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**Soil health- and functionality and air pollution mitigation in urban
permaculture systems**

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Abstract

Urbanisation rapidly transforms natural ecosystems into predominantly surface sealed environments, reducing their provision of vital ecosystem services such as flood mitigation, climate regulation, erosion control and pollution reduction. In this context, urban green spaces such as parks, lawns, and gardens play an important role in maintaining ecosystem functions. Within these spaces, soil health is a key determinant of service provision, with degraded soils exhibiting reduced efficiency. The aim of this study was to evaluate how permaculture-based agroforestry systems influence soil health, soil functionality, and the mitigation of anthropogenic air pollution compared to conventional green space management. Research was conducted in two small-scale permaculture agroforests and two conventional lawns on the campus of the Faculty of Sciences of the University of Lisbon, representing a gradient of structural complexity: a simple lawn, a lawn with trees, and agroforests established in 2016 and 2021, respectively. Soil samples were analysed for key health indicators including soil organic matter (SOM), soil organic carbon (SOC) and bulk density (BD), while soil functionality was assessed via glyphosate degradation potential. Lichen transplants were used as bioindicators for air pollution, measuring photosynthetic performance, heavy metal accumulation and magnetic susceptibility. We hypothesised that (1) soil health indicators improve with structural complexity, (2) soils in more complex systems exhibit greater glyphosate degradation potential and (3) the vertical structure of agroforests reduces air pollution exposure. Results indicate that small-scale urban permaculture agroforests significantly enhance soil health, with higher SOM, reduced BD, and SOC gains of up to 57% relative to conventional management. Glyphosate degradation potential increased with structural complexity, and the vertical vegetation of agroforests acted as an effective barrier to anthropogenic pollutants. These findings highlight the potential of structurally complex green space management to yield relevant ecological benefits both above- and belowground, even when applied at small scales.

Keywords

Permaculture, Urban green spaces, Soil health, Ecosystem services, Air pollution

Resumo

A cobertura do solo urbano está a crescer rapidamente em todo o mundo, transformando ecossistemas naturais em ambientes altamente modificados e predominantemente impermeabilizados. Esta substituição de solos permeáveis, vegetação nativa e habitats complexos por betão, asfalto e paisagens intensivamente geridas reduz significativamente a capacidade dos ecossistemas urbanos para fornecer serviços essenciais, como a mitigação de cheias, a regulação climática, o controlo da erosão, o sequestro de carbono e a filtração de poluentes. Estas alterações impõem desafios significativos à conservação da biodiversidade, à adaptação às alterações climáticas e à manutenção de cidades resilientes e saudáveis.

A incorporação de áreas verdes, incluindo parques urbanos, jardins e relvados, no planeamento urbano emergiu como uma estratégia central para restaurar algumas das funções ecológicas perdidas durante o desenvolvimento urbano. Quando bem concebidas e geridas, estas áreas verdes oferecem não apenas benefícios estéticos e recreativos, mas também desempenham papéis funcionais essenciais nos ecossistemas urbanos. Fornecem habitat para a flora e a fauna, contribuem para a regulação de microclimas locais, melhoram a infiltração e retenção de água no solo e reduzem a poluição atmosférica, beneficiando directamente a saúde humana e o bem-estar da população urbana.

Entre os factores que influenciam a eficácia das áreas verdes urbanas na provisão de serviços ecossistémicos, a saúde do solo destaca-se como um determinante fundamental. Solos saudáveis sustentam funções críticas como a ciclagem de nutrientes, a infiltração e armazenamento de água, a degradação de poluentes e o sequestro de carbono. Em contraste, solos degradados apresentam perda de matéria orgânica, menor actividade microbiana, redução da fertilidade, alterações estruturais, compactação e maior susceptibilidade à poluição. Estas condições limitam a capacidade do solo para fornecer serviços ecossistémicos essenciais, reduzindo a resiliência ambiental e comprometendo o funcionamento urbano sustentável. Melhorar a saúde do solo em áreas urbanas é, por isso, uma prioridade central do planeamento ambiental, contribuindo simultaneamente para a melhoria da qualidade de vida humana.

Os relvados convencionais são o tipo mais comum de área verde urbana, geralmente dominados por uma ou poucas espécies de gramíneas perenes, frequentemente não nativas. Embora possam fornecer algum suporte ao funcionamento ecossistémico, os seus benefícios são controversos. A manutenção dos relvados, especialmente em climas mediterrânicos, exige irrigação e fertilização intensivas. Embora relvados bem mantidos possam apresentar maior sequestro de CO₂ em comparação com relvados não geridos, as práticas de gestão intensiva frequentemente aumentam as emissões de gases com efeito de estufa, o que pode neutralizar os ganhos de carbono e comprometer os benefícios ambientais. Neste contexto, abordagens alternativas de gestão, como a agrofloresta baseada em permacultura, surgem como soluções promissoras, oferecendo potencial para múltiplos benefícios ecológicos em ambientes urbanos.

A agrofloresta de permacultura combina princípios de design ecológico, aumento da biodiversidade e gestão sustentável do solo para criar áreas verdes estruturalmente complexas e multifuncionais. Estes sistemas integram diferentes estratos de vegetação, incluindo árvores de copa, arbustos do sub-bosque e plantas herbáceas, imitando a estrutura vertical das florestas naturais para otimizar a utilização de recursos e promover interacções ecológicas positivas. Em áreas urbanas, pequenas agroflorestas podem ser implementadas em parques, campi universitários e áreas residenciais, criando micro-ecossistemas com

múltiplas camadas de vegetação. A complexidade estrutural destes sistemas é considerada um factor-chave para a melhoria da saúde do solo, uma vez que aumenta a incorporação de matéria orgânica, melhora a ciclagem de nutrientes e promove a diversidade microbiana. Adicionalmente, a vegetação em múltiplos estratos pode influenciar a deposição e retenção de poluentes atmosféricos, reduzindo a exposição humana a contaminantes urbanos, incluindo material particulado, metais pesados e poluentes químicos. Apesar do seu potencial, a agrofloresta urbana permanece pouco estudada e subexplorada em termos de investigação empírica.

O presente estudo avaliou os efeitos de pequenas agroflorestas de permacultura na saúde do solo, na funcionalidade e na mitigação da poluição atmosférica, em comparação com relvados urbanos convencionais. Foram testadas três hipóteses principais: (i) que indicadores de saúde do solo, incluindo matéria orgânica do solo (MOS), carbono orgânico do solo (COS) e densidade aparente do solo (BD), melhoram com o aumento da complexidade estrutural; (ii) que solos em sistemas estruturalmente complexos apresentam maior funcionalidade, medida pela capacidade de degradação do glifosato, um herbicida amplamente utilizado e representativo de poluentes químicos antropogénicos; e (iii) que a estrutura vertical das agroflorestas reduz a exposição à poluição atmosférica em comparação com os relvados.

A investigação foi conduzida no campus da Faculdade de Ciências da Universidade de Lisboa, abrangendo um gradiente de gestão e complexidade estrutural das áreas verdes. Os locais experimentais incluíram dois relvados convencionais (um simples e outro com árvores dispersas) e duas pequenas agroflorestas de permacultura estabelecidas em 2016 e 2021. As agroflorestas foram concebidas com múltiplas camadas de vegetação, incluindo árvores de copa, arbustos do sub-bosque, plantas herbáceas e espécies de cobertura do solo, com o objectivo de maximizar a complexidade estrutural e a funcionalidade ecológica.

A amostragem de solo avaliou indicadores essenciais de saúde, incluindo MOS e COS, que fornecem informação sobre sequestro de carbono, fertilidade e disponibilidade de nutrientes. A densidade aparente do solo (BD) foi medida para avaliar a compactação, a porosidade e as implicações para a infiltração de água e o crescimento radicular. Adicionalmente, foram realizados ensaios de degradação de glifosato, utilizados como proxy da funcionalidade bioquímica do solo, dado que a degradação eficaz de herbicidas depende de comunidades microbianas diversas e metabolicamente activas. Líquenes transplantados foram utilizados como bioindicadores da qualidade do ar, monitorizando o desempenho fotossintético, a acumulação de metais pesados e a suscetibilidade magnética, que reflecte a deposição de partículas antropogénicas.

Os resultados demonstraram que pequenas agroflorestas de permacultura melhoram significativamente a saúde do solo em comparação com os relvados convencionais. Os teores de MOS e COS foram mais elevados nas agroflorestas, reflectindo uma maior incorporação de matéria orgânica e decomposição da serapilheira proveniente da vegetação diversificada. A densidade aparente do solo foi inferior, indicando menor compactação e maior porosidade, factores críticos para o crescimento radicular, a infiltração de água e a actividade microbiana. Estes resultados confirmam que a complexidade estrutural promove a saúde do solo, mesmo em áreas urbanas limitadas, aumentando o potencial de sequestro de carbono e contribuindo para a resiliência dos ecossistemas urbanos.

A funcionalidade do solo, avaliada através do potencial de degradação do glifosato, também aumentou com a complexidade estrutural. Os solos das agroflorestas degradaram o herbicida mais rapidamente do que os

dos relvados convencionais, sugerindo que as comunidades microbianas em sistemas complexos são mais activas e funcionalmente diversas. Estes resultados reforçam a ideia de que práticas de gestão que promovem a diversidade estrutural e biológica fortalecem processos ecossistémicos essenciais, incluindo a mitigação de poluentes químicos.

A estrutura vertical das agroflorestas contribuiu igualmente para a mitigação da poluição atmosférica. Embora a redução do desempenho fotossintético e a acumulação de metais pesados nos líquenes tenham sido semelhantes em todos os locais, a suscetibilidade magnética foi significativamente menor no local de permacultura, indicando uma menor deposição de partículas antropogénicas. A vegetação em múltiplos estratos intercepta provavelmente partículas antes de estas atingirem o sub-bosque ou o solo, funcionando como uma barreira física que reduz a exposição humana a poluentes atmosféricos.

Em síntese, os resultados deste estudo demonstram que pequenas agroflorestas de permacultura podem melhorar substancialmente a saúde do solo, aumentar a funcionalidade microbiana e reduzir a exposição a poluentes atmosféricos, mesmo quando implementadas em áreas limitadas. O estudo destaca a importância de considerar a complexidade estrutural, a diversidade da vegetação e os princípios ecológicos no planeamento e gestão de áreas verdes urbanas. Os relvados convencionais oferecem benefícios estéticos e recreativos, mas apresentam uma capacidade limitada para promover funções ecológicas essenciais, como o sequestro de carbono, a degradação de poluentes e a regulação da qualidade do ar. Em contraste, as agroflorestas, através do seu design em múltiplos estratos e da ênfase na biodiversidade, oferecem uma abordagem multifuncional à gestão urbana de áreas verdes, alinhada com objectivos de sustentabilidade e resiliência. Apesar do potencial evidenciado, estas estratégias permanecem pouco exploradas, sendo necessários estudos adicionais para validar e ampliar os resultados obtidos.

Palavras-chaves

Permacultura, Espaços verdes urbanos, Saúde do solo, Serviços ecossistémicos, Poluição atmosférica

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Acronyms and Symbols

BD	Bulk Density
DTD	Days to Death
FR	FCULresta
GLP	Glyphosate
L	Lawn
LT	Lawn with Trees
MAOC	Mineral-associated Organic Carbon
MAOM	Mineral-associated Organic Matter
OM	Organic Matter
PL	PermaLab
PM	Particulate Matter
POC	Particulate Organic Carbon
POM	Particulate Organic Matter
SOM	Soil Organic Matter
SOC	Soil Organic Carbon
UGS	Urban Green Space
χ	Magnetic susceptibility

1. Introduction

1.1 Urban Green Spaces

1.1.1 Urbanisation

Urban land cover is growing at an enormous rate. While in 1950 only 29.4% of the global population lived in urban settlements, today this number has increased to more than 50% and is projected to reach nearly 68% by 2050 (UN DESA, 2012). This growth is not only demographic but also spatial: cities are expanding in size at a rate roughly twice as fast as their population increase, driven by declining population densities and urban sprawl (Angel et al., 2011; Seto et al., 2012; Seto et al., 2014). Despite the relatively small fraction of total earth surface that is transformed in this way, with current estimates ranging between 0.2% and 2.7% (Schneider et al., 2009), their ecological footprint is disproportionately large. The transformation of natural ecosystems into built environments through often complete removal of aboveground biomass as well as the total sealing of soil, highly alters local ecosystem functioning and decreases the provision of ecosystem services including flood mitigation, climate regulation and carbon storage, which are critical to sustaining quality of life within cities.

As urbanisation accelerates, the challenge is not whether cities will grow, but how they will grow. To shift cities from being drivers of environmental degradation towards becoming contributors to ecological resilience, urban planning and management must integrate ecological considerations. A substantial opportunity for positive change lies in the implementation and use of well-managed urban green spaces (UGS) that maximise ecosystem functioning and may supersede natural ecosystems in providing services in an urban context (Day et al., 2020).

1.1.2 Urban Green Spaces

UGS such as parks, urban forests, community gardens, residential yards and lawns are a crucial component of urban landscapes. They can provide a wide range of ecosystem services that contribute directly and indirectly to the environmental, social, and economic well-being of cities. These services span the four main categories of the ecosystem service framework: provisioning, regulating, cultural and supporting services. Provisioning services include the supply of food, fuel, and other biomass resources in spaces such as community gardens (Vaz Pato et al., 2024). Regulating services are particularly valuable in cities as UGS help mitigate pollution, reduce the urban heat island effect, manage stormwater infiltration, and enhance local climate regulation (Oliveira et al., 2011; Paudel & States, 2023). Supporting services such as soil formation, nutrient cycling and biodiversity conservation facilitate long-term ecological functioning (Norton et al., 2019; Paudel & States, 2023) while cultural services provide opportunities for recreation, education, and aesthetic enjoyment (Pouyat et al., 2020; Vaz Pato et al., 2024).

Beyond their ecological functions, UGS also play a vital role in human health and well-being. Access to UGS has been shown to reduce stress, improve mental health and encourage physical activity (Bedimou-Rung et al., 2005; de Vries et al., 2003).

Considering the loss of ecosystem services through urbanisation, it becomes clear that UGS are not only amenities but essential infrastructures that support the resilience and sustainability of cities in the face of climate change, biodiversity loss and rapid urban expansion.

1.2 Soils in Urban Green Spaces

1.2.1 Soil Health

Within these urban ecosystems, soil health is a major component determining the effectiveness with which ecosystem services are provided. As a fundamental natural resource, soils regulate the majority of ecological processes within terrestrial ecosystems (Pereira et al., 2018). Biodiversity belowground that matches or even exceeds aboveground plant- and animal diversity enables the cycling of nutrients through the decomposition of dead organic matter. At the same time, water is being filtered, stored, and released within the global water cycle (Milly & Dunne, 1994; Wall et al., 2010). Furthermore, soils act as large reservoirs of carbon by sequestering and storing atmospheric CO₂. In fact, more carbon is stored in soils than in vegetation and the atmosphere combined and, when left undisturbed, these carbon pools can remain stable over centuries to millennia (Jobbágy & Jackson, 2000; Lal et al., 2015). Through these mechanisms, soil enables and sustains life on earth (FAO & UNEP, 2021), yet human activities such as urbanisation and agriculture have caused a global decline in soil health, where roughly 33% of the world's soils are now considered moderately to highly degraded (FAO, 2015).

Soil health has been defined as “the capacity of soil to function as a living system, with ecosystem and land use boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health. Healthy soils maintain a diverse community of soil organisms that help to control plant disease, insect and weed pests, form beneficial symbiotic associations with plant roots; recycle essential plant nutrients; improve soil structure with positive repercussions for soil water and nutrient holding capacity, and ultimately improve crop production” (FAO, 2008). Contrary to that, degraded soils are characterised by losses in soil organic matter (SOM) and soil organic carbon (SOC), decreased microbial diversity and fertility as well as changed structural composition, compaction, and pollution (Lal, 2015; León Peláez, 2014). This degradation massively reduces the soil's capacity to provide ecosystem services. For example, soil cultivation and land use change has caused a major fraction of anthropogenic greenhouse gas emissions since the 19th century, and incorrect soil management continues to be a major driver of greenhouse gas emissions today (Amundson et al., 2015; Pereira et al., 2018).

While soil is a non-renewable resource on human time scales, degradation trends can be reversed by conversion to restorative land use and application of sustainable management practices based on minimising erosion, creating positive SOC budgets, increasing activity and diversity of soil biota, and improving structural stability (Lal, 2015; Lehman et al., 2015). This highlights the importance and opportunity that lies in managing soils in UGS with soil health in mind.

1.2.2 Urban Soils

In urban settings, anthropogenic pressures on soils are magnified, as the conversion of natural landscapes to largely artificial environments represent one of the most severe ecosystem alterations imaginable. A defining feature of urban soils is the prevalence of highly physically disturbed or fully sealed surfaces such as roads and construction. This disrupts water and nutrient cycles, alters heat balance, and reduces viable habitat for root growth as well as soil organisms and therefore largely impedes provision of services (Day et al., 2020; Li et al., 2018).

This highlights the importance of UGS, as they provide pockets of uncovered soils that retain the ability to facilitate ecological processes. However, soils in UGS are subject to unique pressures, as they occur at the centre of intense and concentrated human activity. In addition to physical disturbance, they are subject to multiple sources of pollution. Atmospheric deposition of heavy metals from traffic and industry can reduce soil macrofauna densities as well as soil microbial biomass and diversity (Nahmani & Lavelle, 2002; Pan et al., 2020; Panneerselvam et al., 2022; Sazykin et al., 2023). Direct inputs from fertilizers, pesticides, and herbicides in managed green spaces further influence soil chemistry and can have unintended consequences for ecosystem services and human health when contaminants leach into groundwater (FAO & UNEP, 2021; Lu et al., 2015).

Despite these challenges, management practices of UGS have a significant impact on soil health and consequently on how effective they are at providing ecosystem services (Chen et al., 2013; Day et al., 2020).

1.2.3 Management Strategies in UGS

1.2.3.1 Conventional Lawns

Lawns are the most common type of UGS, accounting for up to 75% of all urban green spaces globally (Ignatieva et al., 2015). Within cities lawn cover can reach up to 23% of total urban area (Ignatieva & Hedblom, 2018; Milesi et al., 2005; Robbins & Birkenholtz, 2003). Typically, they are dominated by one or a few, often non-native, perennial grasses and require intensive management including frequent mowing, irrigation, and the use of agrochemicals (Byrne, 2005; Paudel & States, 2023).

Lawns are culturally valued for their aesthetic and recreational benefits (Ignatieva & Hedblom, 2018). Ecologically, they can provide ecosystem services that improve quality of life as well as support ecosystem functioning within cities. Lawns can support heat regulation, erosion control, reduce rainwater runoff and act as carbon sinks (Beard & Green, 1994; Milesi et al., 2005; Townsend-Small & Czimczik, 2010). For example, in urban areas with permeable surfaces like lawns, only 5 - 15% of rainwater becomes runoff, whereas in vegetation free areas that percentage rises as high as 60% (Ignatieva et al., 2015). Flood mitigation and groundwater recharge are essential services, as large portions of urban populations rely on groundwater as a source for clean drinking water. Yet, it must be considered that these benefits are usually seen when comparing lawns to vegetation free areas (Ignatieva & Hedblom, 2018).

The benefits of lawns are context-dependent and often come at a substantial environmental cost. In arid and semi-arid regions, maintaining lawns frequently requires irrigation, where lawns can account for up to 75% of total household water consumption (Milesi et al., 2005). Additionally, lawns are often subjected to heavy

use of fertilisers, with application rates sometimes exceeding cropland systems (Day et al., 2020). These chemicals may leach into, and therefore contaminate, the very groundwater the lawns recharge (FAO & UNEP, 2021). Irrigated and fertilized lawns exhibit enhanced CO₂ sequestration compared to unmanaged lawns, yet these same management practices increase greenhouse gas emissions through nitrous oxide emissions from fertilisers and fossil fuel powered lawn mowers, negating the positive effects of sequestration (Ignatieva et al., 2015; Townsend-Small & Czimeczik, 2010).

Moreover, conventional lawns are structurally simple. They exhibit low plant diversity, underutilise vertical space and provide limited habitat for urban biodiversity. While they perform better than sealed surfaces, their capacity to maximise ecosystem services may be limited compared to more diverse and multifunctional systems. This highlights the need to consider alternative management strategies that enhance ecological functioning while maintaining recreational and aesthetic benefits.

1.2.3.3 Permaculture Based Agroforestry

Permaculture is a holistic design practice that aims to create resilient systems inspired by the processes, structures and patterns observed in nature. The co-founder of the permaculture movement, David Holmgren, defines permaculture as “Consciously designed landscapes which mimic the patterns and relationships found in nature, while yielding an abundance of food, fibre and energy for provision of local needs” (Holmgren, 2002). While permaculture started out as a method for sustainable agriculture, it has evolved to include the design of socio-ecological land-use systems, recognizing that land use systems are never separate from social systems (Krebs & Bach, 2018). Thus, permaculture design operates under three guiding principles: (1) Care for the earth (2) Care for the people and (3) set limits to consumption and reproduction, and redistribute surplus, or fair share (Holmgren, 2002).

On an ecological level, permaculture focuses on regenerating soil, conserving water, reducing waste, and using renewable energy (Mollison, 1988). Permaculture-based agroforests, or food forests, are multifunctional systems designed to mimic the structure and function of natural forests while producing food and other resources. By layering trees, shrubs, vines, herbs, and groundcovers, these systems maximize vertical space and diversify plant functional groups, creating complex three-dimensional habitats (Figure 1.1).

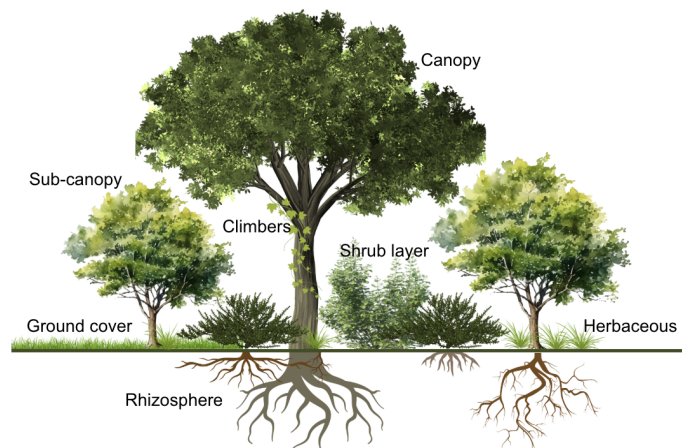


Figure 1.1. The seven layers of a permaculture forest garden.

Much of the ecological rationale for agroforests comes from established evidence in natural forest ecosystems, where continuous organic inputs, structural complexity and high plant diversity contribute to SOC accumulation and microbial diversity (Palandrani & Alberti, 2020). Similarly, rural agroforestry studies have shown that agroforestry systems show significant soil health benefits such as increased SOC and microbial biomass compared to monocultures (Eddy & Yang, 2022).

Translating these mechanisms into urban contexts, agroforests are expected to improve soil health by building organic matter, stabilising soil structure through extensive root networks and supporting diverse microbial communities (Udawatta et al., 2019). They may also enhance water regulation by increasing infiltration, reducing runoff, and conserving soil moisture via canopy cover and ground vegetation (D'Odorico et al., 2007; Udawatta et al., 2002). Nutrient cycling can be strengthened by deep-rooted species capturing minerals from subsoil layers, while decomposing litter and organic inputs enrich topsoil fertility.

Beyond soils, agroforests may support biodiversity by providing habitat and food resources for pollinators, birds, and beneficial insects. Their structural and species diversity enables ecological interactions largely absent in conventional lawns (Torralba et al., 2016). At the same time, they offer provisioning services such as fruits, nuts, and herbs, and cultural services including recreation, education, and opportunities for community engagement (Vaz Pato et al., 2024).

Despite this ecological potential, agroforestry in urban contexts remains underexplored. While principles are grounded in forest ecology, little empirical research has examined how urban agroforests influence soil health, carbon dynamics, or long-term ecosystem service provision. Addressing this gap is essential, as understanding the soil-related impacts of alternative management practices such as agroforestry would provide valuable insights into how urban green spaces can be designed to maximize ecosystem services and sustainability.

1.3 Urban Air Pollution

Urban environments are major sources of anthropogenic air pollutants, including nitrogen oxides, ozone, and particulate matter (PM), primarily emitted by traffic, construction, and industry. Air pollution poses severe environmental and public health risks, contributing to respiratory and cardiovascular diseases and caused an estimated 4.2 million premature deaths worldwide in 2019 (WHO, 2024). Among these pollutants, PM is of particular concern. It consists of a mix of solid particles and liquid droplets suspended in the air and can carry hundreds of different chemicals, including toxic heavy metals (Rai, 2016; EPA, 2025). PM has been classified as carcinogenic by the International Agency for Research on Cancer (Loomis et al., 2014), yet in 2019 it was estimated that ~ 97% of the population within the European region was exposed to PM concentrations that exceeded the WHO air quality guideline level (WHO, 2023). The magnetic fraction of PM stems largely from traffic and industrial emissions, making the magnetic properties of PM useful proxies for anthropogenic air pollution (Winkler et al., 2020).

These pollutants not only affect human health but also settle on terrestrial surfaces and accumulate in urban soils, where elevated heavy metal concentrations are almost universally reported (Pouyat et al., 2010). Heavy metal deposition can alter soil chemistry and degrade microbial communities (Abdu et al., 2017; Sazykin et al., 2023).

Lichens, symbiotic associations between fungi and algae or cyanobacteria, are sensitive bioindicators for air pollution due to their unique biology: they lack roots, cuticles, and stomata, thus absorbing nutrients as well as pollutants directly over their entire surface (Conti & Tudino, 2016; Loppi, 2014). Furthermore, they have been shown to accumulate certain heavy metals at higher concentrations than other higher plants and can retain these over long periods of time (Van Der Wat & Forbes, 2015; Yang et al., 2023). Because they accumulate these substances over time, lichens are ideal for monitoring of long-term exposure levels. Furthermore, lichen physiological response to exposure, specifically photosynthetic activity, is a useful indicator for pollution induced stress and magnetic susceptibility of lichens is used as a proxy for anthropogenic properties in PM. The epiphytic lichen *Evernia prunastri* is one of the most commonly used species for bioindication of air quality, due to its well-known sensitivity to pollution (Contardo et al., 2021; Lackovičová et al., 2013).

By combining soil analyses with lichen-based bioindication across lawns and agroforests, this study explores how different urban green space management practices influence not only belowground soil health but also aboveground ecosystem functions.

1.4 HortaFCUL Project

HortaFCUL is a community-based permaculture project located on the campus of the Faculty of Sciences of the University of Lisbon (FCUL). It was founded in 2009 by a group of students who wanted to promote permaculture practices amongst the university community. Since its inception, HortaFCUL has implemented a multitude of permaculture-based systems throughout the university campus. Among these interventions are the two sites observed in this study: “FCULresta” and “PermaLab”. Both sites are small-scale permaculture-based agroforestry systems, where conventional lawns, much like the ones observed in this study, were transformed into forest-like systems (Figure 1.2). FCULresta has a size of $\sim 300 \text{ m}^2$ and PermaLab a size of $\sim 700 \text{ m}^2$.



Figure 1.2. Development of FCULresta. (a) The site before intervention in 2021 and (b) FCULresta in 2023.

1.5 Objectives

The aim of this study was to examine how different management practices of UGS, specifically conventional lawns and permaculture-based agroforests, affect soil health, soil functionality and air pollution indicators. To assess soil health, soils were sampled across four sites: (1) a lawn, (2) a lawn with trees, (3) a young permaculture agroforest, and (4) an older permaculture agroforest. Key indicators of soil health including soil organic matter, soil organic carbon and bulk density were measured. Soil functionality was evaluated by testing the soils' potential for glyphosate degradation, used here as a proxy for their capacity to buffer and remediate pollution. To assess air pollution, lichens of the species *Evernia prunastri* were used as bioindicators. During a 60-day exposure period, photosynthetic performance was measured as a proxy for vitality. After the exposure period the accumulation of heavy metals and magnetic susceptibility was measured to capture both physiological and chemical responses to urban air pollution.

We hypothesise that (1) soil health indicators improve with structural complexity, (2) soils in more complex systems exhibit greater glyphosate degradation potential and (3) the vertical structure of agroforests reduces air pollution exposure.

2. Materials and Methods

2.1 Study Area

All the sites surveyed in this study were located in close proximity to each other on the campus of the Faculty of Sciences of the University of Lisbon in Lisbon, Portugal. The location of the campus within Lisbon's Campo Grande district, a densely populated urban area near a highway and the city's airport makes it an excellent study area to analyse the impact of different management practices of UGS. The prevalent mild Mediterranean climate, classified on the Köppen-Geiger climate classification as “warm temperate” (Peel et al., 2007), is characterised by mild, rainy winters and hot, dry summers.

2.2 Soils

2.2.1 Site Selection

The study sites were selected to represent contrasting management strategies, namely conventional lawns and permaculture-based agroforests (Figure 2.1). The four chosen sites capture a gradient of structural complexity: a conventionally managed lawn, a managed lawn interspersed with trees and two permaculture systems. Among the multitude of on-campus intervention sites of the HortaFCUL project, the two sites were chosen due to the type of intervention, that is the permaculture-based agroforestry approach, as well as the time that these forests were established. The younger site, “FCULresta”, was established in 2021 and the older site, the “PermaLab” was established in 2016. Previous surveys have documented differences among these sites in terms of relative tree cover and plant diversity, where the permaculture systems exhibited increased values in both measures (Vaz Pato et al., 2024). A schematic visualisation of tree cover in study sites is shown in Figure 2.2.



Figure 2.1. Map of soil sampling sites. All sites were located on the Campus of the Faculty of Sciences of the University of Lisbon. (Source: Google Earth, 2025).

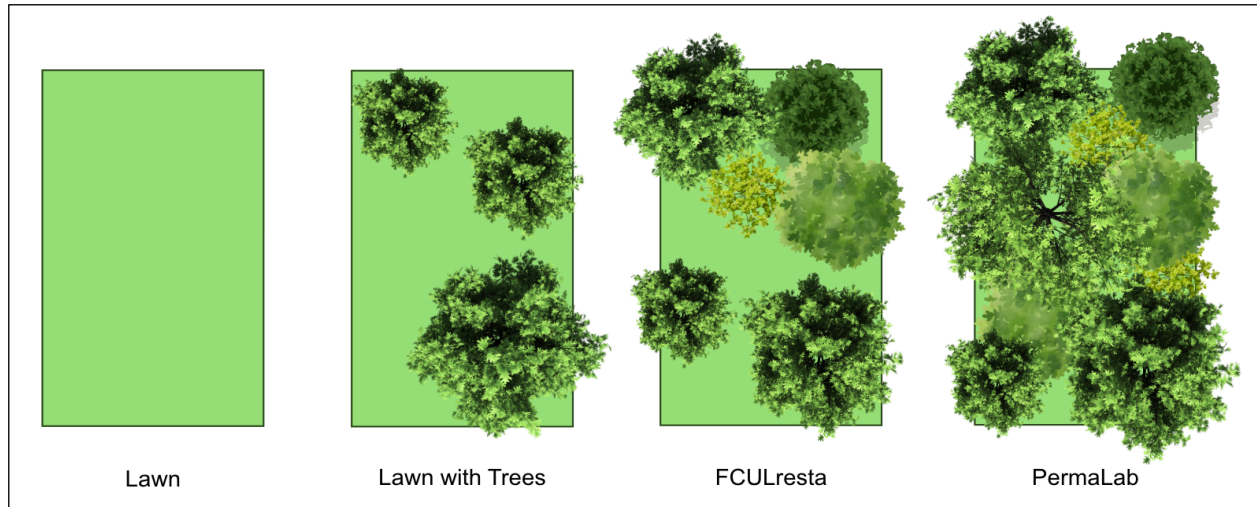


Figure 2.2. Schematic representation of relative tree cover in sites. Figure created by the author using data from Vaz Pato et al., 2024.

2.2.2 Soil Sampling

In each site, five sampling spots were chosen. Due to the relatively small size of the plots, sampling spots were distributed in each corner as well as one in the centre, allowing for sampling of the complete area. Soil samples were taken following - with minor adjustments - the LUCAS guidelines (Fernández-Ugalde et al., 2017). That is, in each sampling spot five subsamples were taken: in the centre and 4 spots at 1 m distance to the north, east, south, and west of the centre spot. The LUCAS guidelines call for 2 m distance from the centre spot but, due to the much smaller size of the sampling sites, this was not always possible, so a distance of 1 m was chosen instead (Figure 2.3, a). The five subsamples created one composite sample for each sampling spot. Furthermore, in each sampling spot composite samples were created at two soil depths: from 0 - 10 cm (hereafter referred to as the surface layer) and from 10 - 20 cm (subsurface layer). This was achieved by using a soil probe and taking out a soil core that spans the whole depth (Figure 2.3, b) and separating the core into the two depth segments. Therefore, in each sampling spot we obtained two samples, in each sampling site 10 samples and in total 40 samples across all four sampling sites and soil depths. All soil samples were taken in late April 2024.

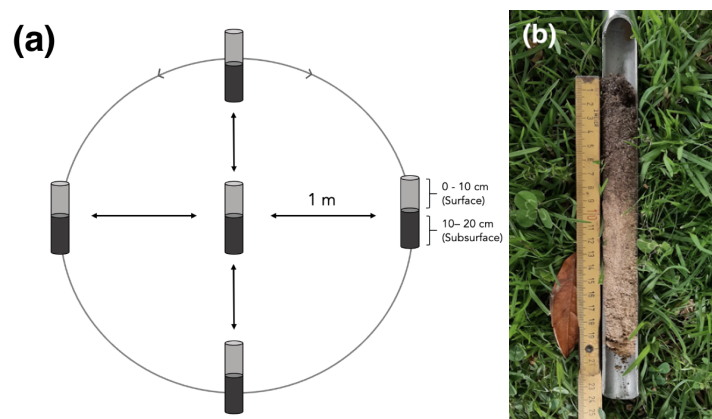


Figure 2.3. Soil sampling. (a) Sampling scheme and (b) example of soil core.

2.2.3 Sample Preparation

After the samples were taken, they were air dried for two weeks until constant weight. Total dry weight of all samples was obtained using a precision scale (AS 220.R2, Radwag, Radom, Poland), then all samples were sieved with 2 mm sieves, separating them into two fractions: > 2 mm and < 2 mm.

2.2.4 Bulk Density

Bulk density (BD) for each sample was calculated as follows:

$$BD (g\ cm^{-3}) = \frac{Dry\ weight\ soil\ (g)}{Total\ soil\ volume\ (cm^3)} \quad (2.1)$$

Soil volume was calculated with the radius of the soil probe (1.25 cm) and the height of the soil sample (10 cm):

$$\begin{aligned} Soil\ volume\ subsample\ (cm^3) &= \pi * 1.25^2\ cm * 10\ cm \\ Soil\ volume\ subsample\ (cm^3) &= 49.087\ cm^3 \end{aligned} \quad (2.2)$$

Each sample was made up of five subsamples, therefore:

$$\begin{aligned} Total\ soil\ volume\ (cm^3) &= 5 * 49.087\ cm^3 \\ Total\ soil\ volume\ (cm^3) &= 245,435\ cm^3 \end{aligned} \quad (2.3)$$

2.2.5 > 2 mm Fraction

Total weight of the > 2 mm fraction was obtained using a precision scale. Furthermore, roots as well as coarse organic matter were separated manually, and each fraction weighed precisely.

Organic matter per unit area ($kg\ m^{-2}$) for roots and coarse organic matter was calculated as follows:

$$OM (kg\ m^{-2}) = \frac{Weight\ fraction\ (g)}{Surface\ soil\ probe\ (cm^2)} * 10 \quad (2.4)$$

Where the surface of the soil probe was calculated:

$$\begin{aligned} Surface\ soil\ probe\ (cm^2) &= \pi * r^2 \\ Surface\ soil\ probe\ (cm^2) &= \pi * 1.25^2 \\ Surface\ soil\ probe\ (cm^2) &= 4.91\ cm^2 \end{aligned} \quad (2.5)$$

2.2.6 < 2 mm Fraction

2.2.6.1 Soil Organic Matter

To estimate soil organic matter (SOM) a loss on ignition method was used (Heiri et al., 2001): for each sample ~ 5 g (4,990 - 5,010 g) of soil was weighed accurately (SWB) with a precision scale, placed into a crucible and burned in a muffle furnace (L3, Nabertherm, Lilienthal, Germany) at 550 °C for 4 h. Afterwards, the samples were weighed again at room temperature (SWA) to obtain weight difference which equates the soil organic matter present:

$$SOM (\%) = \left(\frac{SWB - SWA}{SWB} \right) * 100 \quad (2.6)$$

From this absolute SOM was calculated:

$$SOM (g) = Total\ sample\ weight (g) * \frac{100}{SOM (\%)} \quad (2.7)$$

and SOM per unit area ($kg\ m^{-2}$) was calculated as follows:

$$SOM < 2mm (kg\ m^{-2}) = \frac{SOM (g)}{Surface\ soil\ probe (cm^2)} * 10 \quad (2.8)$$

Total organic matter per $kg\ m^{-2}$ was calculated by adding all obtained organic matter fractions:

$$Total\ OM (kg\ m^{-2}) = SOM < 2mm (kg\ m^{-2}) + Root\ OM (kg\ m^{-2}) + Coarse\ OM (kg\ m^{-2}) \quad (2.9)$$

2.2.6.2 Soil Texture and Organic Matter Stocks

To analyse soil composition ~10 g of each sample was weighed accurately (TW) and sieved using 425 μm and 63 μm sieves in order to separate soil into three fractions: (1) Coarse sand (> 425 μm); (2) Fine sand (> 63 μm) and (3) Silt+clay (<63 μm). To achieve this, the sieves were stacked, the soil placed in the top sieve and shaken manually for 30s in order for the fractions to separate. Then each individual fraction was weighed with a precision scale (WF) and the percentage share calculated:

$$Fraction (\%) = (WF - TW) * 100 \quad (2.10)$$

Furthermore, organic matter content for each soil fraction was analysed using the same loss on ignition method described above. That is, each individual fraction was placed in a crucible and burned in a muffle furnace at 550 °C for 4 h.

Organic matter per unit area ($kg\ m^{-2}$) for each soil fraction was calculated as follows:

$$OM\ soil\ fraction (kg\ m^{-2}) = \frac{OM\ Fraction (g)}{Surface\ soil\ probe (cm^2)} * 10 \quad (2.11)$$

2.2.7 Soil Organic Carbon

Soil organic carbon (SOC) for the < 2 mm fraction of each sample was estimated using the van Bemmelen factor of 0.58 as a conversion factor. The van Bemmelen factor assumes that 58 % of Organic matter is carbon (Pribyl, 2010), therefore SOC per unit area (kg m^{-2}) was calculated as follows:

$$SOC (\text{kg m}^{-2}) = SOM < 2\text{mm} (\text{kg m}^{-2}) * 0.58 \quad (2.12)$$

and SOC concentration was calculated as follows:

$$SOC (\text{g kg}^{-1}) = SOM < 2\text{mm} (\%) * 0.58 * 10 \quad (2.13)$$

To assess how management practices impact carbon sequestration if applied at campus level, all available green areas on the FCUL campus were measured. This was achieved by placing polygons on all available green areas of FCUL using Google Earth (Google Earth, 2025), measuring their areas (m^2), and adding the areas together (Figure 2.4).

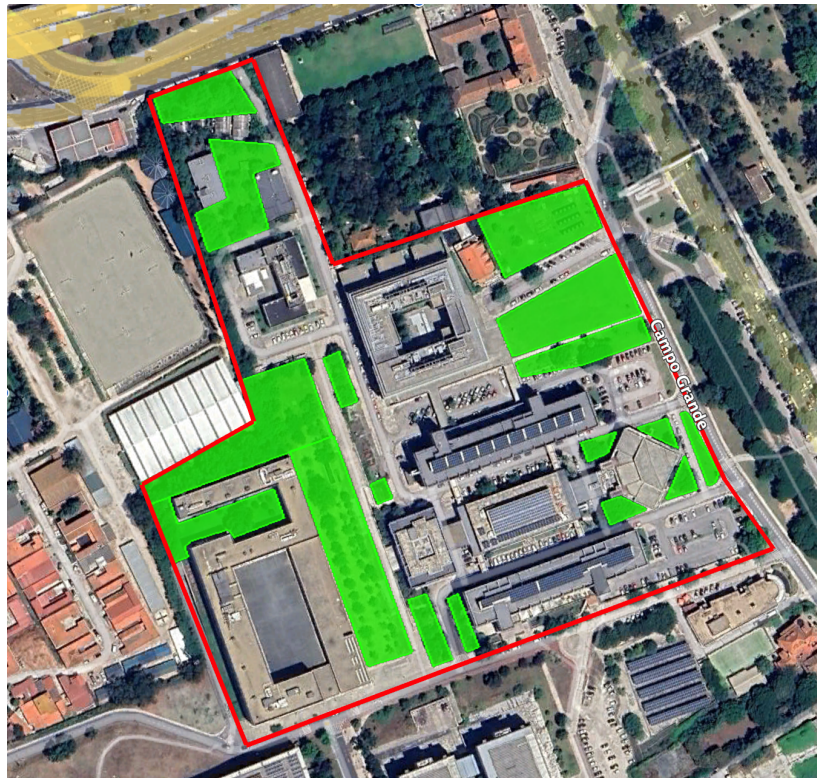


Figure 2.4. Polygons covering all green spaces within the FCUL campus. Campus borders are marked in red. (Source: Google Earth, 2025).

In total, the campus' green spaces cover an area of $16\,864 \text{ m}^2$. Therefore, the potential carbon sequestration at campus scale for each management practice was calculated as:

$$Total\ SOC\ (kg) = SOC\ (\text{kg m}^{-2}) * 16\,864\ \text{m}^2 \quad (2.14)$$

and converted from kg to t as follows:

$$Total\ SOC\ (t) = \frac{Total\ SOC\ (kg)}{1000} \quad (2.15)$$

2.2.8 Soil Capacity for Glyphosate Degradation

First, a 10 mg l⁻¹ glyphosate solution was prepared using distilled water. For the borate buffer, 15.23 g l⁻¹ (0.04 mol) was used. For the soil samples, 100 g of soil from each collected sample was weighed. The samples were placed in containers, and 6 mL of the glyphosate solution was added to each container. Each sample was homogenized using a spatula. All samples were then left to rest in an incubator at 23–25 °C for 40 days. After this period, the samples were derivatized using FMOCl and quantified by UV-Vis spectrophotometry. The quantification of glyphosate was performed using UV-Vis spectrophotometry, based on the derivatization method with 9-fluorenylmethyl chloroformate (FMOCl), as described by Felton et al., 2017, with some modifications. Initially, a calibration curve was constructed using glyphosate standard solutions with concentrations ranging from 0 to 20 ppm, prepared in 0.1 M KCl solution. The standard solutions were subjected to the same derivatization procedure applied to the samples: to 1 ml of each solution, 1 ml of FMOCl solution (2.5 mM in acetonitrile) and 1 ml of borate buffer (pH 9.0) were added. The mixture was kept at room temperature for 30 minutes, followed by extraction of the excess FMOCl with 8 ml of dichloromethane. After phase separation, only the aqueous phase was used for spectrophotometric analysis, performed at 264 nm using quartz cuvettes. For sample analysis, 1 ml of each sample, previously diluted in water, was transferred to Falcon tubes, and the same derivatization procedure used for the standard solutions was applied.

2.3 Lichens

2.3.1 Sampling Sites

The sites chosen for lichen exposure were, with minor adjustments, the same sites where soil sampling took place, but with the focus placed on air pollution exposure: a conventional lawn and permaculture site (PermaLab), located next to the Segunda Circular, a busy motorway, where air pollution pressure was assumed to be increased (+ Exp), and one conventional lawn (Lawn with Trees) and permaculture site (FCULresta) located within the premises of the university campus where traffic is reduced (- Exp) (Figure 2.5).



Figure 2.5. Map of sampling sites. All sites are the same as where soil sampling took place except for the conventional lawn with + Exposure. (Source: Google Earth, 2025).

2.3.2 Lichen Exposure

Thalli of the species *Evernia prunastri* were transplanted and exposed for a duration of 60 days. Exposure took place from July to September 2024. Thalli were collected in a reasonably pristine area. To create the transplants, the ‘lichen bag’ method was used, where lichen thalli were placed in fine grid mesh bags and hung in trees across the sampling sites (Figure 2.6, a). In each sampling spot, a total of five lichen bags were hung. One of the bags contained five thalli, where each thalli equals one replicate. These bags were retrieved for regular fluorescence measurements. The other four bags, containing ~ 5g of lichen, were used for bioaccumulation and magnetic susceptibility analysis, where each bag equals one replicate. As controls for bioaccumulation analysis, four bags containing 5g of lichens were placed in a freezer over the duration of the exposure.

2.3.3 Photosynthetic Activity

Over a period of 60 days, photosynthetic capacity of lichens was measured in 10-day intervals. To achieve this, the sample bags were retrieved, and thalli rehydrated by regularly misting them with water over a period of 4 hours in order to assure photosynthetic activity (Figure 2.6, b). Fluorescence of each thallus was measured using a continuous excitation fluorometer (HandyPEA, Hansatech Instruments Ltd., King’s Lynn, Norfolk, Great Britain). Thalli were dark-adapted for 10 min using a clip and then lightened with a saturating red light for 1 s (650 nm, 3000 $\mu\text{mol photons m}^{-2}\text{s}^{-1}$) with a light-emitting diode (LED).

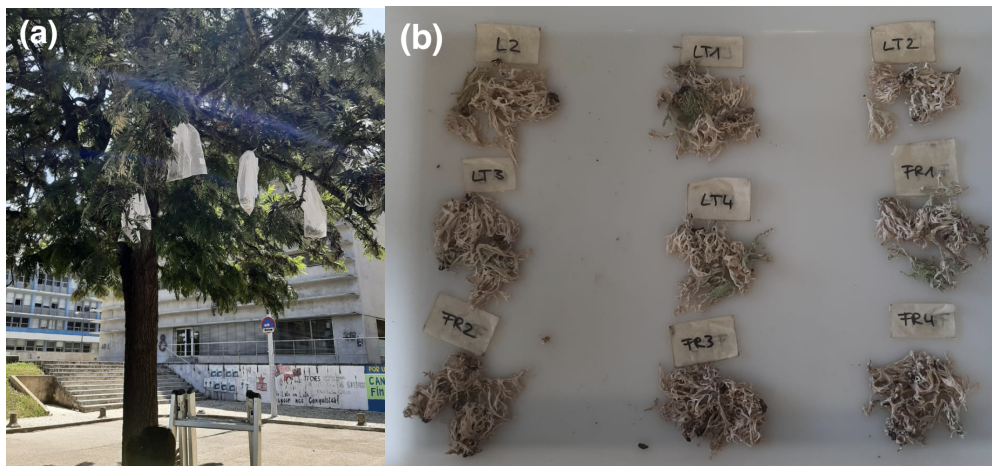


Figure 2.6. Lichen exposure. (a) Lichen transplants hung in a tree. (b) Rehydrated lichen samples after 50 days of exposure.

To assess lichen mortality, a threshold of $F_v/F_m = 0.32$ was established below which lichens were considered dead (Munzi et al., 2019). Linear models were fitted for each sampling point within sites, and Days to Death (DTD) were calculated as follows, where $x = \text{DTD}$:

$$\begin{aligned} 0.32 &= ax + b \\ x &= (0.32 - b)/a \\ x &= \frac{(0.32 - \text{intercept})}{\text{slope}} \end{aligned} \tag{2.16}$$

2.3.4 Metal Accumulation and Magnetic Susceptibility

Magnetic susceptibility and bioaccumulation of metals in exposed and control lichens was analysed in external laboratories.

Rate of accumulation in exposed lichens was assessed using the ratio between the concentration after exposure and the control sample (E/C ratio) and the relative interpretative scale according to Frati et al., 2005.

2.4 Statistical Analysis

All statistical analysis was done using R version 4.5.0 software (R Core Team, 2025) and R studio (Posit, 2025). The following R packages were used to facilitate data processing, visualization, and post-hoc analyses: “dplyr” (v1.1.4; Wickham et al., 2023) and “tidyr” (v1.3.1; Wickham et al., 2024) were used for data manipulation and cleaning, including subsetting, summarizing, and reshaping datasets, “ggplot2” (v3.5.2; Wickham, 2016) was used for data visualization, “car” (v3.1.3; Fox & Weisberg, 2019) was used to perform Levene’s test for assessing homogeneity of variances, “dunn.test” (v1.3.6; Dinno, 2024) was applied to conduct Dunn’s post-hoc test for multiple comparisons following non-parametric Kruskal-Wallis tests and “multcompView” (v0.1.10; Graves et al., 2024) was used to generate compact letter displays to summarize statistically significant differences between groups.

All data was checked for normality with the Shapiro-Wilk test and tested for homogeneity of variances with Levene’s test. When assumptions of normality and homogeneity were met, an analysis of variance was applied, and post hoc comparisons were done using pairwise t-tests with Holm correction. When data met assumptions for normality but were heteroscedastic, comparisons were made using Welch’s t-test. For non-parametric data, Kruskal-Wallis test and Dunn’s test with Holm correction as post-hoc comparisons were used when comparing multiple groups. When comparing two groups, the Wilcoxon rank-sum test was used.

3. Results

3.1 Soils

3.1.1 Bulk Density

A two-way analysis of variance revealed a highly significant effect of site ($F(3) = 13.223, p < 0.005$) and depth ($F(1) = 13.145, p < 0.005$) on soil bulk density (BD), as well as a significant interaction of site * depth ($F(3) = 3.779, p < 0.05$), indicating that the effect of soil depth varied with site.

BD tended to be higher in conventionally managed sites than in permaculture sites, particularly in the surface soil layer (Figure 3.1). Post-hoc comparisons using pairwise t-tests with Holm correction showed that within the surface layer, Lawn with Trees (1.109 ± 0.154) had the highest BD, significantly exceeding both FCULresta ($0.531 \pm 0.137, p < 0.001$) and PermaLab ($0.724 \pm 0.177, p = 0.003$). Lawn (0.89 ± 0.089) had a significantly higher BD than FCULresta ($p = 0.004$), while no difference was detected between Lawn and PermaLab ($p = 0.1$). In the subsurface layer, BD did not differ among sites, although the tendency for higher BD in the lawn sites was maintained, with PermaLab exhibiting the lowest values and Lawn with Trees the highest (Figure 3.1).

Within sites, BD showed a tendency to be higher in subsurface soil layers, where it increased significantly with soil depth in FCULresta ($p = 0.001$). No differences between layers were observed in any other sites.

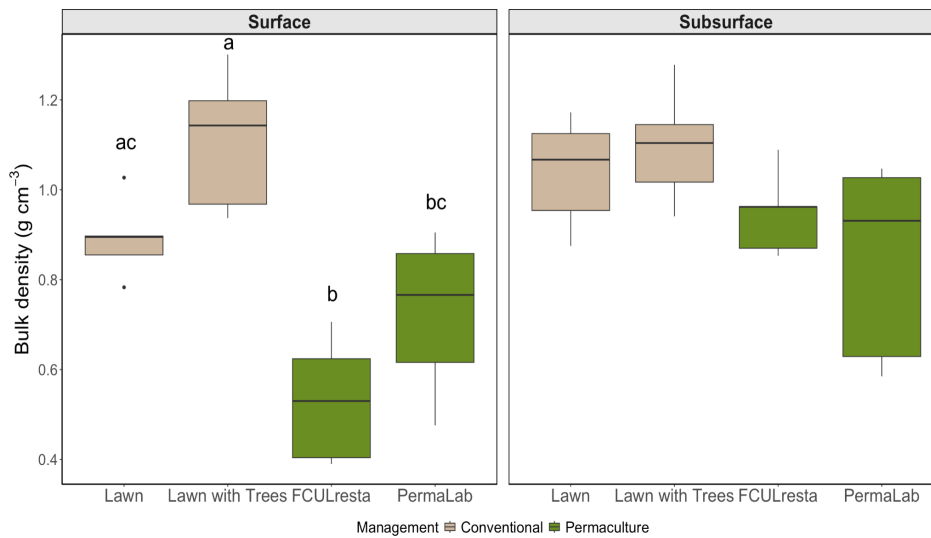


Figure 3.1. Bulk density in g cm^{-3} across sites in surface- and subsurface soil layers. Site differences were assessed using pairwise t-tests with Holm correction ($\alpha = 0.05$). Letters indicate significant differences between sites within each soil layer.

3.1.2 Soil Organic Matter

3.1.2.1 Soil Organic Matter Content < 2mm

Analysis of soil organic matter (SOM) content in sieved soil samples revealed significant site differences, as shown by a Kruskal-Wallis test in both the surface ($\chi^2(3) = 9.971, p = 0.019$), and subsurface soil layer

($\chi^2(3) = 9.988, p = 0.019$). In the surface soil layer, SOM content tended to be higher in permaculture sites (Figure 3.2). Post-hoc comparisons using Dunn’s test with Holm correction showed that Lawn with Trees had significantly lower SOM concentrations than both FCULresta ($p = 0.044$) and PermaLab ($p = 0.033$).

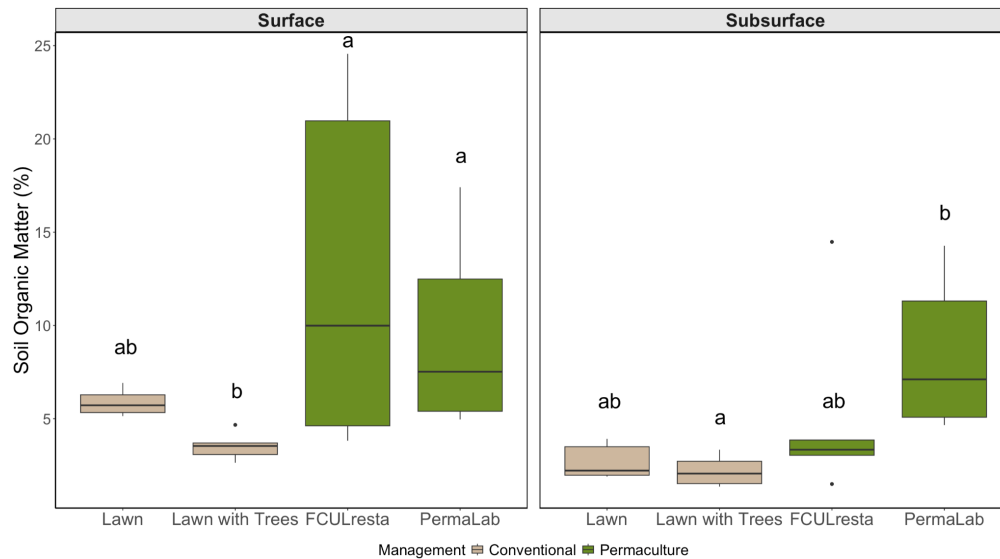


Figure 3.2. Soil organic matter concentration across sites in surface- and subsurface soil layers. Site differences were assessed using pairwise t-tests with Holm correction ($\alpha = 0.05$). Letters indicate significant differences between sites within each soil layer.

In the subsurface soil layer, significant differences were detected only between Lawn with Trees and PermaLab ($p = 0.015$) (Table 3.1).

Table 3.1. Soil Organic matter in sieved samples (<2mm) across sites and soil depths. Values presented depict the mean \pm standard deviation. Letters indicate significant differences between sites within each soil layer resulting from Dunn’s test with Holm correction ($\alpha = 0.5$)

Site	L	LT	FR	PL	L	LT	FR	PL
	Surface				Subsurface			
SOM < 2mm (%)	5.878 ^{ab} ± 0.728	3.526^b ± 0.762	12.792^a ± 9.493	9.556^a ± 5.311	2.702 ^{ab} ± 0.94	2.2^a ± 0.83	5.244 ^{ab} ± 5.237	8.486^b ± 4.17
SOM < 2 mm (kg m ⁻²)	25.311 ± 2.771	18.076 ± 2.794	30.38 ± 19.783	25.934 ± 6.38	13.193 ^{ab} ± 5.675	10.668^a ± 4.199	21.912 ^{ab} ± 19.716	27.568^b ± 6.232

L = Lawn; LT = Lawn with Trees; FR = FCULresta; PL = PermaLab

3.1.2.2 Organic Matter Stock < 2mm

Comparison of SOM stock per unit area showed no significant site differences in the surface layer (Kruskal-Wallis $\chi^2(3) = 4.234, p = 0.237$). In the subsurface soil layer significant differences were found ($\chi^2(3) = 10.131, p = 0.018$). Post-Hoc comparisons using Dunn’s test with Holm correction revealed that Lawn with Trees and PermaLab differed significantly from each other ($p = 0.014$), while all other sites showed no differences.

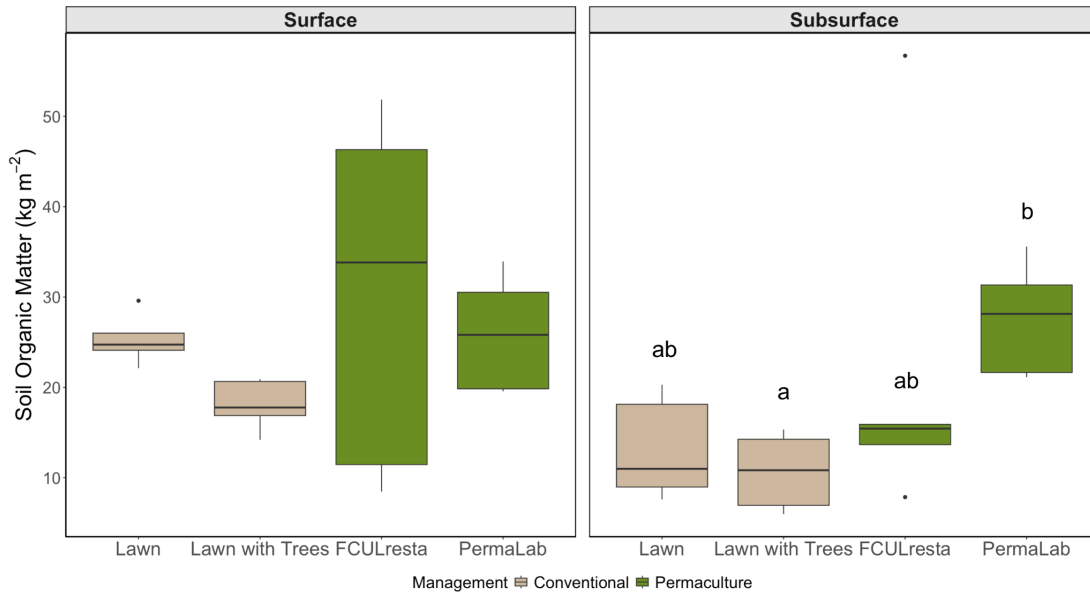


Figure 3.3. Organic matter stock in kg m⁻² across sites in surface- and subsurface soil layers. Site differences were assessed using pairwise t-tests with Holm correction ($\alpha = 0.05$). Letters indicate significant differences between sites within each soil layer.

3.1.3 Soil Texture and Organic Matter Stocks

Evaluation of soil texture revealed that, according to the USDA soil texture triangle, all soils observed in this study could be characterised as sandy (Figure 3.4). That is, all soils had sand contents exceeding 85% (Soil Science Division Staff, 2017).

Despite this, Dunn's tests with Holm correction revealed that the distribution of soil fractions differed significantly among sites within each depth (Table 3.2).

In the surface layer, fine sand ($> 63 \mu\text{m}$) was the dominating fraction making up $> 50\%$ of soil material in all sites except PermaLab, where it was significantly lower compared to Lawn ($p = 0.032$), Lawn with Trees ($p = 0.033$) and FCULresta ($p = 0.04$). In PermaLab coarse sand ($> 425 \mu\text{m}$) was the dominant fraction and significantly larger than in Lawn with Trees ($p = 0.01$). While low across all sites, the silt+clay ($< 63\mu\text{m}$) fraction was greater in Lawn with Trees than in Lawn and PermaLab ($p = 0.002$). Root presence was higher in Lawn with Trees than in FCULresta ($p = 0.045$). Coarse OM did not differ across sites.

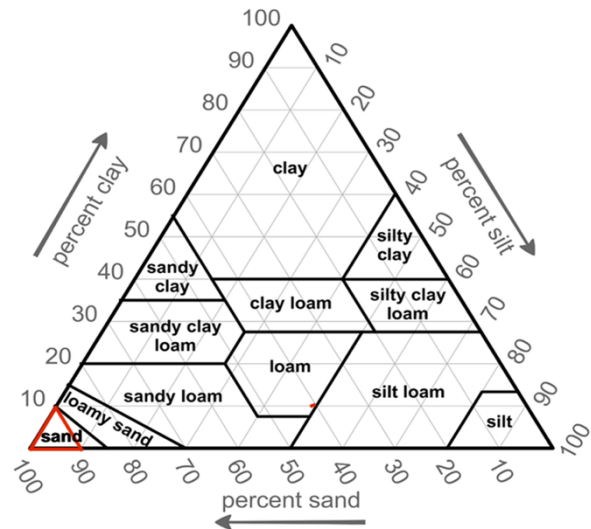


Figure 3.4. USDA soil texture triangle (Soil Science Division Staff, 2017). All observed soils fall within the red lines within the sand category.

In the subsurface layer, fine sand remained dominant in all sites except PermaLab, where it was significantly lower than in FCULresta ($p = 0.0006$). Correspondingly, coarse sand was highest in PermaLab and lowest in FCULresta, differing significantly ($p = 0.0004$). As in the surface layer, silt+clay was significantly higher in Lawn with Trees than in Lawn ($p = 0.014$) and PermaLab ($p = 0.01$). Root presence was higher in PermaLab than in Lawn ($p = 0.029$). Coarse OM was significantly higher in PermaLab than in Lawn ($p = 0.012$).

Table 3.2. Soil fractions of total in % and organic matter stock in kg m^{-2} . Values presented depict the mean \pm standard deviation. Letters indicate significant differences between sites within each soil layer resulting from Dunn's test with Holm correction ($\alpha = 0.5$). Fractions with significant site differences are marked in bold.

Site	L	LT	FR	PL	L	LT	FR	PL	
	Surface				Subsurface				
Soil fraction of total (%)	Roots > 2mm	0.326 ^{ab} ± 0.144	0.668^b \pm 0.458	0.078^a \pm 0.064	0.126 ^{ab} ± 0.263	0.014^a \pm 0.013	0.084 ^{ab} ± 0.07	0.09 ^{ab} ± 0.079	0.216^b \pm 0.263
	Coarse OM > 425 μm	0.148 ± 0.22	0.21 ± 0.24	0.098 ± 0.073	0.168 ± 0.285	0.012^a \pm 0.022	0.028 ^{ab} ± 0.016	0.114 ^{ab} ± 0.1	0.336^b \pm 0.285
	> 63 μm	38.326 ^{ab} ± 5.18	34.994^a \pm 2.14	37.64 ^{ab} ± 9.021	59.226^b \pm 8.122	45.992 ^{ab} ± 9.577	40.49 ^{ab} ± 8.26	27.658^a \pm 2.026	64.04^b \pm 4.964
	< 63 μm	57.048^a \pm 5.688	56.922^a \pm 3	56.902^a \pm 8.187	38.16^b \pm 7.167	51.622 ^{ab} ± 10.099	52.87 ^{ab} ± 9.249	66.878^a \pm 2.189	33.47^b \pm 4.546
Soil fraction OM (kg m^{-2})	Root OM	1.403 ^{ab} ± 0.588	3.516^b \pm 2.68	0.219^a \pm 0.225	0.299^a \pm 0.29	0.069^a \pm 0.061	0.412 ^{ab} ± 0.376	0.403 ^{ab} ± 0.357	0.646^b \pm 0.688
	Coarse OM > 425 μm	0.588 ± 0.847	0.97 ± 1.077	0.234 ± 0.137	0.444 ± 0.232	0.056^a \pm 0.104	0.124 ^{ab} ± 0.057	0.49 ^{ab} ± 0.419	1.132^b \pm 1.041
	> 63 μm	5.726 ± 2.076	6.148 ± 4.929	15.884 ± 11.612	14.678 ± 2.82	7.626 ^{ab} ± 7.236	4.364 ^{ab} ± 2.709	2.122^a \pm 1.671	17.698^b \pm 3.779
	< 63 μm	14.398 ± 1.369	10.952 ± 1.534	17.856 ± 9.742	11.636 ± 3.442	5.651 ± 3.107	5.251 ± 1.1	18.876 ± 27.321	11.0 ± 3.121
	5.184^a \pm 1.192	3.881 ^{ab} ± 0.461	3.094 ^{ab} ± 1.708	1.334^b \pm 0.421	1.583 ^{ab} ± 0.185	1.697 ^{ab} ± 0.441	2.782^a \pm 0.6	1.236^b \pm 0.258	

L = Lawn; LT = Lawn with Trees; FR = FCULresta; PL = PermaLab

Within sites, root content between soil depths differed significantly in both Lawn and Lawn with Trees. Wilcoxon rank sum tests showed that root content was significantly higher in surface layers than in subsurface layers (Lawn: $W = 0$, $p = 0.012$ and Lawn with Trees: $W = 2$, $p = 0.037$). In both permaculture sites, no significant difference between soil layers was found (FCULresta: $W = 14$, $p = 0.823$, and PermaLab: $W = 16$, $p = 0.526$).

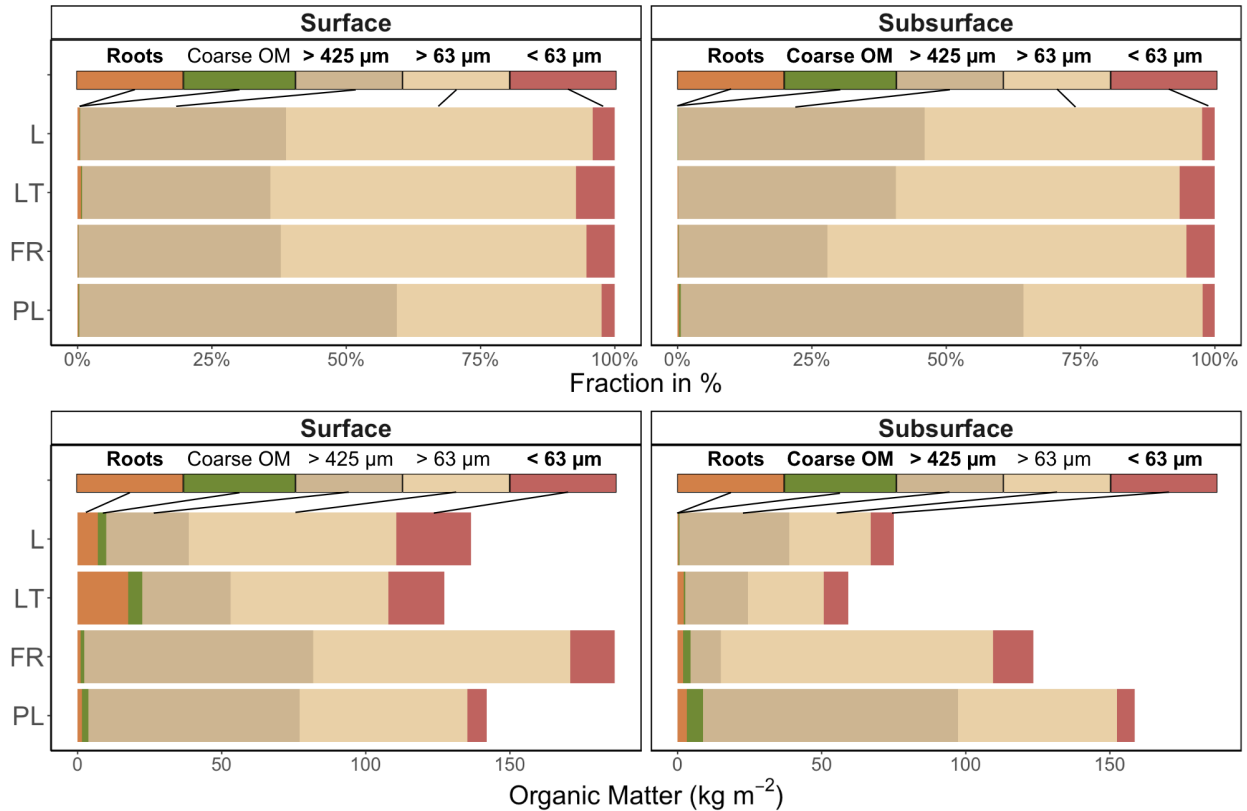


Figure 3.5. Soil fractions of the total in % and organic matter in kg m^{-2} across sites in surface- and subsurface soil layers. Site differences were assessed using Dunn's test with Holm correction ($\alpha = 0.5$). Fractions with significant group differences are marked in bold. L = Lawn; LT = Lawn with Trees; FR = FCULresta; PL = PermaLab.

Kruskal-Wallis test and Dunn's tests with Holm correction as post-hoc comparisons revealed that OM stocks differed among sites in several soil fractions and both soil depths (Table 3.2, Figure 3.5).

In the surface layer, OM stock of coarse and fine sand fractions did not differ between any sites, despite differences in soil fraction distribution. OM stock in the silt+clay fraction was significantly lower in PermaLab than in Lawn ($p = 0.004$). Root OM stock was significantly higher in Lawn with trees than both FCULresta ($p = 0.02$) and PermaLab ($p = 0.032$).

In the subsurface soil layer, variance in OM stocks across sites was greater than in the surface soil layer. PermaLab differed significantly from Lawn in Root OM stock ($p = 0.038$) and Coarse OM stock ($p = 0.01$), with PermaLab showing higher values in both fractions. Coarse sand associated OM was highest in PermaLab and lowest in FCULresta, differing significantly ($p = 0.008$), while fine sand OM stock did not differ across sites. Silt+clay associated OM was highest in FCULresta and lowest in PermaLab, differing significantly ($p = 0.003$).

Organic matter concentrations of fractions differed significantly between sites and across soil layers (Table 3.3). Kruskal-Wallis test and Dunn's tests with Holm correction as post-hoc comparisons revealed that in the surface soil layer, OM concentration in the coarse sand fraction was significantly higher in PermaLab than in Lawn with Trees ($p = 0.04$). In the subsurface layer OM concentration of the coarse sand fraction

was significantly higher in PermaLab than in FCULresta ($p = 0.006$) and OM concentration of the silt+clay fraction was significantly higher in FCULresta than in the Lawn ($p = 0.04$).

Table 3.3. Organic matter concentrations (%) in < 2mm fractions. Values presented depict the mean \pm standard deviation. Letters indicate significant differences between sites within each soil layer resulting from Dunn's test with Holm correction ($\alpha = 0.5$). Fractions with significant site differences are marked in bold.

Site	L	LT	FR	PL	L	LT	FR	PL
	Surface				Subsurface			
> 425 μm	1.328 ^{ab} ± 0.488	1.252^a ± 1.139	6.864 ^{ab} ± 6.071	5.296^b ± 2.555	1.482 ^{ab} ± 1.317	0.882 ^{ab} ± 0.528	0.49^a ± 0.418	5.349^b ± 2.421
> 63 μm	3.345 ± 0.451	2.132 ± 0.409	7.166 ± 4.5	4.278 ± 2.38	1.132 ± 0.6	1.088 ± 0.171	4.488 ± 6.678	3.342 ± 1.907
< 63 μm	1.196 ± 0.214	0.758 ± 0.147	1.186 ± 0.608	0.494 ± 0.32	0.334^b ± 0.056	0.352 ^{ab} ± 0.147	0.628^a ± 0.138	0.372 ^{ab} ± 0.153

L = Lawn; LT = Lawn with Trees; FR = FCULresta; PL = PermaLab

3.1.4 Soil Organic Carbon

Soil organic carbon (SOC) concentrations differed significantly among sites (Kruskal-Wallis $\chi^2(3) = 15.015$, $p = 0.002$)(Figure 3.6). Post-Hoc Dunn test with Holm correction revealed that SOC concentration was significantly lower in Lawn with Trees ($16.60 \pm 5.945 \text{ g kg}^{-1}$) compared to PermaLab ($52.329 \pm 26.311 \text{ g kg}^{-1}$, $p = 0.0009$). SOC in Lawn ($24.879 \pm 10.742 \text{ g kg}^{-1}$) and FCULresta ($52.303 \pm 47.851 \text{ g kg}^{-1}$) did not differ significantly from either site.

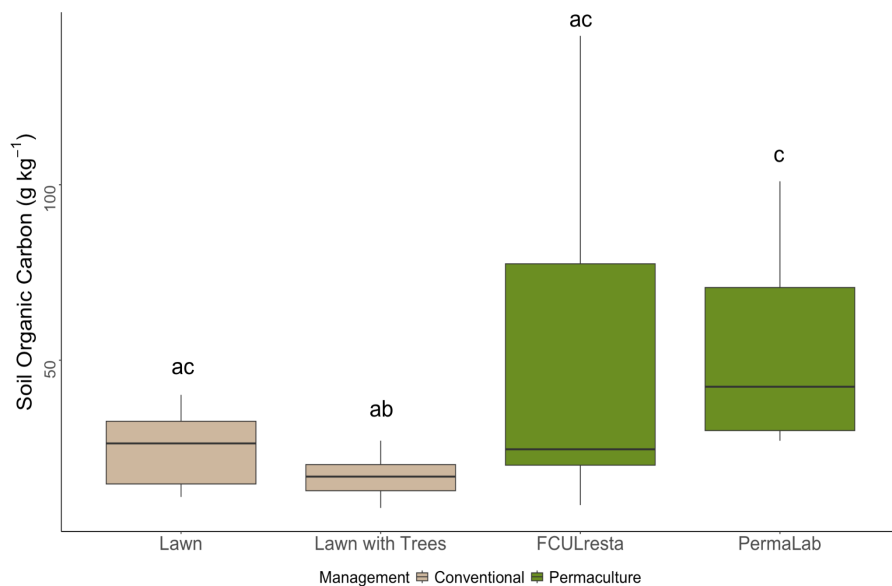


Figure 3.6. Soil organic carbon concentration in g kg^{-1} across sites. Letters indicate significant differences between sites resulting from pairwise t-test with holm correction ($\alpha = 0.05$).

Analysis of SOC stock across sites revealed a trend for higher SOC in permaculture sites (Figure 3.7). This was confirmed by a pairwise t-test with Holm correction, where Lawn with Trees ($16.672 \pm 3.65 \text{ kg m}^{-2}$) differed significantly from FCULresta ($30.33 \pm 11.116 \text{ kg m}^{-2}$, $p = 0.039$) and PermaLab ($31.031 \pm 7.309 \text{ kg m}^{-2}$, $p = 0.034$). SOC stock in Lawn ($22.332 \pm 3.363 \text{ kg m}^{-2}$) did not differ from any site.

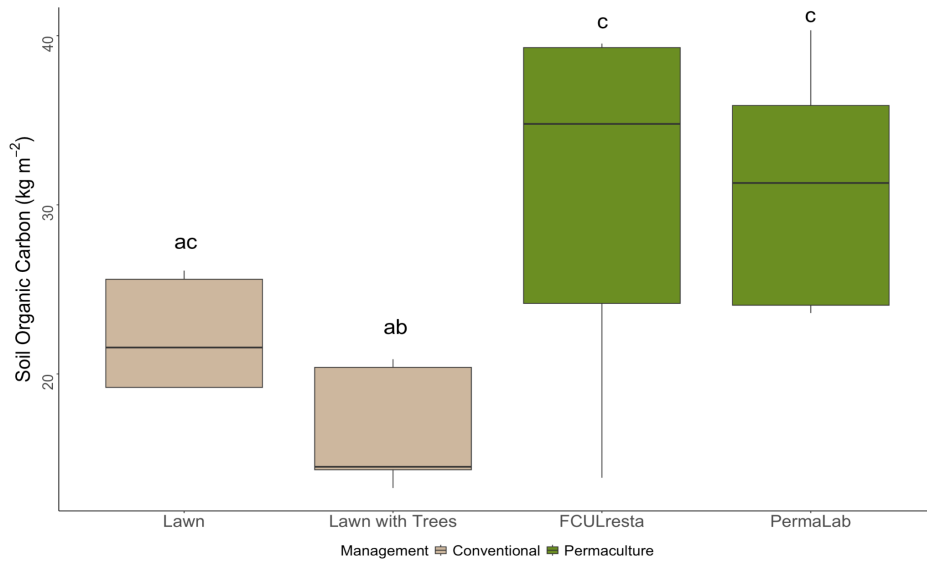


Figure 3.7. Soil organic carbon stock in kg m⁻² across sites. Letters indicate significant differences between sites resulting from pairwise t-test with holm correction ($\alpha = 0.05$).

When comparing management types, a Wilcoxon rank-sum test confirms that SOC concentrations in conventional sites ($20.744 \pm 9.455 \text{ g kg}^{-1}$) were significantly lower than in Permaculture sites ($52.316 \pm 37.584 \text{ g kg}^{-1}$) ($W = 80, p = 0.001$). Similarly, SOC stock was significantly higher under permaculture management ($30.68 \pm 8.876 \text{ kg m}^{-2}$) than under conventional management ($19.502 \pm 4.455 \text{ kg m}^{-2}$) Welch's $t(13.263) = -3.559, p = 0.003$.

Extrapolating SOC stock to all possible green areas within the FCUL campus showed the same pattern: conventional management corresponded to an estimated carbon stock of $328.88 \pm 75.12 \text{ t C}$, whereas permaculture management would result in a carbon stock of $517.4 \pm 149.69 \text{ t C}$ (Figure 3.8).

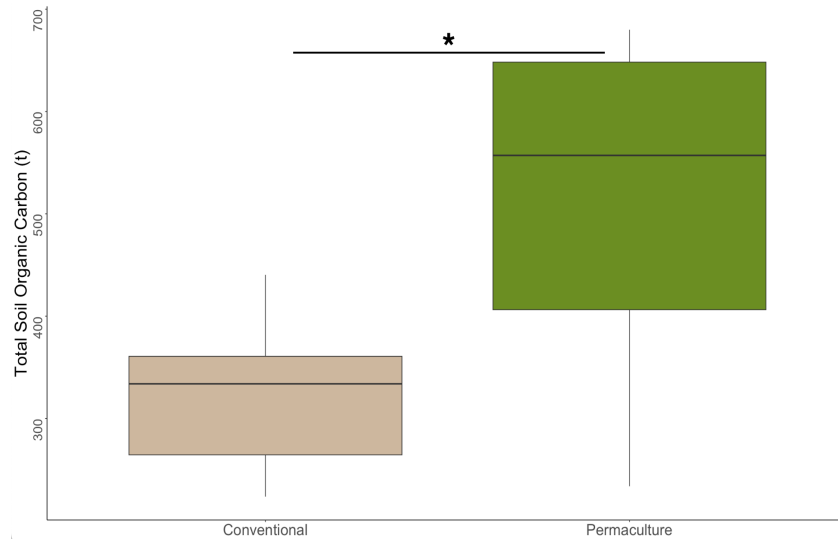


Figure 3.8. Total Soil organic carbon across management types. Management differences were assessed using Welch's t-test. Asterisks indicate significant differences between sites within each exposure level, * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$.

3.1.5 Soil Capacity for Glyphosate Degradation

An analysis of variance revealed that residual glyphosate concentrations differed significantly among sites and the control ($F(8) = 79.56, p < 0.001$) (Figure 3.9).

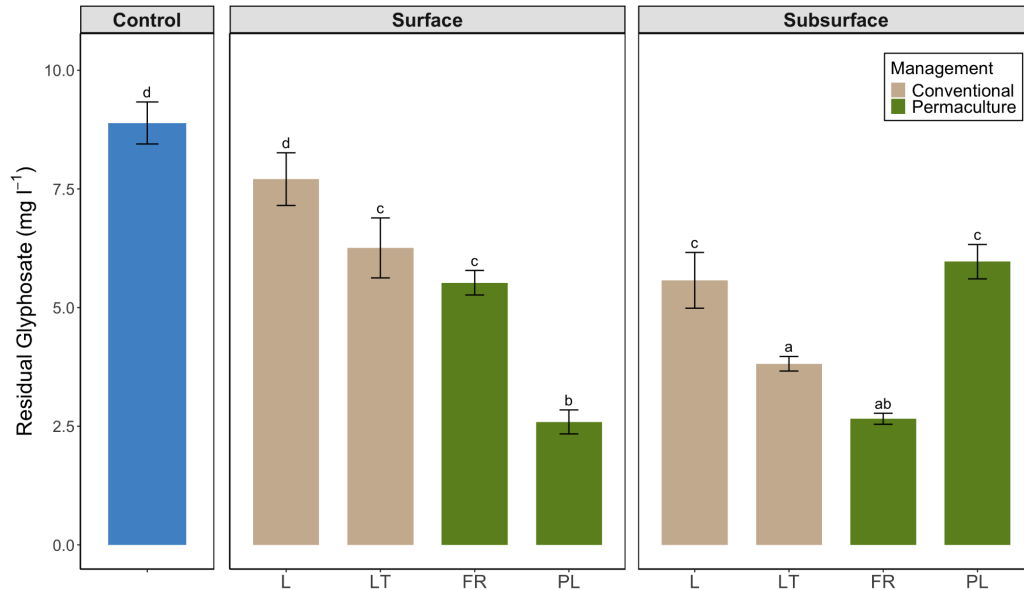


Figure 3.9. Quantification of residual glyphosate in soil samples. Values represent the mean \pm standard deviation. Letters indicate significant differences between sites (Tukey's test, $p < 0.05$). L = Lawn; LT = Lawn with Trees; FR = FCULresta; PL = PermaLab.

In the surface layer, residual glyphosate was highest in the Lawn ($7.707 \pm 0.555 \text{ mg l}^{-1}$), where it didn't differ significantly from the control ($8.89 \pm 0.44 \text{ mg l}^{-1}$, $p = 0.051$), and decreased significantly with Lawn with Trees ($6.257 \pm 0.633 \text{ mg l}^{-1}$, $p < 0.001$) and FCULresta ($5.523 \pm 0.259 \text{ mg l}^{-1}$, $p < 0.001$). It reached its lowest value in PermaLab, differing significantly from all sites and the control ($2.59 \pm 0.252 \text{ mg l}^{-1}$, $p = 0$).

In the subsurface soil layer, the same decreasing pattern is repeated, where residual glyphosate was highest in the Lawn ($5.573 \pm 0.588 \text{ mg l}^{-1}$) but significantly lower than the control ($p < 0.001$). It decreases with Lawn with Trees ($3.817 \pm 0.154 \text{ mg l}^{-1}$, $p = 0$) as well as FCULresta ($2.657 \pm 0.117 \text{ mg l}^{-1}$, $p = 0$), where FCULresta reached a similarly low level as PermaLab in the surface layer. In PermaLab ($5.967 \pm 0.362 \text{ mg l}^{-1}$), residual glyphosate increased to a similar level as the Lawn ($5.967 \pm 0.362 \text{ mg l}^{-1}$) yet was still significantly lower than the Control ($p < 0.001$).

3.2 Lichens

3.2.1 Photosynthetic Activity

Fv/Fm, a measure of photosynthetic activity, declined significantly over time across all sites ($p < 0.0001$), regardless of management type or traffic exposure level (Figure 3.10). Linear mixed-effects models showed no evidence that the rate of decline differed between conventional and permaculture management, neither in low exposure sites ($p = 0.532$), nor in high exposure sites ($p = 0.424$). These findings suggest that temporal declines in lichen vitality were consistent across management and exposure conditions throughout the whole exposure period.

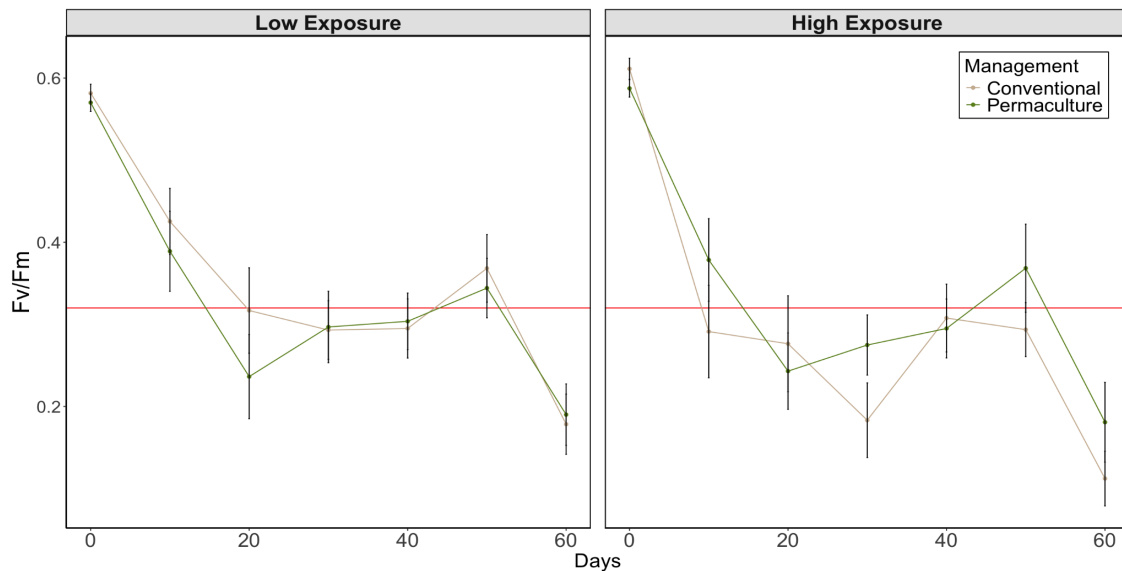


Figure 3.10 Fv/Fm over time and across management types and exposure levels. Values represent the mean \pm standard deviation (Conventional low exposure $n = 4$, Permaculture low exposure $n = 4$, Conventional high exposure $n = 3$, Permaculture high exposure $n = 4$). The red line marks the lichen mortality threshold of Fv/Fm = 0.32.

Differences in Lichen mortality, that is days to death (DTD), between management practices within each exposure level were assessed using student's t -tests. In low exposure sites, conventional (36 ± 13.14 DTD) and permaculture (30 ± 22.17 DTD) did not differ significantly, Students $t(6) = 0.464$, $p = 0.659$. In high exposure sites, conventional (26 ± 11.22 DTD) and permaculture (39 ± 26.77 DTD) also did not differ significantly Students $t(5) = -0.8$, $p = 0.459$.

3.2.2 Magnetic Susceptibility

Mass specific magnetic susceptibility (χ) of lichens differed significantly between sites with high exposure, where it was higher in the conventional site ($4.83 \times 10^{-8} \pm 1.58 \times 10^{-8} \text{ m}^3 \text{ kg}^{-1}$) than in the permaculture site ($3.34 \times 10^{-8} \pm 7.69 \times 10^{-9} \text{ m}^3 \text{ kg}^{-1}$), Welch's $t(14.94) = 3.02$, $p = 0.009$ (Figure 3.11). In low exposure sites no significant difference was found, Welch's $t(23.74) = -0.45$, $p = 0.654$). Across exposure levels, χ differed significantly in both management types, where high exposure sites showed higher values than low exposure sites: Lawn ($2.47 \times 10^{-8} \pm 6.07 \times 10^{-9} \text{ m}^3 \text{ kg}^{-1}$), Welch's $t(13.79) = 4.88$, $p = 0$, and permaculture ($2.56 \times 10^{-8} \pm 4.5 \times 10^{-9} \text{ m}^3 \text{ kg}^{-1}$), Welch's $t(24.17) = 3.48$, $p = 0.002$.

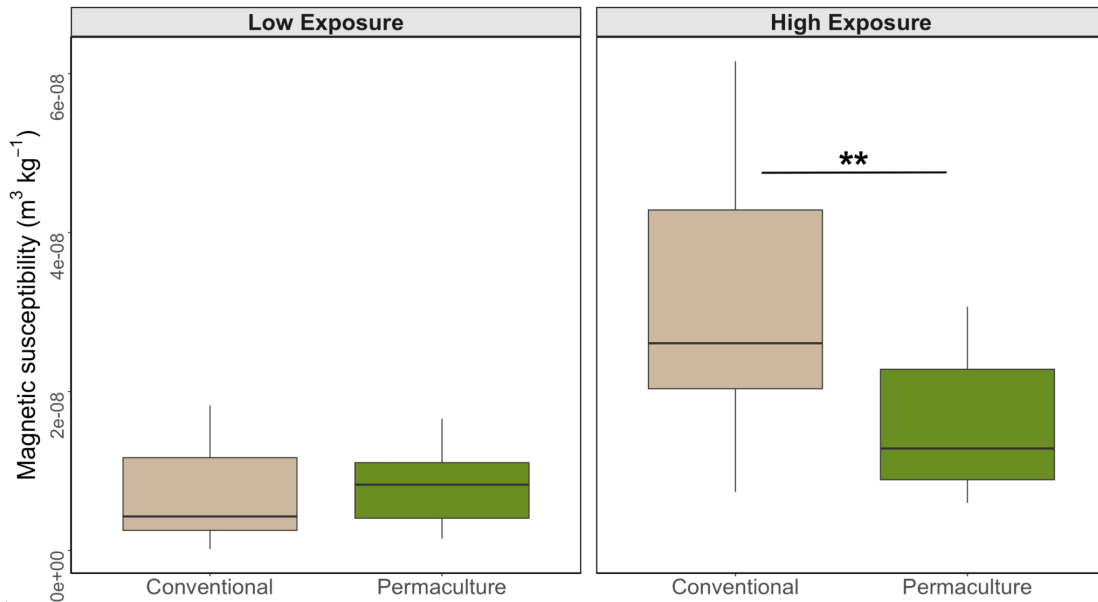


Figure 3.11 Magnetic susceptibility across sites and exposure levels. Management differences within exposure levels were assessed via Welch's t-test. Asterisks indicate significant differences between sites within each exposure level, * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$.

3.2.3 Metal Contamination

Several metals were detected in the exposed lichens (Table 3.4). Management differences in each exposure level were assessed using Wilcoxon rank-sum tests with Holm correction, yet no significant differences were found for any metal.

Lichens across all sites exhibited metal accumulation of metals associated with anthropogenic pollution compared to unexposed controls (Table 3.5). The E/C ratio revealed that lichens from managed sites accumulated greater amounts of metals connected to anthropogenic pollution. In the – Exposure sites both management types saw accumulation of Ba, Cd, Cu, Fe and Mn, whereas the level of Zinc was comparable to control. In the + Exposure sites, both management types saw accumulation of Ba, Cd and Cu. Mn only accumulated in the permaculture site. Fe and Zn didn't show accumulation for either management type.

Table 3.4. Metal contamination measured in Lichens across management types and exposure levels. Depicted are mean \pm standard deviation. Management differences in each exposure level were assessed using Wilcoxon rank-sum tests with Holm correction (n = 16 for Conventional - Exp, n = 14 for Permaculture - Exp, n = 12 for Conventional + Exp, n = 16 for Permaculture + Exp).

Concentration (mg kg ⁻¹)	- Exposure		+ Exposure	
	Conventional	Permaculture	Conventional	Permaculture
Aluminium	940.94 \pm 135.6	866.72 \pm 210.43	725.5 \pm 165.23	789.41 \pm 239.29
Barium	17.68 \pm 4.15	16.15 \pm 2.63	15.54 \pm 5.38	16.11 \pm 7.81
Calcium	7366.19 \pm 1635.69	6311.41 \pm 1189.77	5564.82 \pm 1040.90	5593.21 \pm 1262.23
Cadmium	0.061 \pm 0.026	0.055 \pm 0.018	0.022 \pm 0.033	0.04 \pm 0.035
Copper	7.77 \pm 0.79	7.48 \pm 0.69	7.17 \pm 0.99	6.82 \pm 0.99
Iron	879.8 \pm 132.57	816.03 \pm 196.42	691.83 \pm 118.39	730.84 \pm 218.48
Potassium	3008.6 \pm 317.46	2939.12 \pm 276.94	2394.6 \pm 166.77	2559.51 \pm 273.71
Magnesium	1291.91 \pm 111.44	1275.41 \pm 104.52	1022.99 \pm 78.7	1033.97 \pm 99.57
Manganese	128.85 \pm 38.56	123.92 \pm 46.63	107.04 \pm 36.04	162.46 \pm 172.12
Sodium	907.73 \pm 377.28	763.64 \pm 148.57	751.91 \pm 214.17	583.4 \pm 46.81
Strontium	23.31 \pm 4.76	20.47 \pm 2.82	18.71 \pm 4.59	18.06 \pm 4.13
Zinc	22.29 \pm 2.78	19.66 \pm 1.98	18.95 \pm 2.6	17.8 \pm 2.39

Table 3.5. Exposed to Control Ratio (E/C ratio) and relative accumulation/loss (A/L) class of metals associated with anthropogenic pollution. Values $>$ 1 indicate enrichment of metal in lichens compared to control.

	- Exposure				+ Exposure			
	Conventional		Permaculture		Conventional		Permaculture	
	E/C ratio	A/L class	E/C ratio	A/L class	E/C ratio	A/L class	E/C ratio	A/L class
Barium	1.68	A	1.53	A	1.48	A	1.53	A
Cadmium	4.55	SA	4.09	SA	1.6	A	3	SA
Copper	1.38	A	1.44	A	1.49	A	1.31	A
Iron	1.5	A	1.38	A	1.17	N	1.24	N
Manganese	1.38	A	1.32	A	1.14	N	1.73	A
Zinc	0.99	N	0.87	N	0.84	N	0.79	N

N = normal; A = Accumulation; SA = severe accumulation; L = Loss

4. Discussion

4.1 Soils

Our findings suggest that management practices strongly influence soil properties, with permaculture sites generally exhibiting improved soil health indicators such as lower BD, greater SOM content and enhanced potential for carbon sequestration compared to conventional sites. These interrelated properties govern soil structure, nutrient availability, and hydrological functioning. Furthermore, our results indicate that management-driven changes may also enhance the soil's ability to degrade pollutants such as the herbicide glyphosate, providing additional ecosystem services.

4.1.1 Management Practices Influence Soil Properties

BD varied between sites and across soil depths, with higher BD generally observed under conventional management compared to permaculture. This effect was most pronounced in the surface soil layer, as differences diminished with increasing soil depth. This pattern aligns with established findings that BD increases with depth due to reduced SOM content, lower root density and the compacting weight of overlying soil (Burghardt, 2025; Chaudhari et al., 2013). Furthermore, with increasing soil depth the influence of management practices may be more limited as soil properties are increasingly driven by inherent characteristics such as parent material and long-term management history.

BD is determined by multiple variables, but texture is among the most critical. All soils analysed in this study were composed of more than 90% sand and thus firmly into the “sand” category of the USDA soil texture triangle (Soil Science Staff, 2017). Such coarse textures are typically associated with lower BD compared to silt or clay soils. A common reference value for sandy soils is 1.6 g cm^{-3} , above which compaction can begin to impair root growth, water infiltration, and microbial activity (USDA, 2008). All soils in this study exhibited BD well below this threshold, regardless of management practice, likely due to their inherently loose and coarse structure.

Sandy soils provide both benefits and limitations for ecosystem services. On the one hand, they are highly permeable, promoting water infiltration and reducing surface runoff, thus increasing flood mitigation potential. However, their poor water retention reduces water availability during dry periods, leading to increased drought stress and seasonal dieback of the vegetation. This was visible in the sampled lawn site, where the turfgrass thrived during rainfall season and partially died back during dry periods in summer (Figure 4.1). Previous research has shown that healthy lawns can act as carbon sinks, but drought-stressed lawns may shift into carbon sources due to reduced plant productivity and increased soil respiration (Townsend-Small & Czimczik, 2010). Repeating soil respiration measurements in the wet- and dry season would provide valuable insights into how seasonal vegetation dynamics alter soil carbon balances in these lawns.



Figure 4.1. Lawn sampled in this study in (a) February 2024 and (b) July 2024. (Source: Google Earth, 2025).

4.1.2 Organic Matter Inputs Shape SOM–BD Dynamics

Management practices in permaculture sites further reduced low BD values in surface horizons, particularly in FCULresta. This reduction in BD is consistent with the well-established negative relationship between SOM and BD, where higher SOM inputs reduce BD by increasing soil porosity (Chaudhari et al., 2013). Management in the permaculture sites emphasizes organic matter addition through compost, which is known to decrease BD due to the high porosity of incorporated plant material (Leitão et al., 2018). However, the two sites differ in the temporal dynamics of their SOM accumulation. PermaLab, the older site, received less frequent interventions prior to sampling than the younger site FCULresta. FCULresta is subject to frequent interventions, including the repeated application of compost. The addition of compost has been shown to significantly increase SOM concentrations in topsoil (Ulm et al., 2019). This likely explains the high and variable SOM concentrations observed in the surface soil layer, where compost is inputted but is not distributed evenly throughout the whole site, creating localised “pockets” of extremely high SOM, with individual samples reaching up to 24.57% SOM content. In the subsurface soil layer, SOM concentrations were lower, suggesting that intervention impact has not extended into lower soil layers of FCULresta. In contrast, PermaLab, while not reaching as extreme SOM concentrations as FCULresta, showed similarly high SOM levels in both surface and subsurface layers, indicating a more even vertical distribution of SOM, suggesting that long-term cumulative inputs have led to a more even distribution of SOM throughout the soil layers. These differences illustrate how management intensity and temporal scale interact to shape soil properties. Intensive short-term inputs can rapidly boost SOM, while longer-term management builds a more stable and vertically integrated SOM profile.

4.1.3 Implications for SOM Stocks and Ecosystem Services

While permaculture sites exhibited higher SOM concentrations, the associated reductions in BD partially offset these gains when SOM stocks were calculated on a mass per area basis. This effect was particularly evident in the surface horizons of FCULresta, where very low BD reduced the mass of soil per unit area, thereby limiting the translation of high SOM concentrations into higher SOM stocks. Conversely, in the subsurface horizons where BD did not differ significantly across sites, higher SOM concentrations in PermaLab translated directly into increased SOM stocks relative to conventional sites. These findings highlight the importance of considering both SOM and BD in tandem when estimating SOM storage potential, as changes in one property can alter interpretations of the other.

The implications of these SOM–BD dynamics extend beyond OM storage to broader ecosystem services. Higher SOM enhances nutrient availability, soil aggregation, water-holding capacity, and microbial activity (Janzen et al., 1997; Lehman et al., 2015). In sandy soils, where water retention is inherently limited, the capacity of SOM to improve moisture storage is especially valuable for sustaining vegetation and reducing drought stress. Thus, sustainable management practices not only improve soil health and carbon sequestration but also contribute to pollution mitigation and overall ecosystem resilience.

4.1.4 Soil Fractions and Organic Matter Stock

Understanding how SOM is distributed among soil fractions is essential for assessing quality and turnover. This has direct implications for carbon sequestration potential, since long-term storage is strongly linked to the stabilization of OM in soils (Bramble et al., 2025).

Roots and coarse OM represent the least stable OM pools. Their large surface area and lack of protection within the soil matrix make them easily accessible for microbial decomposition, reducing their turnover time compared to more stable forms of OM. In all our sampled soils both fractions together accounted for <1% of total soil mass, yet their high OM concentrations meant they contributed disproportionately to total OM stocks. For example, in the surface soil of the lawn with trees, roots comprised only ~0.7% of the soil mass but contributed up to 13% of total OM stock.

Within the < 2 mm soil fraction, SOM can be divided into two functionally distinct pools through particle size: Particulate organic matter (POM; 2000 - 63 μm) and mineral-associated organic matter (MAOM; < 63 μm) (Leuthold et al., 2024). This distinction provides valuable insights into OM turnover rate and response to disturbance (Georgiou et al., 2025). As OM decomposes, it binds strongly with silt and clay aggregates, protecting it from microbial decomposition, making MAOM the most stable form of OM (Bramble et al., 2025; Yost & Hartemink, 2019). Studies show that mineral-associated organic carbon (MAOC) has turnover rates up to 1000 times slower than particulate organic carbon (POC), reaching 100 - 10 000 years (Georgiou et al., 2022). Since MAOC is the carbon component of MAOM, this difference broadly reflects the greater stability of MAOM compared to POM.

Reported MAOM saturation levels differ by texture. While in European soils a maximum carbon potential of 45 - 50 g C kg⁻¹ in MOAM is reported, sandy soils reach MAOM saturation at lower concentrations due to limited mineral surface area (Georgiou et al., 2022; Six et al., 2024). Across all sites in our study, the < 63 μm fraction made up only a small proportion of soils and exhibited low OM concentrations (max. ~ 11.2 g OM kg⁻¹ = 6.5 g C kg⁻¹), with concentrations only differing in the subsurface soils of Lawn and FCULresta. These values could suggest near-saturation, yet comparisons between sites point in the opposite direction. For example, lawn with trees had substantially higher silt+clay content than lawn soils, yet OM concentration and stock were similar in these sites, indicating that MAOM saturation in lawn with trees had not been reached. This highlights that while fine particles can stabilize OM in theory, actual OM stock depends on multiple factors such as bulk density and OM input quantity. Limited organic inputs keep OM low, even in soils with higher silt+clay content. Comparisons between management types further show the complex interplay between these factors. In surface soils, Lawn and PermaLab had similar sized silt+clay fractions, yet OM stocks were higher in Lawn despite comparable OM concentrations. In the subsurface, lawn with trees contained significantly more silt+clay than PermaLab, but SOM stocks did not differ, even

though OM concentrations were similar. These observations suggest that management practices had little detectable effect on MAOM accumulation in the sampled soils. Nonetheless, the consistently low OM concentrations in the fine fraction indicate that MAOM saturation has not been reached at any site.

Given the inherently low MAOM capacity of sandy soils, carbon storage potential is more dependent on POM pools (Six et al., 2024). Here, management effects were evident. Coarse sand OM concentrations were lower in conventionally managed sites than in permaculture sites. Temporal effects of management were also apparent: in FCULresta, surface soils displayed high OM concentrations while subsurface soil displayed low OM concentrations. By contrast, PermaLab soils exhibited evenly high OM concentrations throughout the soil profile, suggesting that with time management-driven OM enrichment penetrates deeper into the soil horizons, underscoring the cumulative effect of organic inputs over longer management histories. A similar, though not statistically significant, trend was observed in fine sand fractions, where permaculture sites tended to have higher OM concentrations. Overall, these results indicate that permaculture management significantly increases enrichment of POM compared to conventional management and that these effects accumulate over time.

4.1.5 Soil Organic Carbon

Soil organic carbon (SOC), the primary component of SOM, is a key indicator of soil quality and a critical factor determining carbon sequestration potential (Lal, 2015; Reeves, 1997). SOC promotes water infiltration, storage and drainage and is directly related to the maintenance of soil structure, microbial communities, and nutrient availability (Martinez et al., 2010). SOC in the <2mm fraction is influenced by a multitude of factors including climate, soil texture and land management practices. Generally, SOC increases with higher rainfall, decreases with rising temperatures and is lower in sandy soils than in loam and clay soils (Reyna-Bowen et al., 2019; Yost & Hartemink, 2019). Maintaining SOC above a threshold of 1 - 2% SOC, or 10 - 20 g C kg⁻¹, is considered essential for sustaining soil structure and aggregation as well as nutrient and water retention (Lal, 2014, 2015; Patrick et al., 2013). Our results show that SOC concentrations differed across management types. Permaculture sites exhibited SOC concentrations well above this threshold, whereas conventional sites barely met or fell below it. On average, conventional management maintained SOC concentrations of 2.07% while under permaculture management SOC increased to 5.23%. These results indicate that conventional management barely maintains soil quality, while permaculture management significantly enhances SOC accumulation. SOC stock followed a similar pattern as SOC concentrations, that is SOC stock increased under permaculture management. Extrapolating our findings to all green spaces on the FCUL campus suggests that permaculture management could store an additional ~188.52 t of carbon, representing a 57% increase compared to conventional management. With per capita CO₂ emissions of 3.42 t in Portugal (IEA, 2023), the conversion of lawns to agroforests just on the FCUL campus could buffer the emissions of one citizen for ~ 55 years. This demonstrates that even small-scale interventions can have substantial impacts on carbon storage and urban ecosystem services. The observed differences in SOC are particularly relevant given the study area's sandy soils and hot, dry summers, conditions that naturally limit SOC accumulation. Water availability likely constrains vegetation growth, influencing carbon inputs to the soil, and consequently, SOC levels. Furthermore, changes in SOC affect microbial biomass and community structure. By increasing SOC, sustainable management practices may enhance microbial activity, thus promoting soil functions such as pollution mitigation, (Blanco-Canqui et al., 2013).

Land cover and land-use changes also have significant impacts on SOC. For example, sandy soils under forest can contain up to 90 g kg⁻¹ SOC, yet conversion to agriculture can reduce SOC by up to 40% (Yost & Hartemink, 2019). Our findings support these patterns, demonstrating that management strategies which increase organic inputs and reduce disturbance can substantially enhance SOC, even in environments prone to carbon loss.

SOC constitutes one of the largest and most dynamic carbon pools in the global carbon cycle (Jobbágy & Jackson, 2000). Enhancing SOC through sustainable soil management is therefore not only vital for soil health but also represents an effective strategy for reducing atmospheric carbon (Reeves, 1997). The substantial SOC gains under permaculture management observed in this study highlight the potential of sustainable land management to improve soil quality and promote carbon sequestration, even at relatively small spatial scales. Implementing permaculture systems across urban green spaces can provide tangible benefits for soil health, ecosystem services, and climate change mitigation.

4.1.6 Glyphosate Degradation

Residual glyphosate (GLP) concentrations revealed an increased capacity of permaculture systems to degrade GLP. In the surface soils, the Lawn retained the highest GLP residues while permaculture systems, particularly the older site PermaLab, showed significantly lower residual GLP. This decline suggests that the GLP degradation potential of the soil increases with structural complexity of the system. GLP degradation is considered to be a primarily microbiological process and while accomplished by various microorganisms, seems mainly driven by bacteria such as *Pseudomonas* spp. (Borggaard & Gimsing, 2008). Increases in SOM are intimately linked to size, activity, and composition of the soil microbial community (Lehman et al., 2015). Therefore, it is likely that increased SOM in permaculture systems is accompanied by an increase in microbial abundance and activity, enhancing the capacity for microbial breakdown of GLP.

The fate of GLP in soils is closely linked to sorption dynamics. Degradation has been shown to be inversely correlated with GLP sorption capacity of the soil, that is, strong sorption reduces GLP degradation due to its limited bioavailability (Borggaard & Gimsing, 2008). Yet, sorption of GLP in soils is highly variable. Sandy soils have shown lower sorption than clay soils, suggesting that clay minerals are largely responsible for sorption, leading to faster GLP degradation rates in sandy soils (Vereecken, 2005). The influence of SOM on sorption is controversial, where on one hand studies have shown that GLP sorption is not correlated to SOM (Borggaard & Gimsing, 2008), yet the mechanism seems more complex and varies between soils. In sandy soils, SOM has been shown to compete with GLP for adsorption sites, thus inhibiting GLP sorption, as well as GLP sorption decreasing with increases in SOC (Gerritse et al., 1996). While the sandy soils in the study area naturally seem to facilitate degradation, the accumulation of SOM in the permaculture sites may optimize conditions facilitating degradation by simultaneously stimulating microbial activity as well as reducing sorption sites for GLP.

Nutrient status provides an additional layer of influence. Due to similar sorption mechanisms, phosphate competes with GLP for binding sites, thus increasing GLP mobility with increasing phosphate levels (Borggaard & Gimsing, 2008). Permaculture systems are subject to nutrient enriching practices such as OM inputs through compost and leaf litter, maintaining higher soil fertility. This mechanism may further contribute to the reduced persistence of GLP in these soils. However, in the subsurface layer of PermaLab,

an exception to this trend was observed, with GLP concentrations remaining as high as in the Lawn. While phosphate concentrations were not measured in this study, it is possible that strong nutrient cycling and phosphate uptake from vegetation reduces phosphate availability in the subsurface soil due to decreased leaching. Lower phosphate levels would increase GLP sorption, thus maintaining increased GLP levels. Measurement of phosphate concentrations would bring valuable insights into this dynamic.

Overall, these results suggest that permaculture systems have a higher capacity to degrade GLP. Faster degradation reduces risk of herbicide accumulation in soil as well as its leaching through the soil profile into groundwater (Borggaard & Gimsing, 2008; Sviridov et al., 2015). In terms of ecosystem services, these results suggest that management practices fostering OM and C accumulation, microbial diversity and nutrient cycling enhance the resilience of soils to herbicide inputs.

4.1.7 Management intensity and labour inputs as underlying drivers

Beyond structural differences in vegetation, the observed contrasts between lawns and permaculture agroforests are also rooted in fundamentally different management regimes. Conventional lawns on the FCUL campus are managed through relatively standardised, low-diversity practices where lawns are mowed regularly, and grass clippings are removed consequently. This management results in continuous export of biomass and nutrients, constraining the accumulation of SOM and limiting opportunities for soil recovery. In contrast, the permaculture agroforests are managed through practices that prioritise soil regeneration and system resilience. Organic matter inputs are continuous and diverse, originating from leaf litter, pruning residues, mulch, and compost additions. These inputs not only increase total organic matter stocks but also promote the development of heterogeneous soil microhabitats that support microbial activity and nutrient cycling. An important factor in this context is the integration of the HortaFCUL project within the wider university infrastructure. Virtually all organic waste produced on campus, that is food waste from the cafeterias as well as organic residues produced through routine campus maintenance is delivered to the HortaFCUL headquarters where it is composted. This system enables the production of approximately 8 t of high-quality compost per year (Vaz Pato et al., 2024), which is subsequently distributed throughout the agroforests. These substantial organic inputs are likely a key driver of the rapid and successful establishment of these agroforests. The higher SOM and SOC values and lower BD observed in the agroforests can most likely be traced to these management differences.

Another key distinction between the two management types is time and labour input. The agroforests investigated in this study are largely maintained through weekly volunteer-based work, embedded in the HortaFCUL community. While this results in higher labour inputs per unit area compared to the conventional lawns, it enables low-impact management practices that would be difficult to implement under conventional maintenance regimes. Tasks such as selective pruning, mulching, manual planting, and system observation are time-intensive but directly aligned with ecological processes, contributing to gradual improvements in ecosystem functioning. Furthermore, this volunteer-driven model can provide the possibility of shifting management costs from direct financial expenditure to social engagement and community organisation.

These differences have important implications for the interpretation and transferability of the present results. The enhanced soil health and functionality observed in the agroforests cannot be attributed solely to

vegetation complexity, but rather emerge from interaction between sustained, ecology-oriented management and the structural diversity that this management approach facilitates. The strong performance of the observed agroforests in terms of ecosystem functioning appear to be driven in large parts by the high inputs of both organic material as well as dedicated labour. This raises questions regarding scalability and long-term maintenance if such systems were to be implemented more widely across urban landscapes, where voluntary work and organic material might not be as abundantly available as on the FCUL campus. While this represents a potential limitation, it can also be viewed as an opportunity to foster stronger human–nature interactions and community engagement within urban green space management.

4.2 Lichens

Our results show that *E. prunastri* respond to regionally distributed stressors as well as to local management-driven effects. Decline rate of photosynthetic performance as well as metal accumulation was consistent across management practices and exposure levels, reflecting broad atmospheric stress. Yet, differences in magnetic susceptibility revealed that vertical strata of permaculture agroforests can act as a buffer for the anthropogenic fraction of PM, indicating that management practices can yield relevant ecological results.

4.2.1 Photosynthetic Activity

Fv/Fm indicates the maximum quantum yield of photosystem II and it is a ratio used to evaluate the efficiency of photosynthesis, here used as a proxy for lichen vitality, with optimal dark-adapted values of 0.79 to 0.83 (Bartold & Kluczek, 2024). Fv/Fm decreases when lichens are exposed to biotic or abiotic stress or when the thallus' water content decreases. Before exposure, the average baseline Fv/Fm measured in our thalli was 0.59, indicating that lichens may already have experienced stress prior to exposure. This may have contributed to the uniform decline across sites and needs to be taken into account when interpreting these results.

Lichens are poikilohydric organisms, that is they do not have mechanisms to regulate their water content. Instead, they equilibrate through the moisture available in the surrounding atmosphere, making them sensitive to environmental changes (Stanton et al., 2023). Their desiccation tolerance allows them to survive complete drying by entering anhydrobiosis until the next rehydration. Thus, they theoretically can survive infinite dehydration/rehydration cycles (Gasulla et al., 2021), making it unlikely that the dry climate with low rainfall prevalent during the exposure period was the driving factor behind the decline in photosynthetic activity. Similarly, temperatures during the exposure period ranged from 12 °C to 39 °C, falling within the limits of what *E. prunastri* can survive before deterioration occurs (Pisani et al., 2007), suggesting that thermal stress was also not the main cause of vitality loss. Therefore, the most probable explanation of decline is pollution induced stress.

4.2.2 Magnetic Susceptibility

Although the accumulation of the magnetic metal Fe showed a homogeneous pattern across management types, differences in magnetic susceptibility (χ) were found. In high exposure sites, χ was higher in the conventional site than in the permaculture site, highlighting source differences of Fe. Traffic emissions release tiny magnetite-rich particles, with brake wear being the dominant source of airborne magnetite

(Gonet et al., 2021). Magnetite (Fe_3O_4) is strongly magnetic, while geogenic Fe from soil and dust typically occurs as weakly magnetic minerals such as hematite and goethite. Consequently, even small inputs of anthropogenic magnetite can increase χ without shifting total Fe content. The lower χ observed in the permaculture site suggests reduced exposure to magnetite-bearing PM compared to the conventional site. This may be due to the vertical vegetation structure in the permaculture system, which possibly intercepts and reduces deposition of traffic-derived airborne PM, therefore acting as a barrier for pollution.

4.2.3 Metal Accumulation

The overall metal profile indicates that the dominant pollution sources across all sites are traffic-related, as reflected by the enrichment of Cd, Cu, Fe, Ba, and Mn (Abecasis et al., 2022; X. Li et al., 2019; Sager, 2020). In contrast, elements such as Al, Ca, K, Mg, Na, and Sr are primarily of geogenic origin, representing dust and soil resuspension inputs rather than direct anthropogenic emissions (Tang & Han, 2019). Metal accumulation followed a relatively uniform pattern across management types and exposure levels, indicating that background pollution may be sufficiently high and spatially widespread to mask potential local differences due to green space management.

Metal accumulation in lichens has been shown to induce oxidative stress and stimulate the formation of reactive oxygen species (ROS) (Kováčik et al., 2018; Oztetik & Cicek, 2011). Moreover, metals can directly interfere with photosynthetic electron transport in the photobiont, thereby inhibiting photosynthesis (Sigfridsson et al., 2004; Gaziyev et al. 2011). These processes, in connection to the homogenous presence of metals, likely caused the observed uniform decline in photosynthetic efficiency (F_v/F_m) across all sites.

5. Conclusion and Outlook

This study indicates that permaculture-based agroforestry can provide meaningful contributions to ecosystem functioning in urban green spaces, even at small scales. The permaculture sites showed improvements in several soil properties when compared with conventional lawns, including lower bulk density, higher soil organic matter and soil organic carbon and an enhanced potential for glyphosate degradation. These improvements are associated with management practices typical of permaculture, such as the addition of organic matter through compost and leaf litter, reduced soil disturbance and higher plant diversity. Together, these practices support SOM accumulation, stimulate microbial communities, and contribute to greater pollutant degradation capacity and carbon storage potential.

Differences observed between the two permaculture sites highlight the influence of management duration. In variation in soil organic matter, particularly within the particulate organic matter fraction suggests that longer-term management leads to more stable and evenly distributed improvements in soil health, while younger systems generate quicker but less consistent changes. This points to the importance of cumulative ecological processes in strengthening soil quality over time.

Aboveground, the structural complexity of permaculture systems also provided significant benefits in terms of air pollution exposure. The dense, multi-layered vegetation likely acted as a barrier to particulate matter, and differences in magnetic susceptibility measurements support the idea that vertical vegetation structures can reduce the deposition of traffic-related pollutants.

Conventional lawns, by contrast, generally exhibited lower SOM and SOC levels, limited capacity for pollutant degradation and little evidence of aboveground benefits for air quality. While lawns provide social and recreational value, their ecological contributions appear comparatively modest.

Taken together, these findings provide strong support for the ecological advantages of structurally complex urban ecosystems over simplified systems such as lawns. Permaculture agroforestry demonstrates the potential to create multifunctional landscapes that integrate soil health, carbon storage, microbial functionality, and pollution mitigation into a cohesive ecological framework. However, while the results are compelling, additional empirical research is needed to corroborate and extend these conclusions. Future work should span a holistic perspective, explicitly examining all four categories of ecosystem services — provisioning, regulating, supporting, and cultural — in order to fully capture the breadth of benefits offered by permaculture-based compared to conventional management. Such research would not only strengthen the scientific foundation for permaculture in urban contexts but also inform urban planning and policy, highlighting opportunities to diversify and maximize ecosystem services where conventional approaches remain unnecessarily limited and ecologically constrained.

6. References

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