,* 146**,** 118-124.

BOUCHÉ, M. 1977. Strategies lombriciennes. *Ecological Bulletins***,** 122-132.

BOUCHÉ, M. B. 1972. *Lombriciens de France: écologie et systématique*, Institut national de la recherche agronomique Paris.

BOX, J. 1998. Setting objectives and monitoring for ecological restoration and habitat creation. *In:* FOX, H. R., MCINTOSH, A. D. & MOORE, H. M. (eds.) *In: Land Reclamation: Achieving Sustainable Benefits.* Taylor & Francis.

BOYER, S., WRATTEN, S., PIZEY, M. & WEBER, P. 2011. Impact of soil stockpiling and mining rehabilitation on earthworm communities. *Pedobiologia,* 54**,** S99-S102.

BRADSHAW, A. 2000. The use of natural processes in reclamation — advantages and difficulties. *Landscape and Urban Planning,* 51**,** 89-100.

BRADSHAW, A. D. 1983. The Reconstruction of Ecosystems. *Journal of Applied Ecology,* 20**,** 1-17.

BRADSHAW, A. D. 1987. Restoration: an acid test for ecology.

BRADSHAW, A. D. 1997. Restoration of mined lands - using natural processes. *Ecological Engineering,* 8**,** 255–269.

BRADSHAW, A. D. 1998. From green cover to something more - The development of principles and practice in land reclamation. *In:* FOX, H. R., MCINTOSH, A. D. & MOORE, H. M. (eds.) *In: Land Reclamation: Achieving Sustainable Benefits.* Taylor & Francis.

BRADSHAW, A. D. & CHADWICK, M. J. 1980. *The restoration of land : the ecology and reclamation of derelict and degraded land,* Oxford, Oxford : Blackwell Scientific.

BRADSHAW, A. D. & HÜTTL, R. F. 2001. Future minesite restoration involves a broader approach. *Ecological Engineering,* 17**,** 87-90.

BRAND, J. H., SPENCER, K. L., O'SHEA, F. T. & LINDSAY, J. E. 2018. Potential pollution risks of historic landfills on low-lying coasts and estuaries. *Wiley Interdisciplinary Reviews: Water,* 5.

BRIONES, M. J. I. 2018. The Serendipitous Value of Soil Fauna in Ecosystem Functioning: The Unexplained Explained. *Frontiers in Environmental Science,* 6.

BROWN, G. G., BAROIS, I. & LAVELLE, P. 2000. Regulation of soil organic matter dynamics and microbial activityin the drilosphere and the role of interactionswith other edaphic functional domains§§Paper presented at the 16th World Congress of Soil Science, 20–26 August 1998, Montpellier, France. *European Journal of Soil Biology,* 36**,** 177-198.

BRUSSAARD, L., PULLEMAN, M. M., OUÉDRAOGO, É., MANDO, A. & SIX, J. 2007. Soil fauna and soil function in the fabric of the food web. *Pedobiologia,* 50**,** 447-462.

BUCK, C., LANGMAACK, M. & SCHRADER, S. 2000. Influence of mulch and soil compaction on earthworm cast properties. *Applied Soil Ecology,* 14**,** 223-229.

BÜNEMANN, E. K., BONGIORNO, G., BAI, Z., CREAMER, R. E., DE DEYN, G., DE GOEDE, R., FLESKENS, L., GEISSEN, V., KUYPER, T. W., MÄDER, P., PULLEMAN, M., SUKKEL, W., VAN GROENIGEN, J. W. & BRUSSAARD, L. 2018. Soil quality – A critical review. *Soil Biology and Biochemistry,* 120**,** 105-125.

BURTON, V., MOSELEY, D., BROWN, C., METZGER, M. J. & BELLAMY, P. 2018. Reviewing the evidence base for the effects of woodland expansion on biodiversity and ecosystem services in the United Kingdom. *Forest Ecology and Management,* 430**,** 366-379.

BUTENSCHOEN, O., POLL, C., LANGEL, R., KANDELER, E., MARHAN, S. & SCHEU, S. 2007. Endogeic earthworms alter carbon translocation by fungi at the soil–litter interface. *Soil Biology and Biochemistry,* 39**,** 2854-2864.

BUTT, K. 1999. Inoculation of Earthworms into Reclaimed Soils - The UK Experience. *Land Degradation and Development,* 10**,** 565-575.

BUTT, K. & PUTWAIN, P. 2017. Earthworm community development in organic matter-amended plots on reclaimed colliery spoil. *North West Geography,* 17**,** 1-8.

BUTT, K. R. 2008. Earthworms in Soil Restoration: Lessons Learned from United Kingdom Case Studies of Land Reclamation. *Restoration Ecology,* 16**,** 637-641.

BUTT, K. R. & BRIONES, M. J. I. 2017. Earthworms and mesofauna from an isolated, alkaline chemical waste site in Northwest England. *European Journal of Soil Biology,* 78**,** 43-49.

BUTT, K. R. & GRIGOROPOULOU, N. 2010. Basic Research Tools for Earthworm Ecology. *Applied and Environmental Soil Science,* 2010**,** 1-12.

BUTT, K. R., LOWE, C. N., FREDERICKSON, J. & MOFFAT, A. J. 2004. The development of sustainable earthworm populations at Calvert landfill site, UK. *Land Degradation & Development,* 15**,** 27-36.

BUTT, K. R. & QUIGG, S. M. 2021. Earthworm community development in soils of a reclaimed steelworks. *Pedosphere,* 31**,** 384-390.

CAPOWIEZ, Y., BOTTINELLI, N., SAMMARTINO, S., MICHEL, E. & JOUQUET, P. 2015. Morphological and functional characterisation of the burrow systems of six earthworm species (Lumbricidae). *Biology and Fertility of Soils,* 51**,** 869-877.

CAPOWIEZ, Y., SAMMARTINO, S. & MICHEL, E. 2011. Using X-ray tomography to quantify earthworm bioturbation non-destructively in repacked soil cores. *Geoderma,* 162**,** 124-131.

CARO, G., DECAËNS, T., LECARPENTIER, C. & MATHIEU, J. 2013. Are dispersal behaviours of earthworms related to their functional group? *Soil Biology and Biochemistry,* 58**,** 181-187.

CARRILLO, Y., BALL, B. A., BRADFORD, M. A., JORDAN, C. F. & MOLINA, M. 2011. Soil fauna alter the effects of litter composition on nitrogen cycling in a mineral soil. *Soil Biology and Biochemistry,* 43**,** 1440-1449.

CAUDULLO, G. & DE RIGO, D. 2016. European Atlas of Forest Tree Species - Acer platanoides in Europe: distribution, habitat, usage and threats. *In:* SAN MIGUEL AYANZ, J., DE RIGO, D., CAUDULLO, G., HOUSTON DURRANT, T. & MAURI, A. (eds.) *European Atlas of Forest Tree Species.* Luxembourg: Official EU.

CAUDULLO, G. & MAURI, A. 2016. European Atlas of Forest Tree Species - Alnus cordata in Europe: distribution, habitat, usage and threats. *In:* SAN MIGUEL AYANZ, J., DE RIGO, D., CAUDULLO, G., HOUSTON DURRANT, T. & MAURI, A. (eds.) *European Atlas of Forest Tree Species.* Luxembourg: Official EU.

CECH, T. & HENDRY, S. 2003. A review of diebacks and declines of alder (Alnus spp.) in Europe. *Forestry Commission Bulletin***,** 15-24.

CHATELAIN, M. & MATHIEU, J. 2017. How good are epigeic earthworms at dispersing? An investigation to compare epigeic to endogeic and anecic groups. *Soil Biology and Biochemistry,* 111**,** 115-123.

COPPIN, N. J. & BOX, J. 1998. After-use options for reclaimed land - A land suitability approach. *In:* FOX, H. R., MCINTOSH, A. D. & MOORE, H. M. (eds.) *In: Land Reclamation: Achieving Sustainable Benefits.* Taylor & Francis.

CRAVEN, D., THAKUR, M. P., CAMERON, E. K., FRELICH, L. E., BEAUSÉJOUR, R., BLAIR, R. B., BLOSSEY, B., BURTIS, J., CHOI, A., DÁVALOS, A., FAHEY, T. J., FISICHELLI, N. A., GIBSON, K., HANDA, I. T., HOPFENSPERGER, K., LOSS, S. R., NUZZO, V., MAERZ, J. C., SACKETT, T., SCHARENBROCH, B. C., SMITH, S. M., VELLEND, M., UMEK, L. G. & EISENHAUER, N. 2017. The unseen invaders: introduced earthworms as drivers of change in plant communities in North American forests (a meta-analysis). *Glob Chang Biol,* 23**,** 1065-1074.

CRAWFORD, J. W., DEACON, L., GRINEV, D., HARRIS, J. A., RITZ, K., SINGH, B. K. & YOUNG, I. 2012. Microbial diversity affects self-organization of the soil-microbe system with consequences for function. *J R Soc Interface,* 9**,** 1302-10.

CUI, J., ZHU, Z., XU, X., LIU, S., JONES, D. L., KUZYAKOV, Y., SHIBISTOVA, O., WU, J. & GE, T. 2020. Carbon and nitrogen recycling from microbial necromass to cope with C:N stoichiometric imbalance by priming. *Soil Biology and Biochemistry,* 142**,** 107720.

CUI, X. & SONG, J. 2007. Soil NH4+/NO3− nitrogen characteristics in primary forests and the adaptability of some coniferous species. *Frontiers of Forestry in China,* 2**,** 1-10.

CURRY, J. P., DOHERTY, P., PURVIS, G. & SCHMIDT, O. 2008. Relationships between earthworm populations and management intensity in cattle-grazed pastures in Ireland. *Applied Soil Ecology,* 39**,** 58-64.

CURRY, J. P. & SCHMIDT, O. 2007. The feeding ecology of earthworms – A review. *Pedobiologia,* 50**,** 463-477.

DA RONCH, F., CAUDULLO, G., TINNER, W. & DE RIGO, D. 2016. Larix decidua and other larches in Europe: distribution, habitat, usage and threats. *In:* SAN-MIGUEL-AYANZ, J. (ed.) *European Atlas of Forest Tree Species.* Luxembourg.

DARWIN, C. 1881. *The formation of vegetable mould through the action of worms: with observations on their habits,* London, John Murray.

DAYNES, C. N., FIELD, D. J., SALEEBA, J. A., COLE, M. A. & MCGEE, P. A. 2013. Development and stabilisation of soil structure via interactions between organic matter, arbuscular mycorrhizal fungi and plant roots. *Soil Biology and Biochemistry,* 57**,** 683-694.

DE VRIES, F. T., THEBAULT, E., LIIRI, M., BIRKHOFER, K., TSIAFOULI, M. A., BJORNLUND, L., BRACHT JORGENSEN, H., BRADY, M. V., CHRISTENSEN, S., DE RUITER, P. C., D'HERTEFELDT, T., FROUZ, J., HEDLUND, K., HEMERIK, L., HOL, W. H., HOTES, S., MORTIMER, S. R., SETALA, H., SGARDELIS, S. P., UTESENY, K., VAN DER PUTTEN, W. H., WOLTERS, V. & BARDGETT, R. D. 2013. Soil food web properties explain ecosystem services across European land use systems. *Proc Natl Acad Sci U S A,* 110**,** 14296-301.

DE WANDELER, H., BRUELHEIDE, H., DAWUD, S. M., DĂNILĂ, G., DOMISCH, T., FINÉR, L., HERMY, M., JAROSZEWICZ, B., JOLY, F.-X., MÜLLER, S., RATCLIFFE, S., RAULUND-RASMUSSEN, K., ROTA, E., VAN MEERBEEK, K., VESTERDAL, L. & MUYS, B. 2018. Tree identity rather than tree diversity drives earthworm communities in European forests. *Pedobiologia,* 67**,** 16-25.

DE WANDELER, H., SOUSA-SILVA, R., AMPOORTER, E., BRUELHEIDE, H., CARNOL, M., DAWUD, S. M., DĂNILĂ, G., FINER, L., HÄTTENSCHWILER, S., HERMY, M., JAROSZEWICZ, B., JOLY, F.-X., MÜLLER, S., POLLASTRINI, M., RATCLIFFE, S., RAULUND-RASMUSSEN, K., SELVI, F., VALLADARES, F., VAN MEERBEEK, K., VERHEYEN, K., VESTERDAL, L. & MUYS, B. 2016. Drivers of earthworm incidence and abundance across European forests. *Soil Biology and Biochemistry,* 99**,** 167-178.

DEEB, M., DESJARDINS, T., PODWOJEWSKI, P., PANDO, A., BLOUIN, M. & LERCH, T. Z. 2017. Interactive effects of compost, plants and earthworms on the aggregations of constructed Technosols. *Geoderma,* 305**,** 305-313.

DICKINSON, N. M., RAWLINSON, H., NOLAN, P. & PUTWAIN, P. 2004. Community forestry on old landfill sites. *Quarterly Journal of Forestry*.

DOICK, K. & WILLOUGHBY, I. 2011. Investigation into low tree establishment rates at Ingrebourne Hill's phase IIA restoration area. *Forest Research, Farnham*.

DOICK, K. J., SELLERS, G., CASTAN-BROTO, V. & SILVERTHORNE, T. 2009. Understanding success in the context of brownfield greening projects: The requirement for outcome evaluation in urban greenspace success assessment. *Urban Forestry & Urban Greening,* 8**,** 163-178.

DOMINATI, E., PATTERSON, M. & MACKAY, A. 2010. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecological Economics,* 69**,** 1858-1868.

DOUBE, B. M., SCHMIDT, O., KILLHAM, K. & CORRELL, R. 1997. Influence of mineral soil on the palatability of organic matter for lumbricid earthworms: A simple food preference study. *Soil Biology and Biochemistry,* 29**,** 569-575.

DROBIG, P. K. 1999. Making New Soils from Waste: The Humberside Project. *The Journal of North West Environmental Trust Ltd,* 1**,** 6-7.

EA 2016. Notice of variation and consolidation with introductory note, permit number EPR/PP3830BV/V005. *In:* AGENCY, E. (ed.). United Kingdom: Environment Agency.

EATON, E., CAUDULLO, G., OLIVEIRA, S. & DE RIGO, D. 2016. Quercus robur and Quercus petraea in Europe: distribution, habitat, usage and threats. *In:* SAN-MIGUEL-AYANZ, J. (ed.) *European Atlas of Forest Tree Species.* Luxembourg.

EDWARDS, C. A. & BOHLEN, P. J. 1996. *Biology and Ecology of Earthworms*, Springer Netherlands.

EIJSACKERS, H. 2010. Earthworms as colonisers: Primary colonisation of contaminated land, and sediment and soil waste deposits. *Science of The Total Environment,* 408**,** 1759-1769.

EIJSACKERS, H. 2011. Earthworms as colonizers of natural and cultivated soil environments. *Applied Soil Ecology,* 50**,** 1-13.

EISENHAUER, N. 2010. The action of an animal ecosystem engineer: Identification of the main mechanisms of earthworm impacts on soil microarthropods. *Pedobiologia,* 53**,** 343-352.

EISENHAUER, N. & SCHEU, S. 2008. Earthworms as drivers of the competition between grasses and legumes. *Soil Biology & Biochemistry,* 40**,** 2650-2659.

EISENHAUER, N., SCHUY, M., BUTENSCHOEN, O. & SCHEU, S. 2009a. Direct and indirect effects of endogeic earthworms on plant seeds. *Pedobiologia,* 52**,** 151-162.

EISENHAUER, N., STRAUBE, D., JOHNSON, E. A., PARKINSON, D. & SCHEU, S. 2009b. Exotic Ecosystem Engineers Change the Emergence of Plants from the Seed Bank of a Deciduous Forest. *Ecosystems,* 12**,** 1008-1016.

ERKTAN, A., OR, D. & SCHEU, S. 2020. The physical structure of soil: Determinant and consequence of trophic interactions. *Soil Biology and Biochemistry,* 148**,** 107876.

FAO 2011. *The state of the world's land and water resources for food and agriculture (SOLAW) – Managing systems at risk,* London, Food and Agriculture Organization of the United Nations, Rome and Earthscan.

FARRELL, H. L., LÉGER, A., BREED, M. F. & GORNISH, E. S. 2020. Restoration, Soil Organisms, and Soil Processes: Emerging Approaches. *Restoration Ecology,* n/a.

FC 2010. Thames Beat – Brownfield Project Little Gerpins Species Selection. Forestry Commission: Forestry Commission.

FERLIAN, O., CESARZ, S., MARHAN, S. & SCHEU, S. 2014. Carbon food resources of earthworms of different ecological groups as indicated by 13C compound-specific stable isotope analysis. *Soil Biology and Biochemistry,* 77**,** 22-30.

FERLIAN, O., EISENHAUER, N., AGUIRREBENGOA, M., CAMARA, M., RAMIREZ-ROJAS, I., SANTOS, F., TANALGO, K. & THAKUR, M. P. 2018. Invasive earthworms erode soil biodiversity: A meta-analysis. *Journal of Animal Ecology,* 87**,** 162-172.

FIERER, N., GRANDY, A. S., SIX, J. & PAUL, E. A. 2009. Searching for unifying principles in soil ecology. *Soil Biology and Biochemistry,* 41**,** 2249-2256.

FISHER, B., TURNER, R. K. & MORLING, P. 2009. Defining and classifying ecosystem services for decision making. *Ecological Economics,* 68**,** 643-653.

FOOT, K. & MOFFAT, A. 2008. Woodland Establishment on Landfill Sites - Ten Years of Research. *In:* GOVERNMENT, C. A. L. (ed.). London: Queen's Printer and Controller of Her Majesty's Stationery Office.

FOOT, K. & SINNETT, D. 2014. Best Practice Guidance Note 5 - Imported Soil or Soil-Forming Materials Placement. *In:* GROUP, T. L. R. A. U. G. R. (ed.). Farnham, Surrey: Forest Research.

FOOT, K. J., HISLOP, M. & MCNEILLY, S. 2003. The effect of composted green waste on tree establishment on landfill. *Land Reclamation - Extending Boundaries.* Taylor & Francis.

FORESTRY-COMMISSION 2017. The UK Forestry Standard *In:* COMMISSION, F. (ed.) Fourth Edition ed. Edinburgh: Forestry Commission.

FRAGOSO, C. & LAVELLE, P. 1992. Earthworm communities of tropical rain forests. *Soil Biology and Biochemistry,* 24**,** 1397-1408.

FROUZ, J., ELHOTTOVÁ, D., KURÁŽ, V. & ŠOURKOVÁ, M. 2006. Effects of soil macrofauna on other soil biota and soil formation in reclaimed and unreclaimed post mining sites: Results of a field microcosm experiment. *Applied Soil Ecology,* 33**,** 308-320.

FROUZ, J., KEPLIN, B., PIZL, V., TAJOVSKY, K., LUKES, A., NOVAKOVA, A., BALIK, V., MATERNA, J., DUKER, C., HEINKELE, T., HANELA, L., RUSEK, J. & CHALU, J. 2001. Soil biota and upper soil layer development in two contrasting post-mining chronosequences. *Ecological Engineering* 17**,** 275-284.

FROUZ, J., LIVEČKOVÁ, M., ALBRECHTOVÁ, J., CHROŇÁKOVÁ, A., CAJTHAML, T., PIŽL, V., HÁNĚL, L., STARÝ, J., BALDRIAN, P., LHOTÁKOVÁ, Z., ŠIMÁČKOVÁ, H. & CEPÁKOVÁ, Š. 2013. Is the effect of trees on soil properties mediated by soil fauna? A case study from post-mining sites. *Forest Ecology and Management,* 309**,** 87-95.

FROUZ, J., PIŽL, V., CIENCIALA, E. & KALČÍK, J. 2009. Carbon storage in post-mining forest soil, the role of tree biomass and soil bioturbation. *Biogeochemistry,* 94**,** 111-121.

FROUZ, J. & VINDUŠKOVÁ, O. 2018. Chapter 8 - Soil Organic Matter Accumulation in Postmining Sites: Potential Drivers and Mechanisms. *In:* MUÑOZ, M. Á. & ZORNOZA, R. (eds.) *Soil Management and Climate Change.* Academic Press.

FROUZ, J., VOBOŘILOVÁ, V., JANOUŠOVÁ, I., KADOCHOVÁ, Š. & MATĚJÍČEK, L. 2015. Spontaneous establishment of late successional tree species English oak (Quercus robur) and European beech (Fagus sylvatica) at reclaimed alder plantation and unreclaimed post mining sites. *Ecological Engineering,* 77**,** 1-8.

GANN, G. D., MCDONALD, T., WALDER, B., ARONSON, J., NELSON, C. R., JONSON, J., HALLETT, J. G., EISENBERG, C., GUARIGUATA, M. R., LIU, J., HUA, F., ECHEVERRÍA, C., GONZALES, E., SHAW, N., DECLEER, K. & DIXON, K. W. 2019. International principles and standards for the practice of ecological restoration. Second edition. *Restoration Ecology,* 27**,** S1-S46.

GÓMEZ-BAGGETHUN, E., DE GROOT, R., LOMAS, P. L. & MONTES, C. 2010. The history of ecosystem services in economic theory and practice: From early notions to markets and payment schemes. *Ecological Economics,* 69**,** 1209-1218.

GOODMAN, A. 1998. Practical experience of growing trees on derelict and contaminated sites. *In:* FOX, H. R., MCINTOSH, A. D. & MOORE, H. M. (eds.) *In: Land Reclamation: Achieving Sustainable Benefits.* Taylor & Francis.

GOSS-SAMPSON, M. A. 2018. *Statistical Analysis in JASP: A guide for students*.

GREGORY, A. S., RITZ, K., MCGRATH, S. P., QUINTON, J. N., GOULDING, K. W., JONES, R. J., HARRIS, J. A., BOL, R., WALLACE, P., PILGRIM, E. S. & WHITMORE, A. P. 2015. A review of the impacts of degradation threats on soil properties in the UK. *Soil Use Manag,* 31**,** 1-15.

GRIGAL, D. & VANCE, E. 2000. Influence of soil organic matter on forest productivity. *New Zealand Journal of Forestry Science,* 30**,** 169-205.

GROOT, R., BLIGNAUT, J., VAN DER PLOEG, S., ARONSON, J., ELMQVIST, T. & FARLEY, J. 2013. *Benefits of Investing in Ecosystem Restoration*.

GROSSMAN, J. J., CAVENDER-BARES, J. & HOBBIE, S. E. 2020. Functional diversity of leaf litter mixtures slows decomposition of labile but not recalcitrant carbon over two years. *Ecological Monographs,* 90**,** e01407.

HAIGH, M. J. 2000. *Reclaimed land : erosion control, soils and ecology : essays towards improving the geoecological self-sustainability of lands reconstructed after surface coal mining,* Rotterdam, Rotterdam : A.A. Balkema.

HAINES-YOUNG, R. & POTSCHIN, M. 2010. The links between biodiversity, ecosystem services and human well-being. *Ecosystem Ecology: a new synthesis,* 1**,** 110-139.

HANDLEY, J. F., GRIFFITHS, E. J., HILL, S. L. & HOWE, J. M. 1998. Land restoration using an ecologically informed and participative approach. *In:* FOX, H. R., MCINTOSH, A. D. & MOORE, H. M. (eds.) *In: Land Reclamation: Achieving Sustainable Benefits.* Taylor & Francis.

HARRIS, J. A., PALMER, J. & BIRCH, P. 1996. *Land Restoration and Reclamation: Principles and Practice*, Pearson.

HARRIS, P., BREARLEY, A. & DOICK, K. 2014. Best Practice Guidance Note 17 - Lowland Neutral Grassland - Creation and management in land regeneration. *In:* GROUP, T. L. R. A. U. G. R. (ed.). Farnham, Surrey: Forest Research.

HÄTTENSCHWILER, S. & GASSER, P. 2005. Soil animals alter plant litter diversity effects on decomposition. *Proceedings of the National Academy of Sciences of the United States of America,* 102**,** 1519-1524.

HÄTTENSCHWILER, S., TIUNOV, A. V. & SCHEU, S. 2005. Biodiversity and Litter Decomposition in Terrestrial Ecosystems. *Annual Review of Ecology, Evolution, and Systematics,* 36**,** 191-218.

HAYGARTH, P. M. & RITZ, K. 2009. The future of soils and land use in the UK: Soil systems for the provision of land-based ecosystem services. *Land Use Policy,* 26**,** S187-S197.

HAYNES, R. J., BELYAEVA, O. N. & ZHOU, Y. F. 2015. Particle size fractionation as a method for characterizing the nutrient content of municipal green waste used for composting. *Waste Manag,* 35**,** 48-54.

HEAVEN, F. W. & RICHARDSON, G. R. 2007. Soil Investigation at Ingrebourne Valley, Phase IIA, Rainham. Derby: Land Research Associates.

HEIN, S., COLLET, C., AMMER, C., GOFF, N. L., SKOVSGAARD, J. P. & SAVILL, P. 2009. A review of growth and stand dynamics of Acer pseudoplatanus L. in Europe: implications for silviculture. *Forestry,* 82**,** 361-385.

HENEGHAN, L., MILLER, S. P., BAER, S., CALLAHAM, M. A., MONTGOMERY, J., PAVAO-ZUCKERMAN, M., RHOADES, C. C. & RICHARDSON, S. 2008. Integrating Soil Ecological Knowledge into Restoration Management. *Restoration Ecology,* 16**,** 608-617.

HINES, J. 2019. Ecosystem Functioning: How Much System Is Needed to Explain Function? *Current Biology,* 29**,** R1072-R1074.

HISLOP, A. M. & HARDING, K. 1999. Research into the use of Composted Green Waste for Tree Establishment on Landfill Sites. *In:* HISLOP, A. M. (ed.) *Second Annual Contract Report.* Enventure.

HM DEPT FOR BUSINESS, E. I. S. 2016. Historical coal data: coal production, availability and consumption 1853 to 2015. *In:* HM DEPARTMENT FOR BUSINESS, E. I. S. (ed.).

HOOGSTEEN, M. J. J., LANTINGA, E. A., BAKKER, E. J., GROOT, J. C. J. & TITTONELL, P. A. 2015. Estimating soil organic carbon through loss on ignition: effects of ignition conditions and structural water loss. *European Journal of Soil Science,* 66**,** 320-328.

HOUSTON DURRANT, T., DE RIGO, D. & CAUDULLO, G. 2016. Pinus sylvestris in Europe: distribution, habitat, usage and threats. *In:* SAN-MIGUEL-AYANZ, J. (ed.) *European Atlas of Forest Tree Species.* Luxembourg.

HUANG, W., GONZÁLEZ, G. & ZOU, X. 2020. Earthworm abundance and functional group diversity regulate plant litter decay and soil organic carbon level: A global meta-analysis. *Applied Soil Ecology,* 150**,** 103473.

HUTCHINGS, T. R., SINNETT, D., PEACE, A. J. & MOFFAT, A. J. 2006. The effect of woodland growth on a containment landfill site in Hertfordshire, UK. *Urban Forestry & Urban Greening,* 5**,** 169-176.

HYNYNEN, J., NIEMISTÖ, P., VIHERÄ-AARNIO, A., BRUNNER, A., HEIN, S. & VELLING, P. 2009. Silviculture of birch (Betula pendula Roth and Betula pubescens Ehrh.) in northern Europe. *Forestry: An International Journal of Forest Research,* 83**,** 103-119.

JACKSON, R. B., LAJTHA, K., CROW, S. E., HUGELIUS, G., KRAMER, M. G. & PIÑEIRO, G. 2017. The Ecology of Soil Carbon: Pools, Vulnerabilities, and Biotic and Abiotic Controls. *Annual Review of Ecology, Evolution, and Systematics,* 48**,** 419-445.

JACOB, M., VIEDENZ, K., POLLE, A. & THOMAS, F. M. 2010. Leaf litter decomposition in temperate deciduous forest stands with a decreasing fraction of beech (Fagus sylvatica). *Oecologia,* 164**,** 1083-94.

JÄRVIS, J., IVASK, M., NEI, L., KUU, A. & LUUD, A. 2016. Effect of green waste compost application on afforestation success. *Baltic Forestry,* 22**,** 90-97.

JOLLIFFE, I. T. & CADIMA, J. 2016. Principal component analysis: a review and recent developments. *Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences,* 374**,** 20150202.

JONES, A., PANAGOS, P., BARCELO, S., BOURAOUI, F., BOSCO, C., DEWITTE, O., GARDI, C., ERHARD, M., HERVÁS, J., HIEDERER, R., JEFFERY, S., LÜKEWILLE, A., MARMO, L., MONTANARELLA, L., OLAZÁBAL, C., PETERSEN, J., PENIZEK, V., STRASSBURGER, T., TÓTH, G., VAN DEN EECKHAUT, M., VAN LIEDEKERKE, M., VERHEIJEN, F., VIESTOVA, E. & YIGINI, Y. 2012. The State of Soil in Europe: A contribution of the JRC to the European Environment Agency's Environment State and Outlook Report. *In:* AGENCY, E. E. (ed.).

JONES, C. G., LAWTON, J. H. & SHACHAK, M. 1994. Organisms as Ecosystem Engineers. *Oikos,* 69**,** 373-386.

JONES, H. P., JONES, P. C., BARBIER, E. B., BLACKBURN, R. C., REY BENAYAS, J. M., HOLL, K. D., MCCRACKIN, M., MELI, P., MONTOYA, D. & MATEOS, D. M. 2018. Restoration and repair of Earth's damaged ecosystems. *Proc Biol Sci,* 285.

JÓNSSON, J. Ö. G. & DAVÍÐSDÓTTIR, B. 2016. Classification and valuation of soil ecosystem services. *Agricultural Systems,* 145**,** 24-38.

JOUQUET, P., DAUBER, J., LAGERLÖF, J., LAVELLE, P. & LEPAGE, M. 2006. Soil invertebrates as ecosystem engineers: Intended and accidental effects on soil and feedback loops. *Applied Soil Ecology,* 32**,** 153-164.

JOUQUET, P., HUCHET, G., BOTTINELLI, N., THU, T. D. & DUC, T. T. 2012. Does the influence of earthworms on water infiltration, nitrogen leaching and soil respiration depend on the initial soil bulk density? A mesocosm experiment with the endogeic species Metaphire posthuma. *Biology and Fertility of Soils,* 48**,** 561-567.

JÓZEFOWSKA, A., PIETRZYKOWSKI, M., WOŚ, B., CAJTHAML, T. & FROUZ, J. 2017. The effects of tree species and substrate on carbon sequestration and chemical and biological properties in reforested post-mining soils. *Geoderma,* 292**,** 9-16.

JÓZEFOWSKA, A., WOŚ, B. & PIETRZYKOWSKI, M. 2016. Tree species and soil substrate effects on soil biota during early soil forming stages at afforested mine sites. *Applied Soil Ecology,* 102**,** 70-79.

KARDOL, P. & WARDLE, D. A. 2010. How understanding aboveground–belowground linkages can assist restoration ecology. *Trends in Ecology & Evolution,* 25**,** 670-679.

KENDLE, T. 1996. Natural versus artificial methods of woodland establishment. *In:* MOFFAT, A. (ed.) *Recycling Land for Forestry - Technical Paper 22 - Proceedings of the Forestry Commission and British Land Reclamation Society Conference.* Wrecclesham: Forestry Commission.

KIBBLEWHITE, M. G., RITZ, K. & SWIFT, M. J. 2008. Soil health in agricultural systems. *Philosophical Transactions of the Royal Society B: Biological Sciences,* 363**,** 685-701.

KILBRIDE, C. 2014. Best Practice Guidance Note 6 - Application of Sewage Sludges and Composts. *In:* RESEARCH, F. (ed.). Surrey: Forest Research.

LA NOTTE, A., D'AMATO, D., MAKINEN, H., PARACCHINI, M. L., LIQUETE, C., EGOH, B., GENELETTI, D. & CROSSMAN, N. D. 2017. Ecosystem services classification: A systems ecology perspective of the cascade framework. *Ecol Indic,* 74**,** 392-402.

LANG, B. & RUSSELL, D. J. 2020. Effects of earthworms on bulk density: A meta-analysis. *European Journal of Soil Science,* 71**,** 80-83.

LARNEY, F. J. & ANGERS, D. A. 2012. The role of organic amendments in soil reclamation: A review. *Canadian Journal of Soil Science,* 92**,** 19-38.

LAVELLE, P. 1988. Earthworm activities and the soil system. *Biology and Fertility of Soils,* 6**,** 237-251.

LAVELLE, P., DECAËNS, T., AUBERT, M., BAROT, S., BLOUIN, M., BUREAU, F., MARGERIE, P., MORA, P. & ROSSI, J. P. 2006. Soil invertebrates and ecosystem services. *European Journal of Soil Biology,* 42**,** S3-S15.

LAVELLE, P. & SPAIN, A. 2001. *Soil ecology*, Springer Science & Business Media.

LEHMANN, J. & KLEBER, M. 2015. The contentious nature of soil organic matter. *Nature,* 528**,** 60-68.

LIANG, C., SCHIMEL, J. P. & JASTROW, J. D. 2017. The importance of anabolism in microbial control over soil carbon storage. *Nature Microbiology,* 2**,** 17105.

LIU, T., CHEN, X., GONG, X., LUBBERS, I. M., JIANG, Y., FENG, W., LI, X., WHALEN, J. K., BONKOWSKI, M., GRIFFITHS, B. S., HU, F. & LIU, M. 2019. Earthworms Coordinate Soil Biota to Improve Multiple Ecosystem Functions. *Current Biology,* 29**,** 3420-3429.e5.

LORENZ, M., HOFMANN, D., STEFFEN, B., FISCHER, K. & THIELE-BRUHN, S. 2021. The molecular composition of extractable soil microbial compounds and their contribution to soil organic matter vary with soil depth and tree species. *Science of The Total Environment,* 781**,** 146732.

LOWE, C. & BUTT, K. 2002. Influence of organic matter on earthworm production and behaviour: a laboratory-based approach with applications for soil restoration. *European Journal of Soil Biology,* 38**,** 173−176.

LOWE, C. N., BUTT, K. R. & ORMEROD, T. 2008. Development of inoculated and naturally colonised earthworm populations on landfill cap ameliorated with composted green waste. *Communications in Waste and Resource Management,* 9**,** pp. 87-92.

LUBBERS, I. M., PULLEMAN, M. M. & VAN GROENIGEN, J. W. 2017. Can earthworms simultaneously enhance decomposition and stabilization of plant residue carbon? *Soil Biology and Biochemistry,* 105**,** 12-24.

LUBBERS, I. M., VAN GROENIGEN, K. J., FONTE, S. J., SIX, J., BRUSSAARD, L. & VAN GROENIGEN, J. W. 2013. Greenhouse-gas emissions from soils increased by earthworms. *Nature Climate Change,* 3**,** 187-194.

LUTZOW, M. V., KOGEL-KNABNER, I., EKSCHMITT, K., MATZNER, E., GUGGENBERGER, G., MARSCHNER, B. & FLESSA, H. 2006. Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions - a review. *European Journal of Soil Science,* 57**,** 426-445.

MACKAY, J. M. & HESKETH, F. B. 1998. The Mersey Forest and Red Rose Forest Landfill Woodlands Project. *In:* FOX, H. R., MCINTOSH, A. D. & MOORE, H. M. (eds.) *In: Land Reclamation: Achieving Sustainable Benefits.* Taylor & Francis.

MAES, J., LIQUETE, C., TELLER, A., ERHARD, M., PARACCHINI, M. L., BARREDO, J. I., GRIZZETTI, B., CARDOSO, A., SOMMA, F., PETERSEN, J.-E., MEINER, A., GELABERT, E. R., ZAL, N., KRISTENSEN, P., BASTRUP-BIRK, A., BIALA, K., PIRODDI, C., EGOH, B., DEGEORGES, P., FIORINA, C., SANTOS-MARTÍN, F., NARUŠEVIČIUS, V., VERBOVEN, J., PEREIRA, H. M.,

BENGTSSON, J., GOCHEVA, K., MARTA-PEDROSO, C., SNÄLL, T., ESTREGUIL, C., SAN-MIGUEL-AYANZ, J., PÉREZ-SOBA, M., GRÊT-REGAMEY, A., LILLEBØ, A. I., MALAK, D. A., CONDÉ, S., MOEN, J., CZÚCZ, B., DRAKOU, E. G., ZULIAN, G. & LAVALLE, C. 2016. An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. *Ecosystem Services,* 17**,** 14-23.

MARASHI, A. R. A. & SCULLION, J. 2003. Earthworm casts form stable aggregates in physically degraded soils. *Biology and Fertility of Soils,* 37**,** 375-380.

MCDONALD, T., GANN, G., JONSON, J. & DIXON, K. 2016. International standards for the practice of ecological restoration – including principles and key concepts. Washington D.C.: Society for Ecological Restoration.

MCKAY, H. 2011. Short Rotation Forestry: review of growth and environmental impacts. *Forest Research Monograph, Forest Research,* 2**,** 212.

MEA 2005. *Millennium Ecosystem Assessment - Ecosystems and Human Well-being: Synthesis Report,* Washington, DC., Island Press.

MEDINA-SAUZA, R. M., ÁLVAREZ-JIMÉNEZ, M., DELHAL, A., REVERCHON, F., BLOUIN, M., GUERRERO-ANALCO, J. A., CERDÁN, C. R., GUEVARA, R., VILLAIN, L. & BAROIS, I. 2019. Earthworms Building Up Soil Microbiota, a Review. *Frontiers in Environmental Science,* 7.

MET-OFFICE. 2011a. *Met Office UK Climate Averages - Data comparison of Hull and Cleethorpes weather stations [https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/gcxcb0epz]* [Online]. Available: https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/gcxcb0epz [Accessed 10/02/2021 2021].

MET-OFFICE. 2011b. *Met Office UK Climate Averages - Glasgow Springburn weather station [https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/gcuvzvbnt]* [Online]. Available: https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/u10m1cc30 [Accessed 10/02/2021 2021].

MET-OFFICE. 2011c. *Met Office UK Climate Averages - Stanford Le Hope weather station [https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/u10m1cc30]* [Online]. Available: https://www.metoffice.gov.uk/research/climate/maps-and-data/uk-climate-averages/u10m1cc30 [Accessed 10/02/2021 2021].

MEYSMAN, F. J. R., MIDDELBURG, J. J. & HEIP, C. H. R. 2006. Bioturbation: a fresh look at Darwin's last idea. *Trends in Ecology & Evolution,* 21**,** 688-695.

MOFFAT, A. 1996a. Recycling Land for Forestry. *In:* COMMISSION, F. (ed.) *Technical Paper 22 - Proceedings of the Forestry Commission and British Land Reclamation Society Conference.* Wrecclesham: Forestry Commission.

MOFFAT, A. 1996b. Site Preparation - Getting it right. *In:* MOFFAT, A. (ed.) *Recycling Land for Forestry - Technical Paper 22 - Proceedings of the Forestry Commission and British Land Reclamation Society Conference.* Wrecclesham: Forestry Commission.

MOFFAT, A. 2006. Use of Sewage Sludges and Composts in Forestry. *In:* RESEARCH, F. (ed.). Edinburgh: Forestry Commission.

MOFFAT, A., FOOT, K., KENNEDY, F., DOBSON, M. & MORGAN, G. 2008. Experimental Tree Planting on U.K. Containment Landfill Sites: Results of 10 Years' Monitoring. *Arboriculture & Urban Forestry,* 34**,** 163-172.

MOFFAT, A. J. 2001. Increasing woodland in urban areas in the UK – meeting ecological and environmental standards. *The International Forestry Review,* 3**,** 198-205.

MOFFAT, A. J. & LAING, J. 2003. AN AUDIT OF WOODLAND PERFORMANCE ON RECLAIMED LAND IN ENGLAND. *Arboricultural Journal,* 27**,** 11-25.

MOFFAT, A. J. & MCNEILL, J. 1994. *Reclaiming disturbed land for forestry*, HM Stationery Office.

MONTECCHIO, L., SCATTOLIN, L., SQUARTINI, A. & BUTT, K. R. 2015. Potential spread of forest soil-borne fungi through earthworm consumption and casting. *iForest - Biogeosciences and Forestry,* 8**,** 295-301.

MORIN, X., FAHSE, L., SCHERER-LORENZEN, M. & BUGMANN, H. 2011. Tree species richness promotes productivity in temperate forests through strong complementarity between species. *Ecol Lett,* 14**,** 1211-9.

MORRIËN, E., HANNULA, S. E., SNOEK, L. B., HELMSING, N. R., ZWEERS, H., DE HOLLANDER, M., SOTO, R. L., BOUFFAUD, M.-L., BUÉE, M., DIMMERS, W., DUYTS, H., GEISEN, S., GIRLANDA, M., GRIFFITHS, R. I., JØRGENSEN, H.-B., JENSEN, J., PLASSART, P., REDECKER, D., SCHMELZ, R. M., SCHMIDT, O., THOMSON, B. C., TISSERANT, E., UROZ, S., WINDING, A., BAILEY, M. J., BONKOWSKI, M., FABER, J. H., MARTIN, F., LEMANCEAU, P., DE BOER, W., VAN VEEN, J. A. & VAN DER PUTTEN, W. H. 2017. Soil networks become more connected and take up more carbon as nature restoration progresses. *Nature Communications,* 8**,** 14349.

NE 2014. Earthworms in England: distribution, abundance and habitats. *Natural England,* Report no. NECR145.

NFM 2017. Northern Forest Manifesto - A New Northern Forest. *In:* CITY OF TREES, C. C., HEYWOODS,, THE MERSEY FOREST, W. R. F. & TRUST, A. T. W. (eds.).

NLLP 2003. North Lincolnshire Local Plan - Chapter 11 - The Ironstone Gullets. *In:* COUNCIL, N. L. (ed.).

NOËL, J.-F. & O'CONNOR, M. 1998. Strong sustainability and critical natural capital. *Valuation for Sustainable Development: Methods and Policy Indicators. Edward Elgar, Cheltenham, pp. 75Á/97.*

NOLAN, P. 2017. Why a new Northern forest is worth the investment. *Town and Country Planning***,** 407-414.

OLDFIELD, E. E., FELSON, A. J., AUYEUNG, D. S. N., CROWTHER, T. W., SONTI, N. F., HARADA, Y., MAYNARD, D. S., SOKOL, N. W., ASHTON, M. S., WARREN II, R. J., HALLETT, R. A. & BRADFORD, M. A. 2015. Growing the urban forest: tree performance in response to biotic and abiotic land management. *Restoration Ecology,* 23**,** 707-718.

OLIVER, T. H., ISAAC, N. J. B., AUGUST, T. A., WOODCOCK, B. A., ROY, D. B. & BULLOCK, J. M. 2015. Declining resilience of ecosystem functions under biodiversity loss. *Nature Communications,* 6**,** 10122.

OS-MAPS. 1956. *TA 21 SW - A/*, Surveyed / Revised: Pre-1930 to 1956, Published: 1956*. Surrey: Ordnance Survey.

OS-MAPS. 1966. *TA 21 SW - A, Surveyed / Revised: 1944 to 1970, Published: 1966*, 1:10,000. Surrey: Ordnance Survey.

PALMER, M. W. & DAVIES, R. D. 2014. The effects of organic amendment and soil emplacement method on manufactured soil quality and growth of amenity woodland at a former cokeworks site. *Ecological Engineering,* 68**,** 1-7.

PAOLETTI, M. G. 1999. The role of earthworms for assessment of sustainability and as bioindicators. *Invertebrate Biodiversity as Bioindicators of Sustainable Landscapes.* Elsevier.

PASTA, S., DE RIGO, D. & CAUDULLO, G. 2016. European Atlas of Forest Tree Species - Acer pseudoplatanus in Europe: distribution, habitat, usage and threats. *In:* SAN-MIGUEL- & AYANZ, J. (eds.) *European Atlas of Forest Tree Species.* Luxembourg: Official EU.

PAUL, E. A. 2016. The nature and dynamics of soil organic matter: Plant inputs, microbial transformations, and organic matter stabilization. *Soil Biology and Biochemistry,* 98**,** 109-126.

PERRY, D. & HANDLEY, J. 2000. The Potential for Woodland on Urban and Industrial Wasteland in England and Wales. *In:* COMMISSION, F. (ed.). Edinburgh.

PEY, B., CORTET, J., WATTEAU, F., CHEYNIER, K. & SCHWARTZ, C. 2013. Structure of earthworm burrows related to organic matter of a constructed Technosol. *Geoderma,* 202-203**,** 103-111.

PHILLIPS, H. R. P., GUERRA, C. A., BARTZ, M. L. C., BRIONES, M. J. I., BROWN, G., CROWTHER, T. W., FERLIAN, O., GONGALSKY, K. B., VAN DEN HOOGEN, J., KREBS, J., ORGIAZZI, A., ROUTH, D., SCHWARZ, B., BACH, E. M., BENNETT, J. M., BROSE, U., DECAËNS, T., KÖNIG-RIES, B., LOREAU, M., MATHIEU, J., MULDER, C., VAN DER PUTTEN, W. H., RAMIREZ, K. S., RILLIG, M. C., RUSSELL, D., RUTGERS, M., THAKUR, M. P., DE VRIES, F. T., WALL, D. H., WARDLE, D. A., ARAI, M., AYUKE, F. O., BAKER, G. H., BEAUSÉJOUR, R., BEDANO, J. C., BIRKHOFER, K., BLANCHART, E., BLOSSEY, B., BOLGER, T., BRADLEY, R. L., CALLAHAM, M. A., CAPOWIEZ, Y., CAULFIELD, M. E., CHOI, A., CROTTY, F. V., CRUMSEY, J. M., DÁVALOS, A., DIAZ COSIN, D. J., DOMINGUEZ, A., DUHOUR, A. E., VAN EEKEREN, N., EMMERLING, C., FALCO, L. B., FERNÁNDEZ, R., FONTE, S. J., FRAGOSO, C., FRANCO, A. L. C., FUGÈRE, M., FUSILERO, A. T., GHOLAMI, S., GUNDALE, M. J., LÓPEZ, M. G., HACKENBERGER, D. K., HERNÁNDEZ, L. M., HISHI, T., HOLDSWORTH, A. R., HOLMSTRUP, M., HOPFENSPERGER, K. N., LWANGA, E. H., HUHTA, V., HURISSO, T. T., IANNONE, B. V., IORDACHE, M., JOSCHKO, M., KANEKO, N., KANIANSKA, R., KEITH, A. M., KELLY, C. A., KERNECKER, M. L., KLAMINDER, J., KONÉ, A. W., KOOCH, Y., KUKKONEN, S. T., LALTHANZARA, H., LAMMEL, D. R., LEBEDEV, I. M., LI, Y., JESUS LIDON, J. B., LINCOLN, N. K., LOSS, S. R., MARICHAL, R., MATULA, R., MOOS, J. H., MORENO, G., MORÓN-RÍOS, A., MUYS, B., NEIRYNCK, J., NORGROVE, L., NOVO, M., NUUTINEN, V., et al. 2019. Global distribution of earthworm diversity. *Science,* 366**,** 480-485.

PICHANCOURT, J. B., FIRN, J., CHADES, I. & MARTIN, T. G. 2014. Growing biodiverse carbon-rich forests. *Glob Chang Biol,* 20**,** 382-93.

PIEARCE, T. G., BUDD, T., HAYHOE, J. M., SLEEP, D. & CLASPER, P. J. 2003. Earthworms of a land restoration site treated with paper mill sludge. *Pedobiologia,* 47**,** 792-795.

PIOTTO, D. 2008. A meta-analysis comparing tree growth in monocultures and mixed plantations. *Forest Ecology and Management,* 255**,** 781-786.

PONGE, J.-F., ZANELLA, A., SARTORI, G. & JABIOL, B. 2010. *Terrestrial humus forms: ecological relevance and classification*.

PULLEMAN, M., CREAMER, R., HAMER, U., HELDER, J., PELOSI, C., PÉRÈS, G. & RUTGERS, M. 2012. Soil biodiversity, biological indicators and soil ecosystem services—an overview of European approaches. *Current Opinion in Environmental Sustainability,* 4**,** 529-538.

QUINE, C., CAHALAN, C., HESTER, A., HUMPHREY, J., KIRBY, K., MOFFAT, A. & VALATIN, G. 2011. Woodlands: Chapter 8 - UK National Ecosystem Assessment Technical Report. *In:* DEFRA (ed.).

RAJAPAKSHA, N. S. S., BUTT, K. R., VANGUELOVA, E. I. & MOFFAT, A. J. 2013. Earthworm selection of Short Rotation Forestry leaf litter assessed through preference testing and direct observation. *Soil Biology and Biochemistry,* 67**,** 12-19.

RAWLIK, M., KASPROWICZ, M., JAGODZIŃSKI, A. M., KAŹMIEROWSKI, C., ŁUKOWIAK, R. & GRZEBISZ, W. 2018. Canopy tree species determine herb layer biomass and species composition on a reclaimed mine spoil heap. *Science of The Total Environment,* 635**,** 1205-1214.

RAWLINSON, H., DICKINSON, N., NOLAN, P. & PUTWAIN, P. 2004. Woodland establishment on closed old-style landfill sites in N.W. England. *Forest Ecology and Management,* 202**,** 265-280.

RAYMOND, C. M., SINGH, G. G., BENESSAIAH, K., BERNHARDT, J. R., LEVINE, J., NELSON, H., TURNER, N. J., NORTON, B., TAM, J. & CHAN, K. M. A. 2013. Ecosystem Services and Beyond: Using Multiple Metaphors to Understand Human–Environment Relationships. *BioScience,* 63**,** 536-546.

REICH, P. B. 2014. The world-wide 'fast–slow' plant economics spectrum: a traits manifesto. *Journal of Ecology,* 102**,** 275-301.

REICH, P. B., OLEKSYN, J., MODRZYNSKI, J., MROZINSKI, P., HOBBIE, S. E., EISSENSTAT, D. M., CHOROVER, J., CHADWICK, O. A., HALE, C. M. & TJOELKER, M. G. 2005. Linking litter calcium, earthworms and soil properties: a common garden test with 14 tree species. *Ecology Letters,* 8**,** 811-818.

ROBERTSON, G. P., COLEMAN, D. C., BLEDSOE, C. S. & SOLLINS, P. 1999. *Standard Soil Methods for Long Term Ecological Research,* New York, New York, Oxford University Press.

ROUBÍČKOVÁ, A. & FROUZ, J. 2014. Performance of the earthworm Aporrectodea caliginosa on unreclaimed spoil heaps at different successional stages. *European Journal of Soil Biology,* 65**,** 57-61.

ROWELL, D. L. 1994. *Soil Science: Methods & Applications*, Routledge.

RUTGERS, M., ORGIAZZI, A., GARDI, C., RÖMBKE, J., JÄNSCH, S., KEITH, A. M., NEILSON, R., BOAG, B., SCHMIDT, O., MURCHIE, A. K., BLACKSHAW, R. P., PÉRÈS, G., CLUZEAU, D., GUERNION, M., BRIONES, M. J. I., RODEIRO, J., PIÑEIRO, R., COSÍN, D. J. D., SOUSA, J. P., SUHADOLC, M., KOS, I., KROGH, P.-H., FABER, J. H., MULDER, C., BOGTE, J. J., WIJNEN, H. J. V., SCHOUTEN, A. J. & ZWART, D. D. 2016. Mapping earthworm communities in Europe. *Applied Soil Ecology,* 97**,** 98-111.

SAID-PULLICINO, D., MASSACCESI, L., DIXON, L., BOL, R. & GIGLIOTTI, G. 2010. *Organic Matter Dynamics in a Compost-amended Anthropogenic Landfill Capping-soil*.

SAYER, E. J. 2006. Using experimental manipulation to assess the roles of leaf litter in the functioning of forest ecosystems. *Biological Reviews,* 81**,** 1-31.

SCHELFHOUT, S., MERTENS, J., VERHEYEN, K., VESTERDAL, L., BAETEN, L., MUYS, B. & DE SCHRIJVER, A. 2017. Tree Species Identity Shapes Earthworm Communities. *Forests,* 8**,** 85.

SCHMIDT, M. W., TORN, M. S., ABIVEN, S., DITTMAR, T., GUGGENBERGER, G., JANSSENS, I. A., KLEBER, M., KOGEL-KNABNER, I., LEHMANN, J., MANNING, D. A., NANNIPIERI, P., RASSE, D. P., WEINER, S. & TRUMBORE, S. E. 2011. Persistence of soil organic matter as an ecosystem property. *Nature,* 478**,** 49-56.

SCHOLES, R., MONTANARELLA, L., BRAINICH, A., BARGER, N., TEN BRINK, B., CANTELE, M., ERASMUS, B., FISHER, J., GARDNER, T., HOLLAND, T. G., KOHLER, F., KOTIAHO, J. S., VON MALTITZ, G., NANGENDO, G., PANDIT, R., PARROTTA, J., POTTS, M. D., PRINCE, S., SANKARAN, M. & WILLEMEN, L. 2018. Summary for policymakers of the thematic assessment report on land degradation and restoration. *In:* SERVICES, I.-I. S.-P. P. O. B. A. E. (ed.). Bonn, Germany: IBPES.

SCULLION, J. & MALIK, A. 2000. Earthworm activity affecting organic matter, aggregation and microbial activity in soils restored after opencast mining for coal. *Soil Biology and Biochemistry,* 32**,** 119-126.

SCULLION, J., MOHAMMED, A. R. A. & RICHARDSON, H. 1988. Effect of storage and reinstatement procedures on earthworm populations in soils affected by opencast coal mining. *Journal of Applied Ecology,* 25**,** 233-240.

SEMCHENKO, M., LEFF, J. W., LOZANO, Y. M., SAAR, S., DAVISON, J., WILKINSON, A., JACKSON, B. G., PRITCHARD, W. J., DE LONG, J. R., OAKLEY, S., MASON, K. E., OSTLE, N. J., BAGGS, E. M., JOHNSON, D., FIERER, N. & BARDGETT, R. D. 2018. Fungal diversity regulates plant-soil feedbacks in temperate grassland. *Science Advances,* 4**,** eaau4578.

SERCU, B. K., HERTZOG, L. R., DE GROOTE, S. R. E., BAETEN, L., LENS, L., MARTEL, A., BONTE, D. & VERHEYEN, K. 2019. Soil heterogeneity in tree mixtures depends on spatial clustering of tree species. *Basic and Applied Ecology,* 39**,** 38-47.

SHLENS, J. 2014. A tutorial on principal component analysis. *arXiv preprint arXiv:1404.1100*.

SIMPSON, T. G. 1998. The role of the mineral planning system in England in achieving the reclamation of mineral workings. *In:* FOX, H. R., MCINTOSH, A. D. & MOORE, H. M. (eds.) *In: Land Reclamation: Achieving Sustainable Benefits.* Taylor & Francis.

SIMS, R. W. & GERARD, B. M. 1999. *Earthworms: Keys and Notes for the Identification and Study of the Species,* Shrewsbury, Field Studies Council.

SMALL, N., MUNDAY, M. & DURANCE, I. 2017. The challenge of valuing ecosystem services that have no material benefits. *Global Environmental Change,* 44**,** 57-67.

SÖCHTIG, W. & LARINK, O. 1992. Effect of soil compaction on activity and biomass of endogeic lumbricids in arable soils. *Soil Biology and Biochemistry,* 24**,** 1595-1599.

SOMERVILLE, P. D., MAY, P. B. & LIVESLEY, S. J. 2018. Effects of deep tillage and municipal green waste compost amendments on soil properties and tree growth in compacted urban soils. *Journal of Environmental Management,* 227**,** 365-374.

SPARKE, S., PUTWAIN, P. & JONES, J. 2011. The development of soil physical properties and vegetation establishment on brownfield sites using manufactured soils. *Ecological Engineering,* 37**,** 1700-1708.

STANTURF, J. A., PALIK, B. J. & DUMROESE, R. K. 2014. Contemporary forest restoration: A review emphasizing function. *Forest Ecology and Management,* 331**,** 292-323.

STEIDINGER, B. S., CROWTHER, T. W., LIANG, J., VAN NULAND, M. E., WERNER, G. D. A., REICH, P. B., NABUURS, G., DE-MIGUEL, S., ZHOU, M., PICARD, N., HERAULT, B., ZHAO, X., ZHANG, C., ROUTH, D., PEAY, K. G., ABEGG, M., ADOU YAO, C. Y., ALBERTI, G., ALMEYDA ZAMBRANO, A., ALVAREZ-DAVILA, E., ALVAREZ-LOAYZA, P., ALVES, L. F., AMMER, C., ANTÓN-FERNÁNDEZ, C., ARAUJO-MURAKAMI, A., ARROYO, L., AVITABILE, V., AYMARD, G., BAKER, T., BAŁAZY, R., BANKI, O., BARROSO, J., BASTIAN, M., BASTIN, J.-F., BIRIGAZZI, L., BIRNBAUM, P., BITARIHO, R., BOECKX, P., BONGERS, F., BOURIAUD, O., BRANCALION, PEDRO H. S., BRANDL, S., BREARLEY, F. Q., BRIENEN, R., BROADBENT, E., BRUELHEIDE, H., BUSSOTTI, F., CAZZOLLA GATTI, R., CESAR, R., CESLJAR, G., CHAZDON, R., CHEN, H. Y. H., CHISHOLM, C., CIENCIALA, E., CLARK, C. J., CLARK, D., COLLETTA, G., CONDIT, R., COOMES, D., CORNEJO VALVERDE, F., CORRAL-RIVAS, J. J., CRIM, P., CUMMING, J., DAYANANDAN, S., DE GASPER, A. L., DECUYPER, M., DERROIRE, G., DEVRIES, B., DJORDJEVIC, I., IÊDA, A., DOURDAIN, A., OBIANG, N. L. E., ENQUIST, B., EYRE, T., FANDOHAN, A. B., FAYLE, T. M., FELDPAUSCH, T. R., FINÉR, L., FISCHER, M., FLETCHER, C., FRIDMAN, J., FRIZZERA, L., GAMARRA, J. G. P., GIANELLE, D., GLICK, H. B., HARRIS, D., HECTOR, A., HEMP, A., HENGEVELD, G., HERBOHN, J., HEROLD, M., HILLERS, A., HONORIO CORONADO, E. N., HUBER, M., HUI, C., CHO, H., IBANEZ, T., JUNG, I., IMAI, N., JAGODZINSKI, A. M., et al. 2019. Climatic controls of decomposition drive the global biogeography of forest-tree symbioses. *Nature,* 569**,** 404-408.

STOCKMANN, U., ADAMS, M. A., CRAWFORD, J. W., FIELD, D. J., HENAKAARCHCHI, N., JENKINS, M., MINASNY, B., MCBRATNEY, A. B., COURCELLES, V. D. R. D., SINGH, K., WHEELER, I., ABBOTT, L., ANGERS, D. A., BALDOCK, J., BIRD, M., BROOKES, P. C., CHENU, C., JASTROW, J. D., LAL, R., LEHMANN, J., O'DONNELL, A. G., PARTON, W. J., WHITEHEAD, D. & ZIMMERMANN, M. 2013. The knowns, known unknowns and unknowns of sequestration of soil organic carbon. *Agriculture, Ecosystems & Environment,* 164**,** 80-99.

STROMBERGER, M. E., KEITH, A. M. & SCHMIDT, O. 2012. Distinct microbial and faunal communities and translocated carbon in Lumbricus terrestris drilospheres. *Soil Biology and Biochemistry,* 46**,** 155-162.

SUTTON, P. C., ANDERSON, S. J., COSTANZA, R. & KUBISZEWSKI, I. 2016. The ecological economics of land degradation: Impacts on ecosystem service values. *Ecological Economics,* 129**,** 182-192.

SYMES, D. 1987. *Humberside in the Eighties : A Spatial View of the Economy,* Hull, Hull University Department of Geography.

THAMES-CHASE-TRUST 2014. The Thames Chase Trust Partnership - Annual Report. Upminster, Essex: Thames.

TOÏGO, M., VALLET, P., PEROT, T., BONTEMPS, J.-D., PIEDALLU, C., COURBAUD, B. & CANHAM, C. 2015. Overyielding in mixed forests decreases with site productivity. *Journal of Ecology,* 103**,** 502-512.

TOMLINSON, P. 1984. Evaluating the success of land reclamation schemes. *Landscape Planning,* 11**,** 187-203.

UVAROV, A. V. 2009. Inter- and intraspecific interactions in lumbricid earthworms: Their role for earthworm performance and ecosystem functioning. *Pedobiologia,* 53**,** 1-27.

VAN DER PLAS, F., MANNING, P., ALLAN, E., SCHERER-LORENZEN, M., VERHEYEN, K., WIRTH, C., ZAVALA, M. A., HECTOR, A., AMPOORTER, E., BAETEN, L., BARBARO, L., BAUHUS, J.,

BENAVIDES, R., BENNETER, A., BERTHOLD, F., BONAL, D., BOURIAUD, O., BRUELHEIDE, H., BUSSOTTI, F., CARNOL, M., CASTAGNEYROL, B., CHARBONNIER, Y., COOMES, D., COPPI, A., BASTIAS, C. C., MUHIE DAWUD, S., DE WANDELER, H., DOMISCH, T., FINÉR, L., GESSLER, A., GRANIER, A., GROSSIORD, C., GUYOT, V., HÄTTENSCHWILER, S., JACTEL, H., JAROSZEWICZ, B., JOLY, F.-X., JUCKER, T., KORICHEVA, J., MILLIGAN, H., MÜLLER, S., MUYS, B., NGUYEN, D., POLLASTRINI, M., RAULUND-RASMUSSEN, K., SELVI, F., STENLID, J., VALLADARES, F., VESTERDAL, L., ZIELÍNSKI, D. & FISCHER, M. 2016. Jack-of-all-trades effects drive biodiversity–ecosystem multifunctionality relationships in European forests. *Nature Communications,* 7**,** 11109.

VAN GROENIGEN, J. W., VAN GROENIGEN, K. J., KOOPMANS, G. F., STOKKERMANS, L., VOS, H. M. J. & LUBBERS, I. M. 2019. How fertile are earthworm casts? A meta-analysis. *Geoderma,* 338**,** 525-535.

VERHEYEN, K., VANHELLEMONT, M., AUGE, H., BAETEN, L., BARALOTO, C., BARSOUM, N., BILODEAU-GAUTHIER, S., BRUELHEIDE, H., CASTAGNEYROL, B., GODBOLD, D., HAASE, J., HECTOR, A., JACTEL, H., KORICHEVA, J., LOREAU, M., MEREU, S., MESSIER, C., MUYS, B., NOLET, P., PAQUETTE, A., PARKER, J., PERRING, M., PONETTE, Q., POTVIN, C., REICH, P., SMITH, A., WEIH, M. & SCHERER-LORENZEN, M. 2016. Contributions of a global network of tree diversity experiments to sustainable forest plantations. *Ambio,* 45**,** 29-41.

VESTERDAL, L., CLARKE, N., SIGURDSSON, B. D. & GUNDERSEN, P. 2013. Do tree species influence soil carbon stocks in temperate and boreal forests? *Forest Ecology and Management,* 309**,** 4-18.

VESTERDAL, L., SCHMIDT, I. K., CALLESEN, I., NILSSON, L. O. & GUNDERSEN, P. 2008. Carbon and nitrogen in forest floor and mineral soil under six common European tree species. *Forest Ecology and Management,* 255**,** 35-48.

VIDAL, A., QUENEA, K., ALEXIS, M., NGUYEN TU, T. T., MATHIEU, J., VAURY, V. & DERENNE, S. 2017. Fate of 13 C labelled root and shoot residues in soil and anecic earthworm casts: A mesocosm experiment. *Geoderma,* 285**,** 9-18.

VIDAL, A., WATTEAU, F., REMUSAT, L., MUELLER, C. W., NGUYEN TU, T.-T., BUEGGER, F., DERENNE, S. & QUENEA, K. 2019. Earthworm Cast Formation and Development: A Shift From Plant Litter to Mineral Associated Organic Matter. *Frontiers in Environmental Science,* 7.

VINDUŠKOVÁ, O. & FROUZ, J. 2013. Soil carbon accumulation after open-cast coal and oil shale mining in Northern Hemisphere: a quantitative review. *Environmental Earth Sciences,* 69**,** 1685-1698.

WAGG, C., BENDER, S. F., WIDMER, F. & VAN DER HEIJDEN, M. G. 2014. Soil biodiversity and soil community composition determine ecosystem multifunctionality. *Proc Natl Acad Sci U S A,* 111**,** 5266-70.

WALL, M. 2014. Greenoakhill landfill reclamation monitoring report - Forest Research Technical Development. *In:* RESEARCH, F. (ed.). Edinburgh: Forestry Commission.

WALL, M. 2016. Forestry Trials of Compost & AD in Central Scotland - Forest Research Technical Development. *In:* RESEARCH, F. (ed.). Edinburgh: Forestry Commission.

WARD, N. 1996. The place of trees in the reclamation of disturbed land. *In:* MOFFAT, A. (ed.) *Recycling Land for Forestry - Technical Paper 22 - Proceedings of the Forestry Commission and British Land Reclamation Society Conference.* Wrecclesham: Forestry Commission.

WARDLE, D., BARDGETT, R., KLIRONOMOS, J., SETALA, H., VAN DER PUTTEN, W. & WALL, D. 2004. Ecological Linkages Between Aboveground and Belowground Biota. *Science,* 304**,** 1629-1633.

WEBBER, J., GIBBS, J. & HENDRY, S. 2004. *Phytophthora disease of alder*, Forestry Commission.

WEIL, R. & BRADY, N. 2016. *The Nature and Properties of Soils. 15th edition*.

WIESMEIER, M., URBANSKI, L., HOBLEY, E., LANG, B., VON LÜTZOW, M., MARIN-SPIOTTA, E., VAN WESEMAEL, B., RABOT, E., LIEß, M., GARCIA-FRANCO, N., WOLLSCHLÄGER, U., VOGEL, H.-J. & KÖGEL-KNABNER, I. 2019. Soil organic carbon storage as a key function of soils - A review of drivers and indicators at various scales. *Geoderma,* 333**,** 149-162.

WIKI-WASTE. 2022a. *Immingham Landfill* [Online]. Available: https://bit.ly/3Mr51i6 [Accessed 2022].

WIKI-WASTE. 2022b. *Winterton South Landfill* [Online]. Available: https://bit.ly/3Vhjcu8 [Accessed 2022].

WONG, M. H. & BRADSHAW, A. D. 2002. *The restoration and management of derelict land modern approaches,* New Jersey, World Scientific Pub.

WOŚ, B., PIETRZYKOWSKI, M. & JÓZEFOWSKA, A. 2018. Reclaimed mine soil substrates and tree stands vs. successional forest floor vegetation: A case study of developing ecosystems on afforested mine sites. *Ecological Engineering,* 120**,** 504-512.

WRAP 2009. Guidance on the use of BSI PAS 100 compost in topsoil manufacturing - Technical Document. *In:* WRAP (ed.). UK.

WRAP 2010. Good practice guide for the use of PAS 100 in landscape regeneration. *In:* PROGRAMME, W. R. A. (ed.). WRAP.

WRAP 2011a. Post-industrial landscapes bolster energy crops with the help of PAS 100 Compost. *In:* PROGRAMME, W. R. A. (ed.). Banbury: WRAP.

WRAP 2011b. Soil manufacture using PAS100 compost. *In:* CYMRU, W. (ed.). Cardiff: WRAP.

WRAP 2012. Good Practice Guide: Using PAS 100 compost in landscape and regeneration projects. *In:* PROGRAMME, W. R. A. (ed.) *Guidance Document: Second Edition.* Oxon: URS Infrastructure & Environment UK Limited.

ZANELLA, A., JABIOL, B., PONGE, J.-F., SARTORI, G., WAAL, R., DELFT, B., GRAEFE, U., COOLS, N., KATZENSTEINER, K., HAGER, H., ENGLISCH, M. & BRÊTHES, A. 2009. Toward a European Humus Forms Reference Base. *Studi Trentini di Scienze Naturali,* 85**,** 145-151.

ZANELLA, A., JABIOL, B., PONGE, J. F., SARTORI, G., DE WAAL, R., VAN DELFT, B., GRAEFE, U., COOLS, N., KATZENSTEINER, K., HAGER, H. & ENGLISCH, M. 2011. A European morpho-functional classification of humus forms. *Geoderma,* 164**,** 138-145.

ZHANG, W., HENDRIX, P. F., DAME, L. E., BURKE, R. A., WU, J., NEHER, D. A., LI, J., SHAO, Y. & FU, S. 2013. Earthworms facilitate carbon sequestration through unequal amplification of carbon stabilization compared with mineralization. *Nat Commun,* 4**,** 2576.