Improving soil health in apple and pear orchards

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Declaration

This thesis has not been submitted in support of an application for another degree at this or any other university. It is the result of my own work and includes nothing that is the outcome of work done in collaboration except where specifically indicated.

Abstract

Commercial apple and pear orchards represent a small but important source of food production. Contemporary research has targeted ways to improve orchard ecosystem services such as pollination and natural enemies through sown wildflower strips. The potential benefits of increased plant diversity in orchard alleyways to improve soil health have not been considered. This project aimed to answer this question through four research objectives. Initially, apple and pear orchard management and its influence on soil processes was reviewed to provide context and highlight areas for further research. Then, a field study benchmarked soil health in apple orchards with and without wildflower strips. No consistent trends were found in biological, chemical, or physical soil health indicators. Following this I performed a lab study using intact soil cores from wildflower and conventionally managed orchard alleyways to measure the response (CO_2) of orchard soils to drought and rewetting. No difference in responses was detected between management. Results from these studies, alongside the management review, indicated orchard management practices, such as agrochemical application and soil compaction from trafficking, may stifle any benefits plant diversity may have on soil health. A fourth chapter explored the effects of representative orchard covers in mesocosms on soil processes and aboveground productivity. A litter decomposition experiment using mesocosm litter was also performed. Significant effects on nitrogen cycling and aboveground productivity were detected in pots containing two or more functional groups, with dominant biomass Trifolium pratense likely driving these effects. Plant available nitrogen (NH_4^+ + NO_{3^-}) in soil was also shown to be significantly reduced under the presence of forbs. Litter containing two or more functional groups was also found to increase the rate of litter mass loss by 100% over six months. Overall, this thesis examines orchard soil health, and how soil health may change under management, drought stress, or how function may be influenced by plant functional groups.

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1. Introduction

1.1 Introduction – soil health and ecosystem services

In agriculture, the term “soil health” is used to describe the capacity of soils to provide ecosystem services (Kibblewhite et al., 2008). Definitions focus on the primary idea of supporting ecosystem services: “The continued capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain biological productivity, promote the quality of air and water environments, and maintain plant, animal and human health” (Doran & Safley, 1997). The way soil health is determined changes with ecosystems and the services they provide. Agroecosystems provide diverse ecosystem services. Therefore soils must be managed in a way to sustain properties and processes that drive ecosystem services (Kibblewhite et al., 2008).

Dramatic changes have occurred in natural biological, chemical and physical cycles due to anthropogenic modification of soil and its associated habitats. This has effected the ecosystem services soils provide (Doran & Zeiss, 2000; Haygarth & Ritz, 2009). Ecosystem services are defined as the “the benefits people obtain from ecosystems. These include provisioning services such as food, water, timber, and fibre; regulating services that affect climate, floods, disease, wastes, and water quality; cultural services that provide recreational, aesthetic, and spiritual benefits; and supporting services such as soil formation, photosynthesis, and nutrient cycling” (Millenium Ecosystem Assessment, 2005). Ecosystems services are performed over vastly different spatial and temporal scales. Global agricultural systems are highly dependent on ecosystems services provided by soil (Doran & Zeiss, 2000; Haygarth & Ritz, 2009).

Modern day agricultural practices are influenced by crop and management research that occurred during the late 1950s to 1970s commonly known as the Green Revolution. Significant improvements in yields of important crops (such as rice and wheat) were produced through breeding programmes (Evenson & Gollin, 2003). The successes of breeding programmes were also coupled with intensive use of fertilisers, pesticides, and increased water supplies (Tilman et al., 2002). This led to yields of cereal crops tripling. Food was more accessible for less affluent people and helped alleviate starvation. Increases occurred in a period where global human populations underwent increases over 100%, but most impressively, this was achieved in approximately 1/3 more land area than previous agricultural systems (Mahon et al., 2017; Pingali, 2012).

The consequences of intensive agriculture have been laid bare in the following decades (Kopittke et al., 2019). This approach to food production and the associated political forces that fuelled the transition to an agricultural system that favoured highly productive systems (Prăvălie et al., 2021). However, intensive approaches towards production have degraded soils have and reduced their capacity to provide important ecosystem services such as carbon sequestration and biodiversity (Baude et al., 2019; Kopittke et al., 2019). Soil erosion is often highlighted as a significant driver of soil degradation, with agricultural practices such as the maintenance of monocultures linked to 35 Pg (±10) of sediment mobilised annually (Quinton et al., 2010). Furthermore, land use change from natural systems to low diversity systems has also affected biogeochemical cycles that occur in soils, particularly where functional groups of organisms are excluded (Bardgett et al., 2005; Brussaard, 1997; Wardle et al., 2002). Agrochemicals such as fertilisers, pesticides, and herbicides are supplied to sustain productivity in low diversity systems. In natural systems biodiversity is key driver of ecosystem function (Smith et al., 2015). Changes in biodiversity and their effects on ecosystem function have been reported across many studies (Jochum et al., 2020; Oliver et al., 2015). Alterations in soil food webs, such as reductions in fungal biomass have been linked to decreased resistance to environmental perturbations (de Vries et al., 2018), and changes in earthworm diversity appear to alter carbon cycling (Zhang et al., 2021). Plant diversity and its effects on ecosystem function have received extensive research attention (Grime, 1998; Isbell et al., 2011; Wardle, 2016).

1.2 Biodiversity and soil health

Plant community composition and the effects on soil ecosystem processes have been studied at a broad range of scales from microcosm to plot/field scale and from singular growing seasons to decades (Wardle, 2016).

Plant diversity driving increasing Net primary productivity (NPP) has been reported in many studies (Balvanera et al., 2006; Cardinale et al., 2006) with suggestions benefits may extended to belowground systems. However, these effects of plant diversity are debated, with some authors suggesting species identity is a greater driver of productivity or function than species richness. (Hooper et al., 2005). Long term studies such as at Cedar Creek (Tilman et al., 2001) and JENA (Scherber et al., 2010) highlight multiple benefits to ecosystem functions, with benefits increasing overtime (Isbell et al., 2017) Improvements in multiple soil measurements related to biogeochemical cycling improved along a scale of increasing plant diversity in manipulated, unfertilised plots (Tilman et al., 2001). Carbon and Nitrogen, along with several other major nutrients increased along with plant productivity. These studies, along with other long- and short-term studies reported a virtuous, positive feedback cycle of which plant diversity improved soil and subsequent plant productivity (Fornara & Tilman, 2008). However, the applicability of these experiments to non-artificially assembled systems remains open to questions (Hooper et al., 2005; Wardle, 2016).

More recent literature has attempted to relate the results of controlled Biodiversity Ecosystem Function (BEF) experiments with studies of realistic systems (Jochum et al., 2020). For example, in natural systems, increased biodiversity has been linked to some improvements in ecosystem functionality, but effects are limited belowground. This is evidenced in a review (van der Plas, 2019) of 258 published studies exploring biodiversity and ecosystem function in naturally assembled communities, soil carbon storage was not found to relate to biodiversity. Moreover, abiotic variables and functional groups were perceived to have stronger effects on ecosystem functions compared to biodiversity. Despite this, information about the effects of biodiversity on soil ecosystem function is still limited in some systems (Lehmann et al., 2020). Thus, investigating the impacts of plant diversity on soil function across a range of managed systems remains important. Intensively managed systems often manipulate pH and nutrient stocks and are associated with reduced soil ecosystem functions and poor plant diversity

(Kibblewhite et al., 2008). Changes towards less intensive management to sustain particular plant traits is seen as a way to improve soil function in agroecosystems (Abalos et al., 2018; Sharma et al., 2018). Plant traits can be linked to different soil processes and can be targeted to supply desired ecosystem services.

1.3 Plant-soil interactions provide ecosystem services

Soils support and regulate many services including food production, climate and hydrological regulation, nutrient cycling, and is important to industry and culture (Kopittke et al., 2019). These factors are explored in below sections.

1.3.1 Nutrient cycling: Nitrogen and phosphorus

Soils cycle major plant nutrients including N, P, and K (Ghaley et al., 2014). The Earth’s atmosphere is 79% N_2; however, a minute proportion of this is readily available to plants in inorganic forms (NH_4+ and NO_3-). Recent studies have revealed plants may also utilise organic N in simple forms, such as amino acids, to more complex forms like proteins (Moreau et al., 2019). The ecological importance of organic N sources has been considered and remains open to debate. Studies have shown that inorganic N represents over 90% of N uptake in plants compared to Organic N along a gradient on N enrichment (Moreau et al., 2019). Inorganic N is produced in soils from the microbial and enzymatic breakdown of organic N, contained in soil organic matter (SOM). Organic forms of N undergo mineralization in two steps (aminization and ammonification) to produce soluble forms. Inorganic N is also immobilized through conversion to organic N (Cai et al., 2017). A wealth of plant traits can impact N cycling in agroecosystems. Fast growing (acquisitive) plants with high rates of N uptake have exhibited positive effects of N cycling and reducing N losses from soils. These traits include high leaf N content, high specific leaf area, and low leaf dry matter content (Abalos et al., 2018). Similarities in below ground traits such as high rates of N uptake, high N content, and explorative root systems are hypothesised to exert benefits to N cycling (Abalos et al., 2019). Plant ecological strategies exist along an axis of conservative-acquisitive traits (Laliberté, 2017). Plants utilising the aforementioned acquisitive traits are likely to provide advantages in agroecosystems to N cycling over species with more conservative traits, due to their rapid uptake of inorganic N and associated mitigation of N leaching and N_2O emissions (Abalos et al., 2019). Whilst acquisitive species can be linked to faster N turnover and potential losses N from litter compared to more conservative species, these losses are likely to be unsubstantial compared potential inorganic N losses from conservative species under high N application scenarios (Oelmann et al., 2015). Optimal plant communities in agroecosystems for N cycling would likely benefit from a high ratio of acquisitive: conservative species (Abalos et al., 2019). However, identifying species with acquisitive traits is difficult. Above and below ground traits are not consistently correlated. Additionally, root traits are plastic to abiotic and biotic variables (Gao et al., 2023; Kumar et al., 2019; Ryser & Eek, 2000).

More empirical evidence exists for the role of root exudates in nutrient cycling and availability. Mineralisation of SOM can be enhanced by the release of labile carbon rich root exudates to prime the rhizosphere (Bardgett et al., 2014; Moreau et al., 2019). The resultant combination of increased microbial biomass and invigorated soil borne microbial enzymes improve rates of organic N to inorganic N conversion and subsequent plant uptake. Immobilisation of inorganic N by microbes may increase under certain conditions produced under root exudation. Research indicates that the release of root exudates may be dynamic and exudate composition may be altered to influence N mineralisation-immobilisation (Bardgett et al., 2014). For example Liu et al found the addition of carbohydrates or organic acids increased or decreased N fixation relative to mineralisation respectively. Another study examining root exudates of *Panicum virgatum* found total root C exudation doubled under high N conditions compared to low N, indicating soil N status may influence root exudate profiles (Smercina et al., 2021). Increased C content in root exudates under high N conditions can also be linked to increased aboveground productivity and photosynthetic activity (Coskun et al., 2017)

Phosphorus is typically supplied by soils through the breakdown of minerals in weathering processes or by natural acids produced by microorganisms and plants (Menzies, 2015). Plant traits pertaining to phosphorus uptake and mobilisation have been identified.

Similarly to root traits that influence N cycling, P availability in soil is affected by root exudate composition (Pang et al., 2018). Organic acids are exuded into the rhizosphere to desorb or solubilise mineral associated P and C inputs from roots enhance microbially driven SOM breakdown (Li et al., 2007). Many legumes readily mobilise soil organic P due to its importance in N fixation (among other physiological processes). Other non-leguminous species have also been associated with P mobilisation (Dechassa et al., 2003). Phosphorus uptake can be improved in plant communities by the presence of species with P mobilising effects. A study exhibited how some plant species facilitate P uptake for themselves and other non-P mobilising species, suggesting a complementarity effect. Several species were found to mobilise P, but concomitant uptake by non-P mobilising species only occurred where root traits were sufficiently plastic to take advantage (Yu et al., 2020)

Agroecosystems typically apply fertilisers that can supply a readily available source of P. Changes in management practices to reduce fertiliser reliance may target facilitating species in rotations or to complement crops to exploit large sources immobile P that are often present in soils associated with prior fertiliser application. *Trifolium pratense* has been recognised for its capacity to mobilise and uptake P in combination with *Lolium perenne* (Timmermans & van Eekeren, 2016). Studies exploring the effects of plant diversity on phosphorus cycling have found enhanced phosphorus cycling under more diverse covers. One grassland study found legumes exploited inorganic phosphorus more than other functional groups leading to increased phosphorus in aboveground biomass and reduced soil inorganic phosphorus. However, plant available sources of phosphorus in soil increased once legume residues were deposited. Moreover, P cycling was shown to be enhanced under more diverse plant communities independent of the presence of functional groups (Oelmann et al., 2011) .These studies highlight the role of species identity and species richness effects in diverse plant communities and their influence on P cycling.

1.3.2 Carbon cycling

Soils store significant quantities of carbon in terrestrial environments. UK soils store an estimated 7380 – 12300 Mt C. Uncertainty in these estimates is due to spatial variability in soils such as depth or soil texture and the limited ability to measure soil heterogeneity over large scales. As a result, soils play a role in climate regulation (Smith et al., 2013). SOM is also vital aspect of soil health. It drives the provision of services such as nutrient availability/cycling, soil structure, water retention and microbial processes (Lal, 2016). The primary driving forces for the formation of SOM are plants. Plants fix carbon from the atmosphere, in the form of CO_2, into biomass. This biomass, after death, is a source of carbon (organic matter) for the soil.

Microbial decomposition releases some of this carbon as CO_2 through respiration, whereas some of the biomass forms humus, a recalcitrant component of soil (Paul, 2016). Although, the concept of decaying biomass forming humic substances is debated and contemporary research indicates organic matter exists as continuum of complex plant material to more simple forms such as carboxylic acids (Lehmann & Kleber, 2015). Furthermore, different components of SOM have been recognised such as minerally associated organic matter and particulate matter, which differ (longer and shorter respectively) in their rate of turnover (Lehmann & Kleber, 2015).

Root exudates from plants are also a source of carbon. Symbiotic relationships with mycorrhizae provide nutrients such as phosphorus in return for carbon laden exudates (Morgan et al., 2005). The overall influence of root exudates on soil carbon sequestration is largely undetermined, but it is likely carbon is released after microbial priming (SOM breakdown) (Bardgett et al., 2014). SOM often dictates the quality of soil as it provides a stream of plant available nutrients and water, amongst many other benefits including microbial activity and enhanced soil aggregate stability (Lehman et al., 2015).

A variety of plant traits dictate the sequestration of carbon (Fornara & Tilman, 2008). Using plant communities with complementary traits can improve productivity and increase biomass inputs, such as roots, to improve carbon sequestration in agroecosystems. Recent studies have indicated root turnover contributes significantly to SOC pools but accrual is negligible due to priming effects, whereas shoot material contributed less but lead to accrual (Huang et al., 2021). Studies have shown the inclusion of legume species like *Trifolium pratense* in biodiversity restoration experiments can enhance carbon sequestration potential due to high levels of productivity, although it's dominance decreased with increased species richness. (Hector et al., 2002). This highlights the importance of biodiversity but also the impacts of functional groups or even specific species on soil functions such as carbon sequestration.

Agroecosystems are susceptible to diminished soil carbon stores/SOM due to biomass removal, soil management techniques such as tilling and soil erosion. Compaction, through processes such as trafficking, may reduce carbon inputs through decreased plant productivity, but carbon loss may be lower due to reduced aeration and microbial activity (Komatsuzaki & Ohta, 2007). Reduced plant cover following cultivation or application of herbicides can lead to a loss of SOM due to rates breakdown exceeding input.

1.3.3 Physical structure

Plant interactions with soil are a determinant factor of aggregation and stability. Soils are dependent on these interactions to perform ecosystem processes and services (Faucon et al., 2017). Certain root traits are effective at reducing soil erosion. Root biomass is generally related with reduced soil erosion, but more specific aspects of root systems aid formation of soil aggregates and stabilisation of soils. Root architectural traits, such as high degrees of dispersion and branching, and morphological traits, such as low root diameter, benefit soil aggregation. These traits are associated with plants with fibrous rooting systems, such as *Lolium perenne,* which display high root surface to soil contact area, and enhance soil aggregate stability relative to species with taproot systems (Gould et al., 2016). Soil stabilisation is linked with tenacious, long lived roots (Rillig et al., 2015). Root exudation has a significant influence of aggregation and stability. Secretions of polysaccharides and proteins have adhesive properties which cause soil particles to aggregate. The tendency of root exudates to from hydrophobic films on soil aggregates also improves aggregation and stability. Associated mycorrhizae hyphae and compounds also improve these aspects (Bardgett et al., 2014). A 2016 study by Gould et al found plant diversity increased aggregate stability at glasshouse and field scales. Additionally, specific species were found to alter other physical measures of soil such as soil strength and hydrology.

Importantly, soil regulates, filters, and stores water. These ecosystems services are particularly important to human health (Haygarth & Ritz, 2009). Water enters soils through infiltration and is stored a vastly different temporal scale. Water is removed from soil through extractions by plants, evaporation or drainage (Haygarth & Ritz, 2009; Warren et al., 2015). Plant diversity may alter soil water balance. Increased productivity related to diverse plant communities may lead to greater transpiration and loss of soil moisture (Chen & Coughenour, 2004; O'Keefe et al., 2019). One study in an experimental grassland (JENA) found different functional groups effected water balance in different ways. Grasses reduced water content in topsoil, whereas tall herbs reduced water content in subsoil. This was prescribed to differences in their rooting traits with grasses having shallow, fibrous roots and tall herbs having deeper tap roots (Leimer et al., 2014).

1.4 Soil health and systems under stress

Climate change is considered to be a driver of many stressors. Progressive warming, likely driven by anthropogenic greenhouse gas emissions, has impacted natural systems. Climate change presents many problems across spatial and temporal ranges. Existing threats to food security are exacerbated (Brown & Funk, 2008; Dai, 2013; Hertel, 2016; Tai et al., 2014).

Drought occurs when cumulatively low periods of rainfall occur. The onset of drought can also be influenced high temperatures (Burke et al., 2010). Studies predict increasing drought in the future (Dai, 2013).Drought and heat stress pose significant threats to crops and food security. Crop physiology and morphology are negatively altered and this leads to significant reductions in crop yields (Fahad et al., 2017). Soil moisture drought extent and severity is projected to increase if 2°C warming scenarios are surpassed. A 40% larger area may experience soil moisture drought for up to three times longer in Europe under 3°C warming compared to a 1.5°C warming scenario (Samaniego et al., 2018). Drought already represents the leading factor influencing reduced crop yields worldwide (Bodner et al., 2015a).

Soil carbon cycling is altered during periods of drought. Inputs of C through root exudates are reduced and productivity declines, but processes of carbon loss driven by microbial activity are also lower. The balance between soil carbon gain and loss can also be altered through rewetting processes, where large stores of carbon can be relinquished from soils through the Birch Effect (see chapter 4) (Bloor & Bardgett, 2012; S. Zhang et al., 2020).

N cycling is modified by drought. Mineralisation is reduced, but inorganic N accumulates due to low soil moisture. These effects on mineralisation and inorganic N accumulation strengthen with drought intensity. Upon rewetting, mineralisation increases rapidly, and inorganic sources of N are remobilised potentially leading to large losses of N through leaching (Bloor & Bardgett, 2012; de Vries et al., 2016).

Drought in soils has been linked to worsened soil structure. Contraction of soil particles under low soil moisture content has been linked to breakdown of soil aggregates. The breakdown of soil aggregates has implications for soil carbon, where organic matter previously occluded becomes vulnerable to mineralisation by microbes, particularly following rewetting (Quintana et al., 2023).

1.5 Orchards

An intensive approach to food production, driven by a changing global diet, fuelled the transition to an agricultural system that favoured highly productive arable systems. Five cereal crops outstrip fruit production by approximately three times (over millions of tonnes) in production globally (FAO, 2023). Some temperate fruit crops have also expanded significantly during this time, while others have only increased marginally (FAO, 2023) Global apple production has increased over this time, although the rate of increase has been relatively modest with only 50% increase between 2000 and 2022 (FAO, 2023.). However, apples take up notable portion of global fruit production, and are recognised as an important source of nutrition in diets (Hyson, 2011). Whilst production has increased, the area used to grow apples has not increased (FAO, 2023). This is caused by increased productivity per ha from adopting more modern management techniques such as fertigation and newer varieties of apple.

The UK has a long history of apple and pear production with evidence suggesting it may date back to 450 AD, with apple tree husbandry described in literature several centuries later (Veen et al., 2013). Approximately 3000 varieties of apples have been grown across British history, with the vast majority being grown in extensive traditional orchards. In the modern day 2200 of these apple varieties and 550 pear varieties exist in gene stock at The National Fruit Collection Larger scale orchards were introduced in mid to late 19th century with large areas maintained up to 1950. The entry of the UK into the European Economic Community (EEC) exposed consumers to foreign markets and changes in farming subsidies instigated by the Common Agricultural Policy (CAP) (Stoate et al., 2009).

This combination of factors led to a significant reduction in orchard area in the modern day compared to 1940, although with yields (tonnes/ha) increasing rapidly (see figure 1). Remaining orchard systems became more intensified and their capacity to supply ecosystem services, other fruit production, declined (Samnegård et al., 2019).

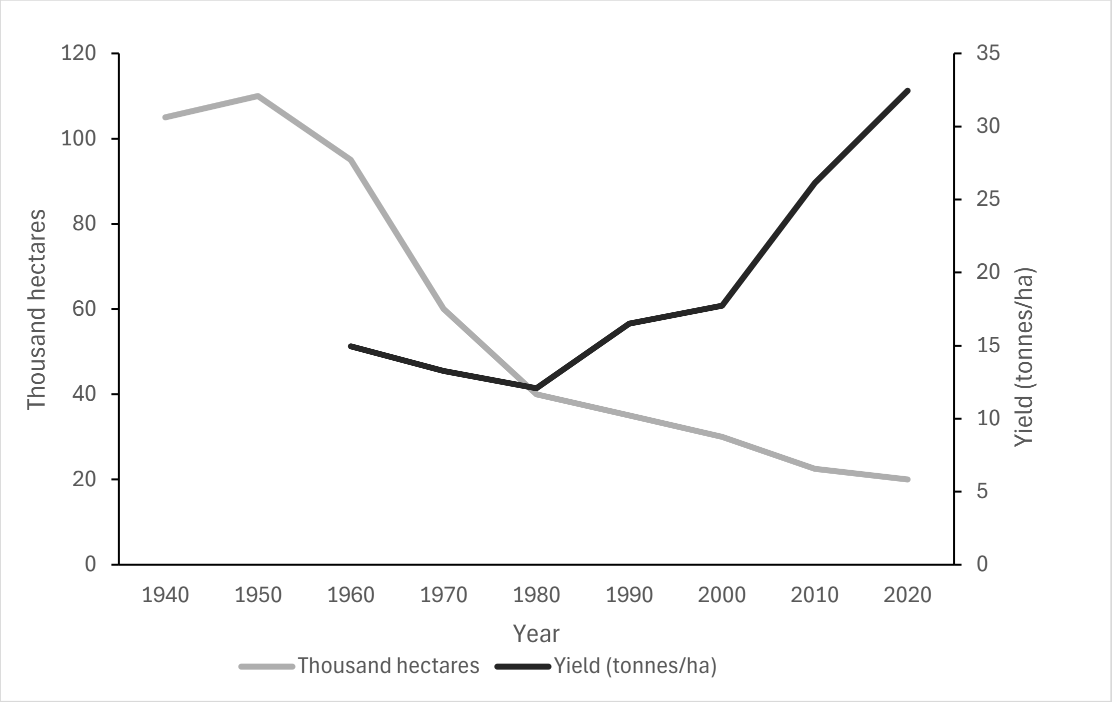

Figure 1.1: UK commercial orchard decline from 1940 -2020. Data source: agCensus (https://agcensus.edina.ac.uk/) for Thousand hectares and FAOstat for Yield (tonnes/ha).

Contemporary approaches and advancements in research have highlighted the importance of agricultural systems not only in their role to supply food but their role in supplying ecosystem services (Isbell et al., 2017). Multiple ecosystem services provided by orchards have recently been acknowledged (Demestihas et al., 2017). They are driven by similar ecosystem processes in other agroecosystem and natural environments. Many of the necessary processes are directly or indirectly mediated by plants, particularly plant-soil interactions (Faucon et al., 2017). Ecosystem services such as carbon sequestration, soil nutrient (N) cycling, and pathogen and pest regulation may be enhanced through concepts of ecological intensification employed in other agroecosystems (Demestihas et al., 2017).

Existing research in orchards has been conducted to evaluate the potential of wildflower strips to boost pollination and pest regulation ecosystem services, with limited focus on other ecosystem services (Campbell et al., 2017; Montanaro et al., 2017). Such studies have found mixed results. Some studies have found management interventions like wildflower stirps or margins can enhance pollinator visitation which may have benefits on fruit yield and quality (Campbell et al., 2017). Others found improvements in natural enemy ecosystem services which reduced populations of undesirable invertebrates such as *Dysaphis plantaginea* and increases in beneficial invertebrates such as *Anthocoridae sp* (Cahenzli et al., 2019; Herz et al., 2019). Studies where no benefits on pollinator and natural enemy services were detected have suggested use of plant protection products such as pesticides can mask benefits associated with wildflower strips (McKerchar et al., 2020). Establishment of wildflower strips represents an additional initial cost to growers which may be prohibitive (Carvell et al., 2022). Understanding the full range of potential benefits provided by wildflower strips to orchard ecosystem services may make uptake more desirable.

Climate change offers broad and discrete challenges to orchards in the future. Many challenges faced by other systems apply to orchards such as water supply and disease prevalence (Gitea et al., 2019). More specific challenges relating to extreme weather such as dormancy periods, frost tolerance and heat resilience are tackled through breeding programmes (Kaukoranta et al., 2010) and variety diversification (with loss of traditional varieties for more commercial varieties) (Gitea et al., 2019). Studies exploring temperate orchard soils under climate change are more limited (Demestihas et al., 2017a).

A study exploring the response of apple orchard soils to increased temperature and elevated CO_2 found limited impacts of soil microbiome diversity under conventional management, but a more pronounced effect on soil microbiomes under organic management. It suggests agrochemical application in the conventional sites had afforded greater resistance to abiotic perturbations. Although, inherent differences between sites such as soil texture may have affected results. This was highlighted by a reverse in the typical trend in bacterial alpha diversity, a measure of localised microbial diversity, with the conventional site showing more diversity compared to organic management (Cook., et al., 2023). Furthermore, studies predict increasing proportions of soilborne pathogens under climate change this may raise concerns for the management of replant disease or the establishment of pathogen populations during an orchard’s lifetime. interventions to manage pathogen populations during production may become more salient (Cook et al., 2023; Pechan et al., 2023)

Adoption of modern soil management techniques such as dripline irrigation can be used to ameliorate soil moisture deficits provided a water supply is readily available (Manning., 2021; Ngai et al., 2020). However, regions such as Kent and Medway, the largest top fruit producing region in the UK, is water limited and likely to become even more restricted in the future (Ngai et al., 2020). Alleyway vegetation sequestrates a relatively small but important fraction of C in orchards. Whilst management tools are available to sustain fruit production, irrigation would not be economically for alleyway cover (Midwood et al., 2020). This highlights the question of C sequestration under elevated CO_2 and temperature, and possible drought. A study in temperate grassland (Dietzen et al., 2019) indicated C sequestration is resistant or even benefitted by these conditions. Whilst contrary results have been found in modelling approaches at broader scales (Crowther et al., 2016). Further study is required to understand the effects of drought on intensively managed apple and pear orchards (see chapter 4).

1.6 Thesis aims, objectives, and structure

This thesis aims to investigate the effects of orchard management, particularly utilising wildflower strips, on soil health. Research was performed in collaboration with World Wide Fruit and growers. This allowed access to commercial orchard sites, including sites planted with wildflower strips. Four main thesis objectives are:

1. To review the impacts of intensive orchard management practices on soil health.
2. To benchmark soil health in UK orchards where previously established wildflower strips were planted compared to conventional orchard alleyways.
3. To investigate the impact of wildflower strips on orchard soil drought resistance and resilience.
4. To compare the effects of representative orchard alleyway ground covers on soil health in mesocosms.

To address these questions four chapters were formulated. Following the introductory literature review, Chapter 2 provides a review of orchard management and the impacts of management techniques on soil health. Three stages of management were considered: Establishment, active management, and tree removal (grubbing).

Chapter 3 aimed to benchmark soil health in commercial orchards planted with wildflower strips for a previous study on pollinator and natural enemy services (McKerchar, 2016). This allowed us to collect soils from seven sites where wildflower strips have been in place for seven years. Chemical, physical and biological indicators of soil health were measured.

Research question 3 was targeted in chapter 4 where I performed a microcosm incubation to assess drought resistance and resilience of orchard soils. Soil cores were collected from two orchard sites planted with and without wildflower strips. Soil cores were exposed to one round of drought and rewetting with CO_2 measured at four time points. This approach allowed us to measure soil respiration as a proxy of soil function under and following drought stress.

A mesocosm experiment was conducted to study the effects of four different ground covers on soil health outside of orchard management perturbations. Ground covers were representative of communities observed in field from chapters 3 and 4. Covers were maintained in mesocosms for one year, with clippings from the first growth season used for an additional decomposition assay lab study which monitored litter mass loss over six months.

2. Reviewing the effects of orchard management on soil heath

2.1 Introduction

Apple and pear orchards represent an important element of the UK food production system. Whilst occupying a relatively modest land area in comparison to other crops (see chapter 1), they provide a range of important ecosystem services, a number of which are driven by soil processes (Demestihas et al., 2017a). Despite the broadly accepted functions outside of food production, little focus exists on the provision of soil ecosystem services and how conventional orchard management affects them. Services including climate regulation, nutrient cycling, water storage and cycling, and biodiversity are provided by orchard soils (Demestihas et al., 2017a). However, soils that provide these services can be degraded by intensive management strategies adopted to promote high yields (Pereira et al., 2018). The deleterious effects on soil health from these intensive practices covers three axes of soil health: biological, chemical, and physical. Use of agrochemical inputs such as insecticides and fertilisers has been linked to reductions in soil biodiversity and increased nutrient leaching (Smith et al., 2016; Wen et al., 2020). The same practices associated with agrochemical input can lead to compaction from trafficking which can compromise soil structure. The impacts of intensive practices can also vary spatially and temporally. For instance, routine herbicide application in tree rows can negatively affect earthworm populations and diversity for the duration of fruit producing years (15-30 years) (Gaupp-Berghausen et al., 2015), whereas the acute use of soil fumigants may affect a broad range of microbes over a shorter time (months) before recovery (Fang et al., 2018). Identifying the scope and impact of orchard management practices is therefore critical to sustain ecosystem services and improve soil health.

The aim of this literature review is to explore the effects of orchard soil management on soil health. Current best practice guides on orchard management neglect soil health, primarily focusing on tree health. Literature covered will highlight areas where current practices degrade soils, and where new opportunities exist to enhance soil health. Three stages of a commercial orchard's lifetime are explored. The initial establishment stage, active management during fruit production, and once trees are removed (grubbed). Following outlining key areas where management can be adapted or new techniques can be adopted in these three stages of orchard management, statements on orchard management to enhance soil health will be provided.

Inherent characteristics of soil such as texture can largely determine baseline soil health characteristics (Amsili et al., 2021). For example, soils comprised of larger particles typically drain well, an important factor to consider when choosing an orchard site. However, soils that are too coarse (sandy soils) retain lower levels of soil organic matter and may not provide a suitable nutrient supply (Chaudhari et al., 2013). Other factors such as land use history can significantly affect soil health (Hazarika et al., 2014; Pulleman et al., 2000; Turley et al., 2020). All elements of soil health, physical, chemical, and biological must be considered when establishing and managing an orchard site to optimise fruit production and tree health (Dupont et al., 2020).

2.2 Pre-establishment

2.2.1 Topography

Topography can dictate several properties of an orchard in relation to its management. Orchards are commonly planted on gradual slopes to allow drainage of cold air to reduce frost damage (FAO, 2005). Bare soil under trees (herbicide strips) also stores heat more effectively for a similar effect. However, this management can have consequences for orchard soils. Bare soil on slopes is vulnerable to soil erosion, although this is marginally mitigated by tree roots. A large body of research exists from Mediterranean orchards where soil erosion poses a significant threat to production (García-Ruiz, 2010; García-Ruiz et al., 2013; Keesstra et al., 2016). Whilst some parallels can be drawn between temperate and Mediterranean orchard management the far more liberal application of herbicides and lower vegetation cover in the latter makes comparisons tenuous (Demestihas et al., 2017b; Gómez et al., 2003)

A study investigating land use and topography indicated soil nutrients increased from the top of orchard slopes to the bottom, indicating topsoil erosion. The reverse of this trend was observed in grassland, shrubland and woodland with more nutrients accumulating at the top of slopes due to increased cover from vegetation stabilising soil (Zhou et al., 2023). Approaches such as mulching could be adopted to reduce soil erosion. Mulching has been shown to reduce soil erosion on slopes with bare soil and may also provide benefits beyond erosion mitigation (Donjadee & Tingsanchali, 2016). Mulching is discussed in detail in chapter 2.

2.2.2 Soil treatment (liming and fumigation)

Targeting and maintaining a suitable soil pH in an orchard is essential component of commercial orchard management. Soils in the range of 6 - 6.5 pH provide the optimal conditions for important nutrient uptake (such as calcium, magnesium, and phosphorus) and supports root growth and microbial communities (AHDB, 2020). Soil pH may become spatially variable in orchards over time, especially in systems where nitrogen is supplied by fertigation (Haynes & Swift, 1987).

Fertigation involves the delivery of fertiliser in irrigation water where it is delivered in the tree root zone allowing improved nitrogen use efficiency. Nitrification of ammonium to nitrate where nitrogen fertilisers are applied can reduce pH over time and lead to sub-optimal conditions for tree/fruit growth. Monitoring pH and adjusting it is important for sustained fruit production and soil health.

The use of pre-plant fumigants containing active ingredients such as Dazomet are commonly used where there is risk of apple replant disease (Cook, Magan, & Xu, 2023). Some fumigants are injected into soils as liquids or gases, but more modern fumigants can be spread as microgranules. Treatment of soils with fumigants can alter soil microbial communities to a disease suppressive state (Jiang et al., 2022). However, whilst this type of treatment is effective, it is non-selective and reduces overall microbial diversity, which may impact soil processes. Studies exploring the effects of microbial diversity on ecosystem function have indicated that while there is a significant degree of functional redundancy concerning soil processes, responses to perturbations may be altered (Griffiths et al., 2000) . A study exploring the effect of fumigation on soil microbial makeup using nematodes diversity as a proxy found whilst bacteria feeding nematodes did not decrease (but did change in composition), fungal feeding nematode numbers decreased. This indicated a shift towards a more bacterially dominated food web (Sánchez-Moreno et al., 2010). This has important consequences for ecosystem function, for instance bacterial dominated food webs have been found to be less resistant to drought when compared to fungal based soil food webs (de Vries et al., 2018). As discussed in chapter 4 drought may threaten orchards in the future due to the concentration of commercial UK orchards in drought threatened regions under climate change.

Other management strategies are used to prepare orchard soils for replanting. Some studies have highlighted simple strategies such as planting trees where drive rows/alleyways were previously present and opting for genetically dissimilar rootstocks. Such spatial differences in soil microbial communities may be sufficient to avoid disease incidence in trees (Cook, Magan, & Xu, 2023). Other strategies include application of organic amendments such as manure and cover crops during a fallow period prior to planting. Organic amendments such as manure (Braun et al., 2010) and Brassicaceous seed meals (Mazzola & Brown, 2010) have proven to be a suitable replacement for fumigation when tree growth and disease control are considered. One study exploring the effects of cover crops as pre-treatment for orchard replanting found a single round of cover crops, leading to increased plant diversity from an initial monoculture, could be sufficient to alter microbial communities. Furthermore, it was suggested specific cover crops such as wheat and barley should be avoided due to their association with *Pythium* spp., which is associated with growth limitations in apple trees (Manici et al., 2015).

2.2.3 Vegetation management

Orchard sites may undergo preparation through the removal of weeds, or grass seed can be sown to establish a cover to outcompete weeds once trees are planted. Sometimes a site is scoured at the surface to completely remove surface vegetation during this process (Atucha et al., 2011).

More diverse vegetation in orchard alleyways in the form of wildflower strips is being adopted by some growers. Establishing and maintaining wildflower strips at meaningful diversity densities has proved problematic. A 2022 study by Carvell *et al* described a method to establish wildflower strips. The method is primarily focused on reducing competition from weeds on sown species. This is managed by an initial broad spectrum herbicide spray on the seedbed, regular mowing in the first year of establishment with removal of clippings (shading reduction), and spot application of selective herbicides where necessary. Among other methods described in the paper, this approach yielded 50-80% coverage sown herb species after three years. Further papers have highlighted important factors affecting establishment such as shading (Schmidt et al., 2022), which can be affected by orchard orientation and overall agronomic interventions such as reseeding to be important (Benvenuti, 2023).

2.2.4 Compaction

Soil compaction can be common in orchard systems due to the nature of management practices involving machinery and very limited or no tillage. Several deleterious effects, such as reduced root growth in plants and decreased aeration in soils, occur after compaction thresholds (approx. 1000Kpa). Soil structure is altered leading to an increase in bulk density and the breakdown of soil aggregates. Soil porosity is also reduced with negative impacts on water infiltration and water holding capacity. Water and gas exchange may be impeded by a reductions or destruction of structural channels within soil and the atmosphere respectively (Nawaz et al., 2013). Additionally, Soil aggregate size has been shown to influence the diversity of soil bacteria and fungi, with different aggregate sizes providing different niches (Zheng et al., 2021). Changes in bacterial and fungal diversity may impact soil functionality under normal conditions but effects can be amplified under perturbation where less functional redundancy may persist (Longepierre et al., 2021; Philippot et al., 2021).

Tree health can be negatively impacted in several ways by soil compaction; development, root architecture, and productivity are constricted (Ferree et al., 2004). Trees have been shown to alter their hormonal regulation in the presence of insufficient plant root–atmosphere gas exchange under compacted soil with reduced pore space. (Habibi et al., 2023). Reduced uptake of water and nutrients effect water states and hinders photosynthesis, harming overall productivity and reducing yield (Kozlowski, 1999). Compaction in orchards varies spatially with a significant proportion occurring at the interface of the herbicide free strip and the vegetated alleyway. Wheel compaction forced vegetated strip roots deeper and increased competition with naturally deeper rootstocks of trees (Whalley et al, 1995).

Compaction during the pre-establishment phase can be combatted with deep ripping, where soils are mechanically broken apart. Such management interventions have implications on soil properties and lead to distinct spatial variation in orchards. A 2012 study characterised spatial variability at an orchard site where deep ripping was used as a management intervention at establishment. Tree line soil was found to have a different distribution of coarse fractions and significantly lower total organic carbon and electrical conductivity when compared to alleyway soil (Umali et al., 2012). This highlights how establishment can significantly impact spatial variability in orchard soils and should be considered when designing soil monitoring strategies.

Figure 2.1. Evidence of compaction caused by machinery at two Kent orchard sites.

2.3 Active system management

2.3.1 Water state and supplementation

Rain fed orchards are typical of a more traditional approach, but this approach is uncommon in new orchards where dripline irrigation is preferred (Manning, 2021). Precision application of water may provide benefits for maximising yields and reducing tree water stress. However, irrigation strategies may have implications for soil health. A meta-analysis exploring differences in SOC in irrigated systems identified contrasting patterns of SOC storage across soil depths. Overall irrigation was determined to beneficial to SOC storage, although this outcome may have been heavily influenced by gains in arid regions. Drip irrigated systems had less beneficial effects on SOC storage compared to other methods (sprinkler and flood) but was the only applicable method in this context. The difference in SOC storage is likely linked to reduced irrigation volumes supplied by drip irrigation (Emde et al., 2021). Another study comparing rainfed vs irrigated systems in non-orchard sites found no significant differences in SOC storage (Dal Ferro et al., 2023). However, carbon sequestration potential in arable systems is typically lower than orchards (Natural England, 2021).

2.3.2 Agrochemical application (herbicides, insecticides, fertiliser)

A range of agrochemicals are used in commercial orchards however, the temporality of application and the method of delivery can vary substantially (Manning, 2021; Srivastava & Malhotra, 2017). Glyphosate is a herbicide commonly used in orchard systems to maintain a section of bare ground around the base of trees. Usage has received criticism due to risks associated with human health and ecosystems (Finger et al., 2023). Studies monitoring life cycles and the behaviour of earthworms have indicated glyphosate has negative effects on reproductive success and cast formation of some earth worm species. In a mesocosm experiment where a glyphosate-based herbicide was applied and compared to control (herbicide free) mesocosms surface casting was shown to decrease nearly by half and then almost cease after three weeks in vertical burrowing anecic earthworms, albeit with an immediate short-lived increase directly after herbicide application. Mesocosms were supplied fresh shredded vegetation on the surface of the mesocosms so a lack of litter could not be attributed for the decline cast production and points towards a direct effect on the organisms (Gaupp-Berghausen et al., 2015). The same study highlighted potential alterations to soil ecosystem processes with increased soil concentrations of nitrate and phosphate which may have implications for fruit production and tree health management (Toselli et al., 2000). Furthermore, it may encourage leaching which can be detrimental to hydrological/water systems (Cui et al., 2020), and despite lower cast formation in herbicide exposed mesocosms water infiltration was greater (Gaupp-Berghausen et al., 2015)

Another study evidenced the effects of a glyphosate-based herbicide on earthworm survival and cocoons with the severity varying across species (Stellin et al., 2018). Worms collected from sites that had not previously been exposed to glyphosate showed lower survival rates than previously exposed worms indicating some developed resistance. Although this resistance still resulted in significant mortality.

There is a possibility of glyphosate bans (Finger et al., 2023; Meftaul et al., 2020). Tillage to remove weeds can be effective but this raises problems around the structure of topsoil and potential damage to tree roots. Other solutions include burning, and the use of other chemical treatments that may be unattractive to growers (Matousek et al., 2022). This raises the questions of the impacts of not having a strip of bare ground and how it effects soil health and apple production. Many short-term studies indicate competing vegetation has negative effects on tree growth and resultant yields. However, few studies consider longer time scales (10+ years). A 16 year study found whilst trees competing with sod were initially stifled, they were not significantly different in tree growth or fruit yield in the concluding 10 years when compared to weed free plots (Atucha et al., 2011). These results indicate trees may adapt to sod competition for water and nutrients or soil conditions may degrade over time without cover. The same study suggests applications of post emergence herbicide during the growing season and then recovery of sod during dormancy to promote soil health in tree rows could be optimal for sustained orchard productivity.

Insecticides have also been linked with deleterious effects for soil fauna and associated soil processes. A review concluded over 70% studies monitoring the effects of pesticides showed negative impacts upon soil invertebrates, with insecticides being comparatively more harmful. Gaps in understanding around the effects of mixed application of pesticides were also identified (Gunstone et al., 2021). Integrated pest management (IPM) may represent a viable alternative to conventional applications when considered alongside soil health. A study monitoring soil health in an IPM orchard, an organic orchard and semi-natural grassland found similar soil health function across all sites. However, further comparisons in terms of productivity and soil health to high input conventional sites would be necessary to validate an IPM approach (Daelemans et al., 2022).

Fertilisers are another agrochemical used in orchards to maximise fruit yield and tree health. However, application rates can frequently exceed the necessary N supply and have deleterious effects. Nitrogen application varies throughout the year in orchards depending on the needs required; too much nitrogen at the fruiting stage can affect fruit storage quality or can encourage excessive vegetative growth rather than fruit production (Toselli et al., 2000). N uptake from common commercial rootstocks, such as M9, are characteristically poor. This has been attributed to the relatively low root biomass density, especially in comparison to grasses (Atkinson et al., 1979). Consequently herbicide strips are maintained to reduce nitrogen resources competition between vegetation and trees. Limiting N resource competition through herbicide application raises concerns around N leaching. Studies exploring approaches to reduce N leaching in orchards have identified high frequency drip fertigation to be the optimal soil-based delivery method of N. Generating a consistent supply of N without large spikes in availability promotes crop N content, nitrogen use efficiency and reduces leaching. The relationship between N application rates and leaching were explored in orchards and it was discovered that N leaching can have a linear or even exponential relationship to N application. The same study also found applying N slightly below the optimal rate can significantly reduce N leaching without compromising yield and had similar benefits to WUE (Cui et al., 2020). Soil texture significantly influences the likelihood of N leaching. More coarse soil textures typically leach N more readily than fine textured soils (Gaines & Gaines, 1994).

2.3.3 Pruning and shredding

Trees are pruned every year to keep trees in their optimal state for fruit production. Clippings are deposited into alleyways where they are pulverised. This is to remove any potentially infected materials away from trees and to accelerate their decomposition to manage disease burden within the orchard (Webber et al., 2022). Pruning residues act as a carbon and nitrogen input for orchard soil. Pruning residues have been found to contribute <10% of total sequestered C to soil. The majority sequestered C was committed to standing tree biomass as this accounts for over 90% of net primary productivity in orchards (Zanotelli et al., 2015). Pruning residues were also found to contribute largely to stable N pools (67-85%) and minimally to mineralised fractions (6-12%) (Tonon et al., 2007).

2.3.4 Mowing intensity

Vegetation strips between trees are commonly comprised of spontaneous cover although this may be influenced heavily during establishment, through sowing sites with grass or alleyways with wildflower mixtures. Cover is often mown frequently to maintain a short sward. This is for several reasons: to reduce establishment of weeds and subsequent life cycles, ease of movement for orchard vehicles and to reduce competition within the treeline and reduce pressure for nutrients and water (Bałuszyńska et al., 2023). However, other management strategies can be used such cover crops or wildflower strips. Different mixtures can be targeted for specific services and may vary depending on soil characteristics such as soils more prone to compaction or to replace certain services that would be provided by agrochemicals such as nitrogen supply (Finney et al., 2017; Isbell et al., 2017). Mowing height and frequency could influence the carbon sequestration potential of alleyway vegetation. Approaches to optimise mowing strategies that encourage root growth, through reduced mowing frequency, could bolster carbon sequestration in orchards (Laihonen et al., 2022), but considerations to extraneous variables, such as trafficking, must be considered (Bondi et al., 2021).

2.3.5 Mulching

Mulching is a common practice in orchards. Typical mulches of organic material such as straw can be applied to improve soil conditions. Other types of mulch include inorganic/plastic mulch around the base of trees, but this is uncommon. Mulching supresses weeds and can also elevate soil temperature and retain soil moisture (Neilsen et al., 1986; Webber et al., 2022) Organic mulches such as compost can be effective at increasing tree row soil C and N but perform poorly for weed suppression and moisture conservation. Straw acts as an effective weed cover and preserves soil moisture well but contributes less to soil C and N (Webber et al., 2022). Alleyway derived 'cover crop' mulching materials can provide similar benefits to traditional mulching materials if applied at an appropriate quantity and rate. This approach could offer a cheaper mulching option over the lifetime of an orchard, but research in this area is currently sparse. Optimal species mixtures and management regimes for such an approach must be developed and applied to a range of orchard systems (Webber et al., 2022).

Application of mulch to bare soils can also act as an effective tool to control soil water erosion of bare soils, such as herbicide strips under apple and pear trees (Prosdocimi et al., 2016). More persistent mulches such as straw may offer superior protection from soil water erosion compared to alleyway derived mulches, which decompose at faster rate. However multiple applications of alleyway derived mulches overtime may allow an accumulation of litter materials which could produce a similar effect to straw (Webber et al., 2022)

2.4 Grubbing and beyond

Commercial orchards typically have lifespans of 15-30 years. When orchard productivity declines or they are no longer economically viable they are grubbed. Grubbing is the process of mechanical removal of trees. During grubbing the entire tree, including roots are removed from the soil. Orchards are placed to sequester carbon differently compared to arable agriculture due to their relative permanence and the nature of the crop (Natural England, 2021) This poses questions about the fate of carbon sequestered during its lifetime and its fate once grubbed. As previously discussed (see Pruning and shredding section), the majority of carbon sequestered in orchard is partitioned in the standing biomass compared to C sequestered in soil during an orchard's lifetime (Zanotelli et al., 2015). A study quantifying soil carbon efflux found a very small fraction (0.01t C ha^{-1}) released, with 95% of which was attributed to loss of CO_2 stored in soil pores (Anthony, 2013). The longer-term effects of grubbing, such as the source/sink potential of previous orchard land allowed to fallow and the following land use, remain uncertain (Natural England, 2021). If used for arable with more destructive management practices, there is potential for significant carbon losses. Whereas a transition to grassland could bolster soil carbon stocks (Ostle et al., 2009)

Orchard land use cover has declined significantly over the last few decades. Grubbing within productive years due to unsustainable return on investments is now common (Manning, L, 2021). Furthermore, replanting orchards in the same area is undesirable due to replant disease and similar concern for viability. This may harm ambitions for increasing carbon sequestration in agricultural land as orchards typically acts as carbon sinks whereas arable land acts as a carbon source on average (Anthony, 2013; Natural England, 2021). Research points toward an agroforestry approach to alleviate this discrepancy between arable and orchard systems (Kay et al., 2019).

There is interest in the generation of biochar from grubbed-up trees to act as a soil amendment (Nematian et al., 2021). Biochar could be applied on existing orchards or added to fallowed areas following tree removal. A modest body of research exists on biochar application in orchards, but further research is required to monitor effects long term. Short term pot and field studies have identified some benefits such as increased SOC (Anthony, 2013; Han et al., 2022). Biochar application can also have a liming effect and could be used to mitigate the acidifying effects associated with fertigation (Bolan et al., 2023). It has also been linked to reduced nitrogen leaching from orchard systems (Hardie et al., 2015). Although, there is uncertainty around biochar and its long-term interactions with carbon mediated soil processes such as soil respiration (Mukherjee & Lal, 2013). Furthermore, studies have found no significant benefits to tree growth or fruit yields (Hardie et al., 2015). The cost of pyrolysis compared to standard burning of tress may also be prohibitive to growers (Anthony, 2013).

A substantial portion of literature examining orchard soils after grubbing is concerned with apple replant disease (Cook, Magan, & Xu, 2023; Liu et al., 2014; Mazzola & Manici, 2012). Ecosystem processes have received little attention in this context. Furthermore, studies considering land use change rarely involve change to orchards rather than from orchards (Deng et al., 2014; Yang et al., 2020). Transition from orchards to other land use and its effects on soil properties and processes could be an important area of research given declining orchard areas in the UK related to economic problems. One study exploring soil erosion and nutrient loss in an area of China where arable land has increasingly transitioned to orchard found benefits to soil conservation in orchard systems. Arable systems experienced 10 times greater soil erosion and contributed disproportionately to organic matter and total nitrogen loss in the catchment. However, orchards lost more available phosphorus due to higher soil phosphorus content compared to arable systems (Chen et al., 2019).

2.5 Conclusion

This review has highlighted areas where orchard management is detrimental to soil health or where management strategies could be changed to improve soil health. In conclusion, the three following recommendations are made to improve orchard soil health:

- **Reduced agrochemical inputs**

  Contemporary management utilises frequent input of herbicides, insecticides and fertilisers which have deleterious effects on soil fauna. Critical soil functions are driven by soil fauna, therefore adapting management to reduce inputs is imperative to improving soil health. Reduction of herbicide inputs through mulching offers a more sustainable alternative to control weeds, particularly where it can be derived from alleyway vegetation.

Fertigation, where fertiliser is fed in situ with irrigation water may offer a viable alternative to broadcast application where reductions in fertiliser application are desired alongside maintenance of high yields.

Reduced insecticide application can be targeted through provision of natural pest control. Wildflower strips have demonstrated potential to enhance natural enemy services in orchards. Additionally, adopting integrated pest management systems may allow reduced insecticide input.

Alternating tree placement between rows and alleyways where replanting occurs may reduce fumigation requirements. Additionally, adopting more diverse gene pools of tree stock may reduce fumigation requirements.

- **Enhancing biodiversity**

Management to enhance soil faunal diversity such as the implementation of wildflower strips in alleyways should be considered. Altering mowing frequency to optimise plant diversity may also be beneficial, even where wildflower strips are not sown. Increased plant diversity is linked to beneficial cascade of effects on soil fauna and is also linked to increased soil multifunctionality, even under future climates. Notably, the beneficial effects of plant diversity on multifunctionality under N addition are inconsistent amongst studies.

- **Minimise soil compaction**

Soil compaction in orchards occurs with regular trafficking associated with mowing and agrochemical application. Reducing trafficking could be achieved through the implementation of recommendations in ‘Reduced agrochemical inputs’ and ‘Enhancing biodiversity’. Reducing compaction is essential to maximise soil-based ecosystem services, particularly where future threats such as drought may harm orchard production.

3. A comparison of soil health under conventional and wildflower sown alleyways in apple orchards

3.1 Introduction

There is a growing interest in the UK to promote practices for the sustainable management of soil. This is linked to future challenges, such as climate change, threatening food production (DEFRA, 2023). However, soil health varies in agroecosystems due to the breadth of management strategies, meaning it is important to assess soil health across distinct systems (Yang et al., 2020). UK apple orchards are an important source of food, but also provide other ecosystems services (Demestihas et al., 2017a). Despite this, there is limited data on apple orchard soil health in the UK and the effects of management practices on orchard soil health. Several definitions of soil health exist (Lal, 2016) but it can broadly be described as a measure of a soil ecosystems capacity to provide ecosystem services. As soils provide a wide range of ecosystem services, there is no single method to measure soil health, but soil health measurements covering chemical, biological and physical indicators are common (Lehman et al., 2015).

Experiments targeting natural and agricultural systems have found consistent improvements in measures such as nutrient retention (Li & Chen., 2019) and yield stability, and other common indicators of ecosystem functionality under management that increases or conserves plant diversity (Isbell et al., 2017). Approaches to increase plant diversity and subsequently soil health, have varied in different agroecosystems, but the fundamental plant traits that can elucidate improved soil health are recognised, if yet far from fully realised (Bardgett et al., 2014). Organic matter and soil carbon constitute important aspects of soil health and are frequently used as an indicator of soil health (Lal, 2016). Studies indicate improved plant diversity can increase sources of soil carbon through diverse litter and other belowground carbon inputs, although the exact mechanisms are not entirely understood (McDaniel *et al*., 2023). Long term experiments in grasslands indicate it may occur through increased rhizosphere inputs relative to decomposition of extant soil carbon (Lange *et al*., 2015).

Increasing plant diversity in agroecosystems has also been linked with greater nitrogen retention and declining nitrous oxide emissions, which are both valuable ecosystem services when aiming to reduce the environmental impact of agriculture (Li & Chen, 2020). However, some studies indicate negative effects on yields due to increased competition for N resources. Alternatively, cover mixes including legumes as well as non-leguminous species have been shown to alleviate these concerns and even promote increases in yield (Li & Chen, 2020). Furthermore, determining C:N ratio is important in agroecosystems as it can approximate the balance of nitrogen in a system. A high C:N ratio can indicate immobilization of N whereas a low C:N ratio can infer excessive plant available N through mineralisation. Studies have shown increased plant diversity can balance (increase or decrease) C:N to more favourable ratios, highlighting benefits to productivity and stoichiometry of agroecosystems (Chen & Chen, 2021).

The diversity of plant cover has been shown to affect soil phosphorus availability and uptake, indicating an influence upon cycling. Grasslands with higher plant diversity were shown to have greater phosphorus levels in their biomass but similar available phosphorus in the soil (Oelmann *et al*., 2011). This is likely driven by complementary uptake strategies of roots and increased release of phosphatase, an enzyme used to acquire phosphate ions from organic sources (Chen et al., 2022). Root traits are heavily linked to microbial community composition and activity, a key aspect of ecosystem function. Root exudates profiles can also foster specific microbial communities in their rhizospheres and effect microbiome diversity. Other traits such as functional type, physiology, and architecture also effect microbiome diversity and overall ecosystem functionality (Bardgett et al., 2014). Some studies in which increased plant diversity and pH have been monitored have observed limited changes in pH (Vanzolini et al., 2017). However, soil management and inherent characteristics such as parent material are considerably more influential on soil pH. Management decisions such as liming can often be the most determinative factor.

Biological indicators are a useful measure of ecosystem functionality. Earthworms have been identified as suitable bioindicators for sustainable management practices in orchards and other agroecosystems (Paoletti et al., 1998). Earthworms, key ecosystem engineers, provide several services. Burrowing can improve the physical structure of soil and enhance soil aeration and drainage. Mucilage in earthworm burrows can also encourage the formation of water stable soil structures and stimulate microbes (Bedano *et al*., 2019; Scheu, 2003).

Beyond inherent soil properties such as texture and topography, other physical indicators are valuable in assessments of soil health. Studies have highlighted complementary benefits of combining root traits and their capacity to improve multiple physical characteristics of soil such as penetration resistance and water infiltration (Saleem *et al.*, 2020). Bulk density and penetrometer resistance are frequently used in assessments of soil health (Shah et al., 2017). They offer insight to several characteristics of a soil. Effects of land management such as compaction will increase bulk density and penetrometer resistance due to a reduction in soil pore space. Other factors such as soil parent material (particle size) and organic matter can affect these measures (Da Silva *et al*., 1997; Gao *et al*., 2012)

Assessing the benefits of increased plant diversity on soil health across different agroecosystems is important. Using a range of indicators (chemical, biological and physical) to identify changes in soil parameters can establish which aspects of plant diversity in orchards may be beneficial to soil health. Apple orchards provide excellent opportunities for improving plant diversity due to their permanent alleyways (non-cropping areas) and longevity. Expanding the potential benefits of increasing plant diversity in apple orchards may encourage growers to adopt more sustainable management practices. Management practices such as wildflower strips in alleyway crops may receive increased interest in the future from growers following the introduction of ELM payments (DEFRA, 2023). The implementation of sown wildflower strips has been trialled in the UK to increase orchard diversity. Thus far, the success of these wildflower strips has been assessed from the perspective of aboveground ecosystem services, such as provision of natural enemies and pollinators (McKerchar et al., 2020). However, studies targeting belowground services such as climate and water regulation, and nutrient cycling in the context of UK orchards are more limited (Demestihas et al., 2017a; Webber et al., 2022).

This study aims to assess soil health in commercial apple orchards with conventional and wildflower alleyway management. Different soil characteristics, encompassing physical, chemical, and biological indicators, were used to gauge soil health. Therefore, it was hypothesised that:

1. Wildflower strips will significantly improve physical soil characteristics (Penetrometer readings, bulk density and aggregate stability) when compared to conventional alleyways.
2. Wildflower strips will significantly improve chemical characteristics (pH, TN, C:N ratio, plant available nitrogen, microbial biomass nitrogen and inorganic phosphorus) when compared to conventional alleyways.
3. Wildflower strips will significantly improve biological indicators (worm counts) when compared to conventional alleyways.

3.2 Methodology

3.2.1 Study site, sampling, and preparation

Text Soils were sampled in September 2020 from orchard alleyways at seven commercial orchard sites in Kent, United Kingdom. All sites contained plots with conventional alleyway management and alleyways previously managed with wildflower strips. Sites varied by soil type, irrigation and apple variety as seen in Table 3.1.

Table 3.1. Descriptions of seven orchard site characteristics and management properties. Sites with comparable characteristics.

Sites	Soil	Irrigation?	Apple variety
Newington, Eastside	Heavy clay loam	Rain fed	Braeburn
Broomside, Shelving	Light clay loam	Irrigated	Jazz
Parasol, Wheatsheaf	Light clay loam	Irrigated	Jazz
North Court	Heavy clay loam	Rain fed	Braeburn

Alleyway vegetation at study sites was composed of common native grassland species, and experimental wildflower strips containing native grass (*Lolium perenne* and *Dactylis glomerata*), legume (*Lotus corniculatus* and *Trifolium pratense*) and forb species (e.g. *Leontodon hispidus* and *Prunella vulgaris).* Plots were established for a previous study on pollinator and natural enemy services in 2013 (McKerchar *et al,.* 2020).

Species lists were recorded for each plot during sampling to determine species richness. Species richness and overlap in species between wildflower managed and conventionally managed alleyways were determined. A Sørensen index analysis was used to indicate similarity between vegetation covers. Results indicated similarity between vegetation cover at all sites (>0.5).

Table 3.2. Species richness in seven orchard sites with conventional and wildflower sown orchard alleyways. Similarity between plant communities in conventional and wildflower alleyways is expressed through Sørensen index analyses – A value of 0 indicates no overlap in species between plots and a value of 1 indicates complete overlap in species between plots.

Site	Conventional alleyway species richness	Wildflower alleyway species richness	Species in common	Sørensen index
Newington	9	13	9	0.81
Eastside	8	11	8	0.84
Broomside	2	4	2	0.67
Shelving	2	4	2	0.67
Parasol	6	8	6	0.85
Wheatsheaf	5	8	5	0.62
North Court	6	13	6	0.63

A stratified random sampling approach was taken to account for the shallow incline observed in orchards and their heterogenous nature. Plots were divided into quarters perpendicular to alleyways and with a random sampling point taken in each quarter. At each sampling point three 10 cm deep auger samples were taken in a 50 cm strip along the centre of an alleyway to form a composite sample. A bulk density ring sample (6 x 5.5 cm) was collected between two of the sampling points. Earthworm counts were performed by extracting a 20 x 20 x 25 cm block of soil using a spade. The block of soil was then deposited into a plastic tray and hand sorted to extract all earthworms for counting in field. Finally, penetrometer readings were taken at three separate points for a target depth of 10 cm.

3.2.2 Soil analyses

Soil auger samples were used for analysis of plant available nitrogen, inorganic P, total nitrogen, microbial biomass nitrogen, pH, organic matter, and C:N ratio.

Microbial biomass nitrogen was determined using chloroform fumigation as described by Vance *et al* (1987). Samples were fumigated for 24 hr and then extracted using 0.5 M K_2SO_4. Sample extracts were analysed using a TOC-L analyser (Shimadzu, Kyoto, Japan). Total N was calculated from the non-fumigated extracts and the difference between the two was used to calculate microbial biomass. Nitrate and ammonium were obtained by shaking 5 g of fresh soil in 25 ml of 1.0 M KCl for one hour on an orbital shaker, with the subsequent mixture being filtered through Whatmann No. 1 filter paper. Extracts were analysed on a Seal Auto-analyser 3 HR (Seal Analytical). Inorganic P was obtained by shaking 5g of fresh soil in 25 ml of 0.5 M $NaHCO_3$ for 30 minutes on an orbital shaker, with the subsequent mixture being filtered through Whatmann No. 42 filter paper. Extracts were analysed on a Seal Auto-analyser 3 HR (Seal Analytical). To measure pH, 10 g of air-dried soil and 25 ml distilled water was added to a 50 ml corning tube. The soil and water were agitated using an orbital shaker for 30 minutes and allowed to settle. A pH reading was then taken at the soil-water interface.

Soil moisture was calculated by drying 5 g of fresh soil at 105 °C to a stable mass. This was used to calculate dilution factors of fresh soil where necessary. Organic matter was calculated by loss on ignition at 550 °C for six hours. To calculate CN ratio oven dry soils were transferred into Eppendorf tubes and ground in a ball mill. Following grinding, samples were dried again overnight 80°C and then weighed into tin cups and rolled. Rolled samples were then analysed for C:N ratio on a Elementar Vario EL Cube (Elementar, UK). Bulk density was determined by drying a soil core of known volume (6 x 5.5 cm) at 105 °C for 24 hours and removing any stones. The volume of removed stones was accounted for by water displacement and calculated by dividing the dry mass by its volume.

3.2.3 Statistical analyses

Data were compiled in Microsoft Excel (Microsoft Corporation, 2018, Microsoft Excel) and analysed in R Studio (Version 3.3.0+, RStudio Team (2023); RStudio: Integrated Development for R. RStudio, Inc., Boston, MA, USA). Data were tested for normality of distribution and homogeneity. An analysis of variance (ANOVA) was used for testing statistical significance, with independent variables represented by site and plot, and a dependent variable being a soil health indicator. This was followed by a Tukey’s post hoc test for pairwise comparisons where appropriate. In addition, a principal component analysis (PCA) of all soil health indicators was performed to quantify the effects of site and plot.

3.3 Results

Consistent trends were identified between soil health indicators and sites. Sites were frequently significantly different whereas wildflower plots had limited effects on most soil health indicators.

3.3.1 Organic Matter

No significant differences in organic matter (%) were found between wildflower (blue) and conventional (red) plots, but there were significant differences between sites (P<0.001) (Figure 3.1). The wildflower plot at Broomside (8.28±0.26 %) displayed significantly higher organic matter than all plots at Newington, Eastside, Parasol, Wheatsheaf, and wildflower plots at Shelving and North Court (P<0.05). Mean organic matter % varied by a factor of two between the lowest at Wheatsheaf conventional (3.93±0.05 %) and highest mean measured at Broomside wildflower.

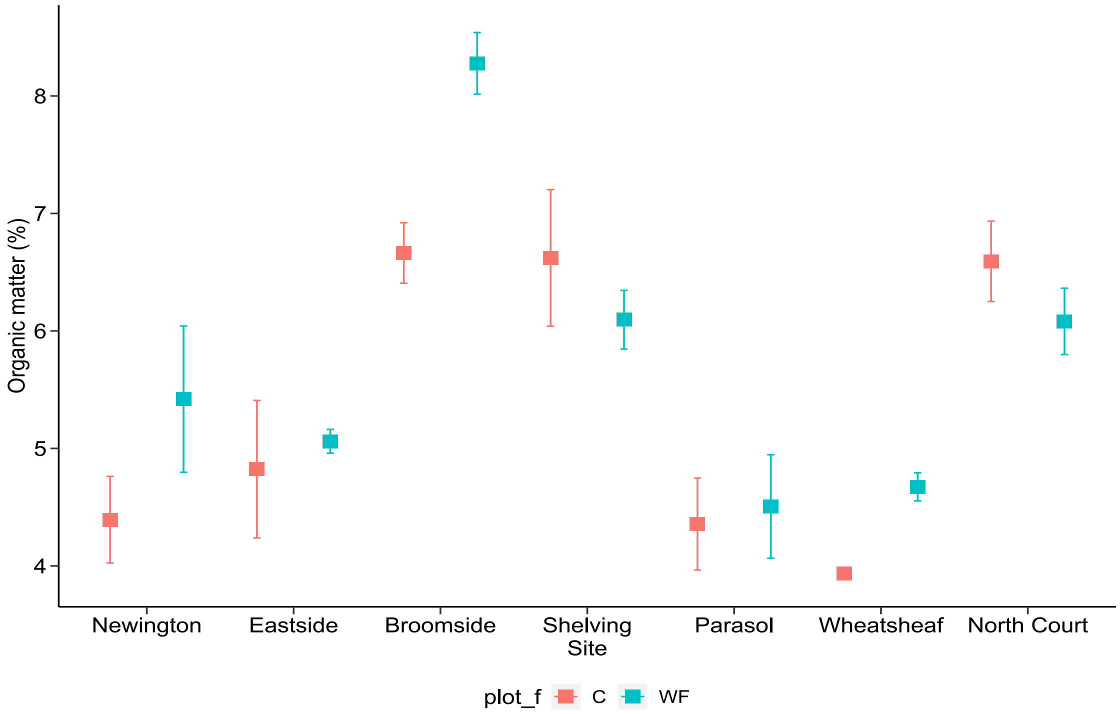

Figure 3.1. An error plot depicting organic matter% (LOI) of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P=0.001>***; Plot P=0.58(NS); Site-plot P=0.59

3.3.2 Plant available nitrogen

Plant available nitrogen was significantly different between sites but not wildflower and conventional plots (Figure 3.2). Plant available nitrogen was highest at Broomside wildflower (32.91±16.60 µ/g) with similar values at Parasol wildflower (30.61±10.57 µ/g) and conventional (28.49 ±10.99 µ/g), the three plots showed high variability compared to other plots. Broomside had the largest difference in plant available nitrogen (23.32 µ/g) between wildflower and conventional plots. However, plant available nitrogen was below 9 µ/g and less variable in wildflower and conventional plots at Newington, Eastside and Shelving.

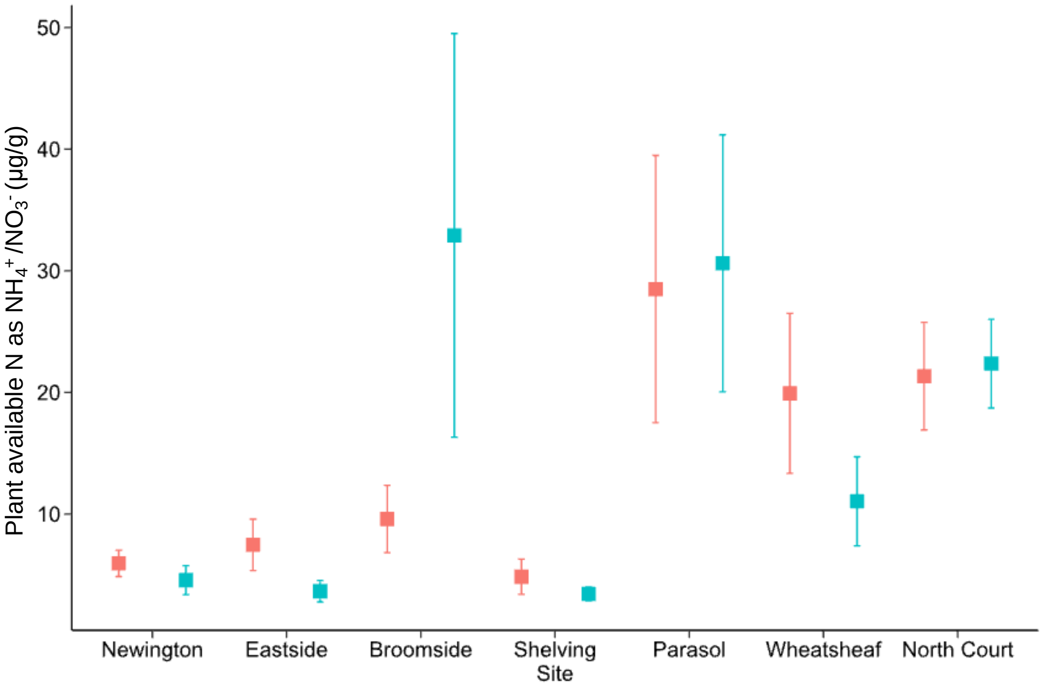

Figure 3.2. An error plot depicting plant available nitrogen NH_4^+/NO_3^- (µg/g) of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P=0.01>**; Plot P=0.66 (NS)

3.3.3 C:N ratio

Wildflower plots had no consistent effect on C:N ratio when compared to conventional plot with increases and decreases observed at sites (Figure 3.3). C:N ratios were significantly different between some wildflower and conventional plots but not between plots at the same sites. This is displayed by a significant site (P<0.01) and site-plot interaction (P<0.01) C:N ratios in conventional North court (7.69±0.29) and Shelving (7.37±0.30) plots were significantly higher than both plots at Parasol and Wheatsheaf, and conventional plots a Newington and Eastside. The wildflower plot at North Court (6.96±0.15) was significantly higher than Wheatsheaf (5.26±0.04) conventional.

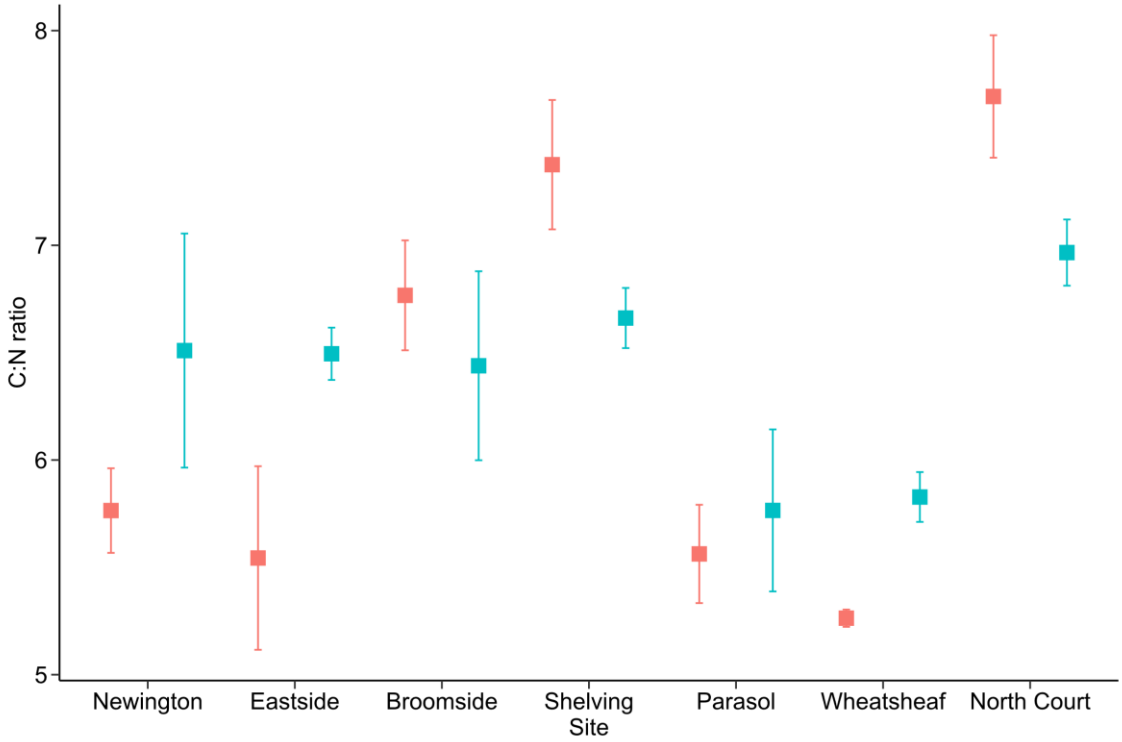

Figure 3.3. An error plot depicting CN ratios of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.001***; Plot P=0.53(NS); Site-plot P<0.01**.

3.3.4 Microbial biomass nitrogen

A significant effect of site on microbial biomass N was observed, but wildflower and conventional plots were not found to be significantly different (Figure 3.4). All significant differences (P<0.05) were provided from Broomside. Conventional (34.19±5.47 ug/g N) and wildflower plots (35.39±4.02 ug/g N) were significantly higher than Newington wildflower (10.04±1.37 ug/g N) and conventional (9.72±3.65 ug/g N), Parasol wildflower (9.55±2.48 ug/g N), and Eastside conventional (6.80± 1.35 ug/g N). Broomside wildflower was also significantly higher than Eastside wildflower (15.94±2.74 ug/g N).

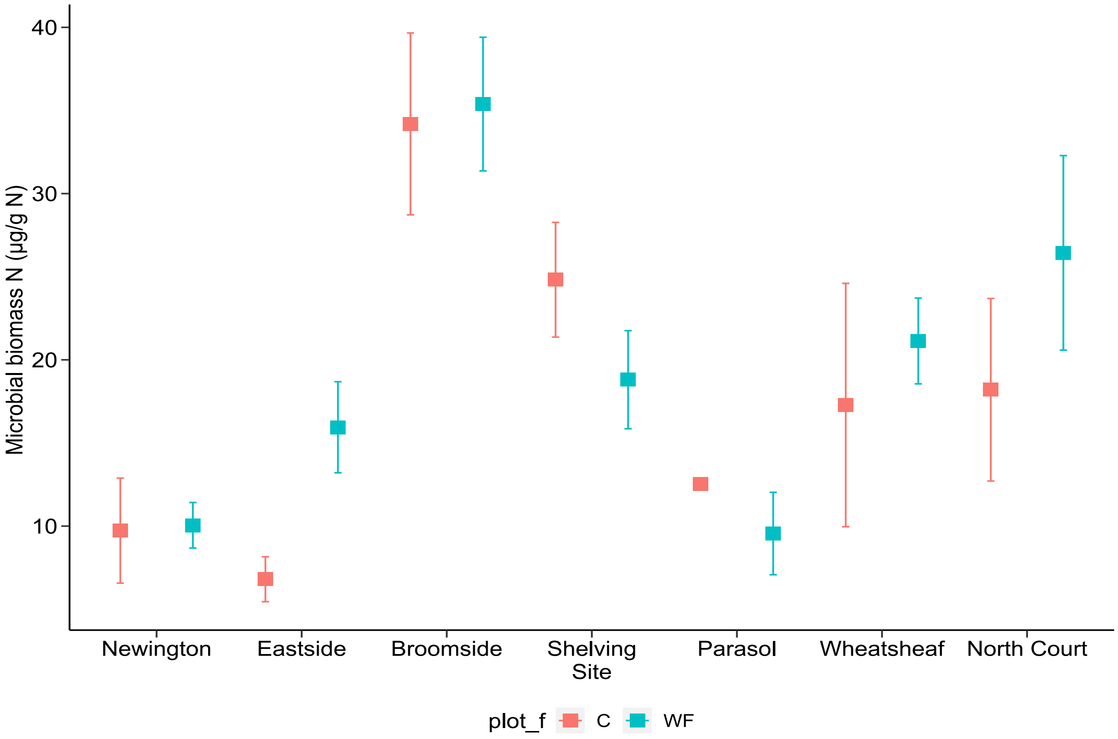

Figure 3.4 An error plot depicting microbial biomass nitrogen (µg/g N) of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.001***; Plot P=0.38(NS); Site-plot= 0.428.

3.3.5 Total nitrogen

Total nitrogen was shown to be significantly different between sites but not plots (Figure 3.5). This significant difference was detected between the Broomside wildflower plot (43.12±13.28 µg/g N) and Eastside wildflower (10.29±3.27 µg/g N; P=0.037). Non-significant decreases in total were detected in the other five plots. Other Mean values of TN between plots differed by a factor of up to 3, but a high degree of variability was also present.

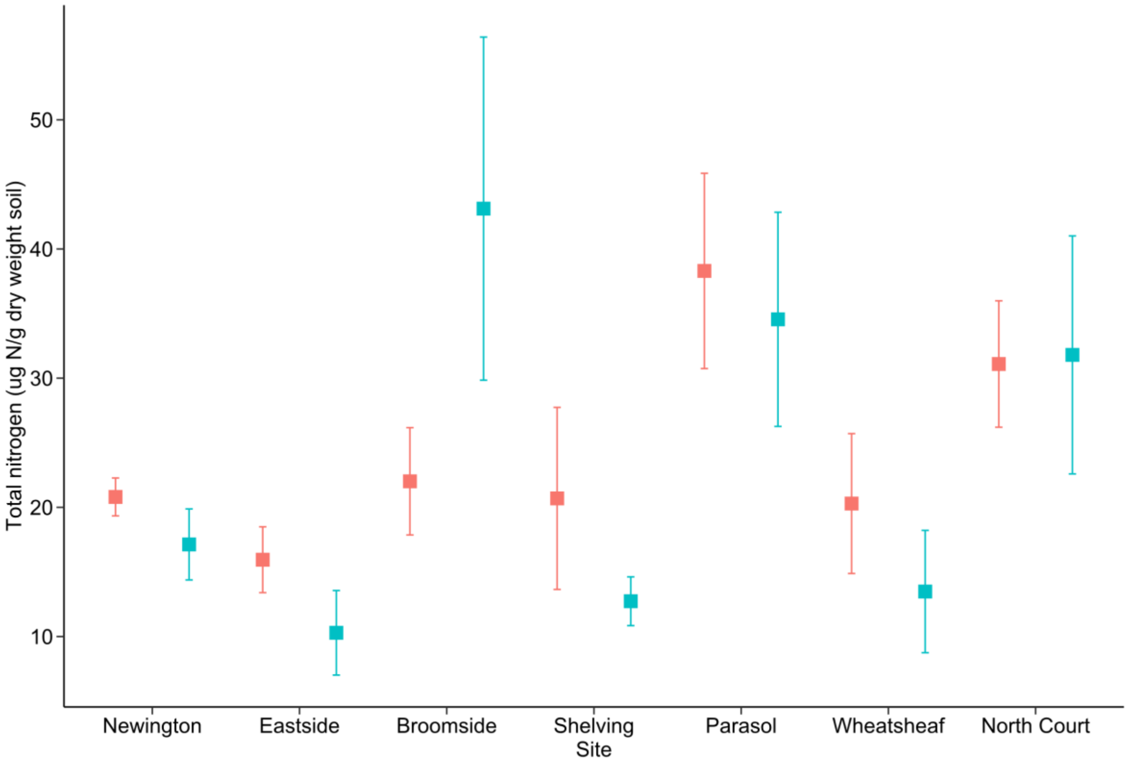

Figure 3.5. An error plot depicting total nitrogen (µg/g N) of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.01**; Plot P=0.79(NS); Site-plot P=0.29(NS).

3.3.6 Inorganic phosphorus

As shown in figure 3.6, Inorganic P was found to be significantly different between sites (P<0.001) and plots (P=0.05; as shown in Figure 3.6). The interactions between sites and plots were also significantly different (P<0.001), meaning that there was an inconsistent effect between treatments at the sites. Inorganic P at Shelving was significantly lower in the wildflower plot (26.77±4.48 g/kg) compared to the conventional plot (65.27±11.23, P<0.001). Inorganic P was also lower in the wildflower plot (11.66±0.87 g/kg) compared to the conventional plot (25.91±4.31 g/kg) at North Court, but this was not significant (P>0.005). Broomside wildflower (66.49±3.30 g/kg) was significantly higher (P<0.05) than all plots except Broomside conventional (65.77±5.23 g/kg) and Shelving conventional. Similar significant differences were found in the conventional plot at Broomside except for Parasol wildflower (41.26±5.36 g/kg).

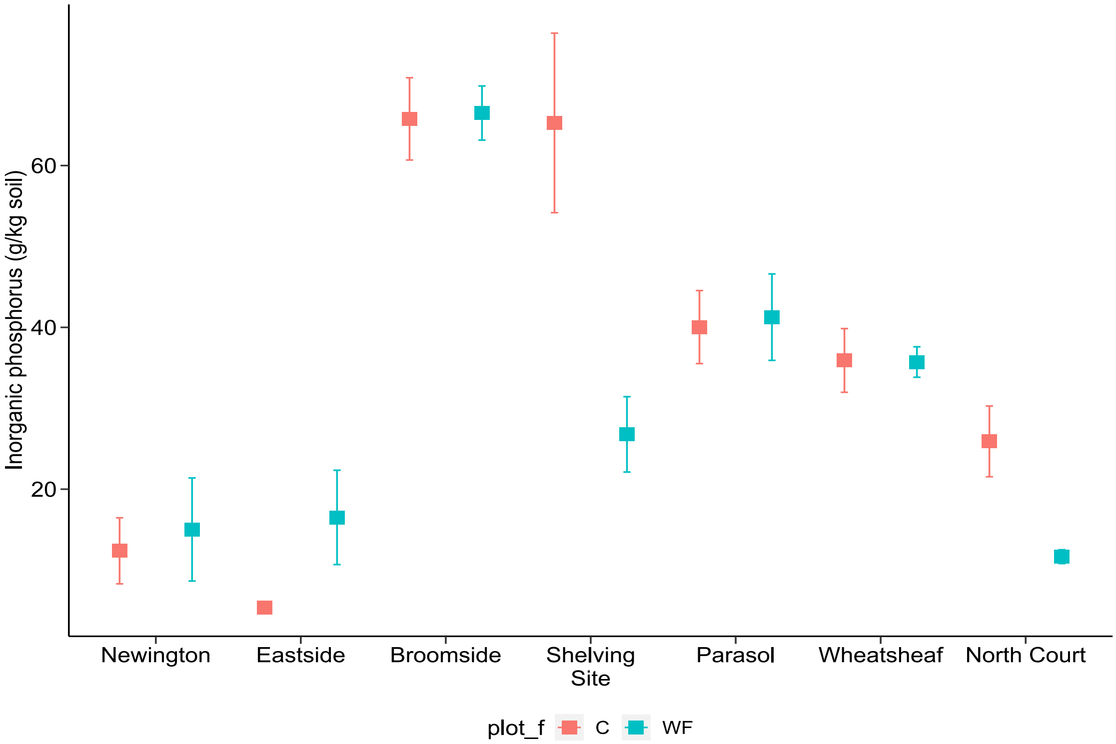

Figure 3.6 An error plot depicting Inorganic Phosphorus (g/kg soil) of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.001***; Plot P<0.05*; Site-plot P<0.001***.

3.3.7 pH

In Figure 3.7, pH was significantly different between sites and the interaction between sites and plots is also significantly different. This is demonstrated by pH increasing or decreasing in wildflower plots relative to conventional plots. Broomside possessed the highest mean pH of sites at 7.28±0.08 and North Court had the lowest mean pH of 5.95± 0.06.

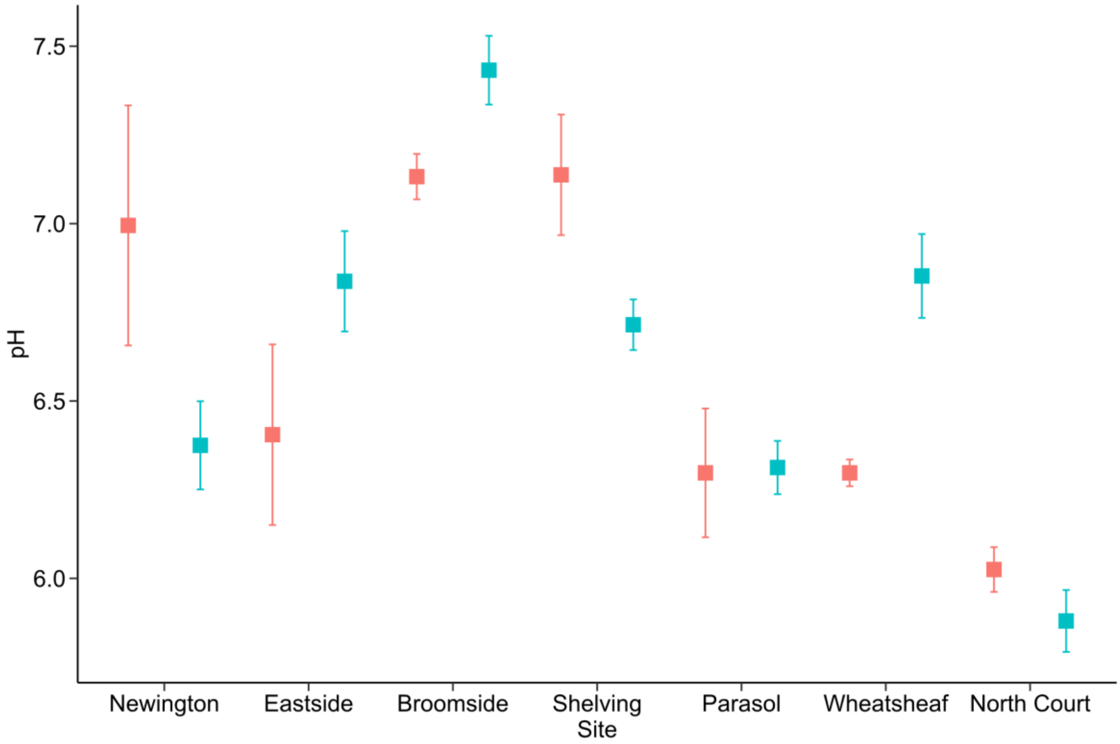

Figure 3.7 An error plot depicting pH of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.001***; Plot P=0.84; Site-plot P<0.01**.

3.3.8 Bulk density

Figure 3.8 shows Bulk density was significantly different between sites but not between wildflower and conventional plots. The wildflower plot at Broomside (0.85±0.06 g/cm^3) was significantly lower than the conventional plot at Wheatsheaf (1.15±0.05 g/cm^3, P=0.027). Otherwise, mean Bulk density showed some variability within plots but values were similar.

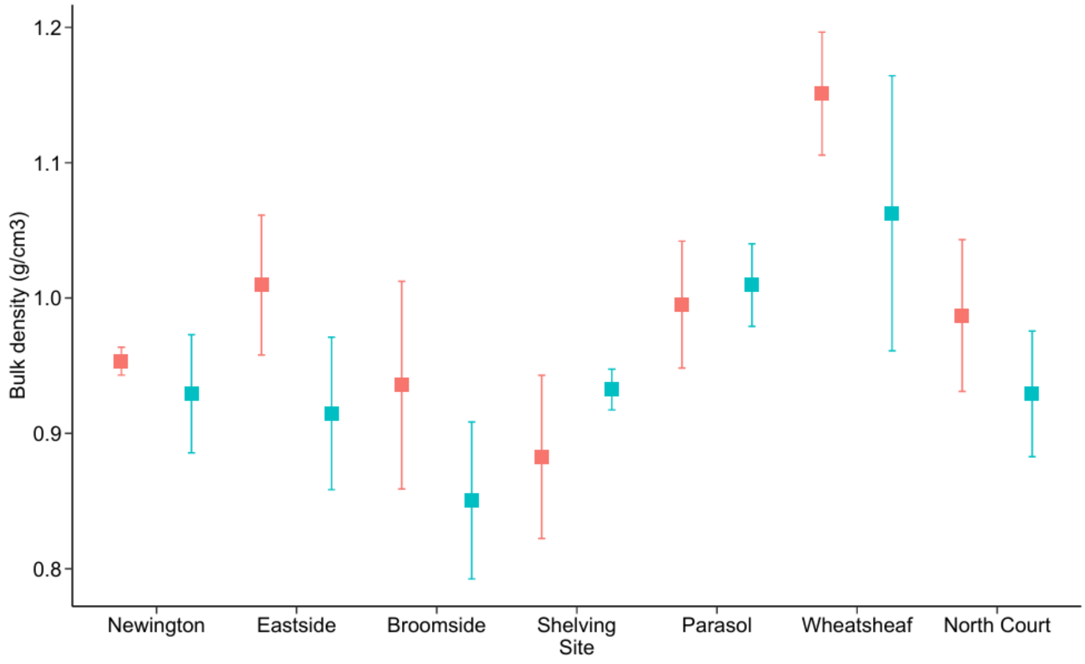

Figure 3.8 An error plot depicting Bulk density (g/cm3) of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.001***; Plot P=0.17(NS); Site-plot P=0.78(NS).

3.3.9 Penetrometer resistance

Penetrometer resistance was significantly different between sites and the interaction between sites-plots was also significantly different, as shown in Figure 3.9. The significant difference in site-plot interactions is exemplified by the contrasting values in Eastside wildflower and Parasol wildflower. The wildflower plot at Eastside (2727±143.34 Kpa) was significantly higher than both plots at Broomside and Parasol, and the wildflower plot at Wheatsheaf. The wildflower plot at Parasol (1342±127.33 Kpa) was significantly lower than all other plots

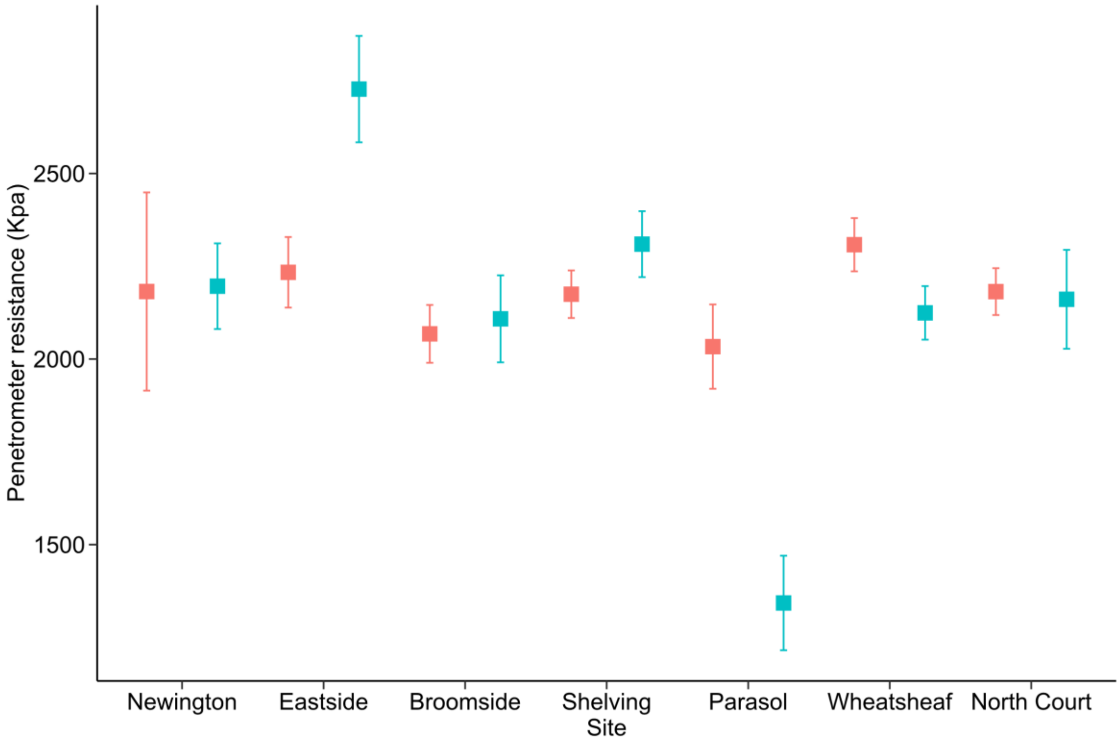

Figure 3.9 An error plot depicting penetrometer resistance (KPa) of soil collected from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.001***; Plot P=0.64(NS); Site-plot P<0.001***.

3.3.10 Worm counts

Worm counts were significantly different between sites (P<0.001) but not wildflower and conventional plots (P=0.09; as shown in Figure 3.10). The interaction between sites and plots were marginally insignificant (P=0.051). Worms counts in both Broomside plots and the conventional plot in Shelving had no worms present. The wildflower plot at Parasol had the highest mean worm count (5.00±1.58), which was significantly higher (P=0.05) than the conventional plot at Parasol (0.50±0.29) and both Broomside and Shelving plots (WF= 0.25±0.25).

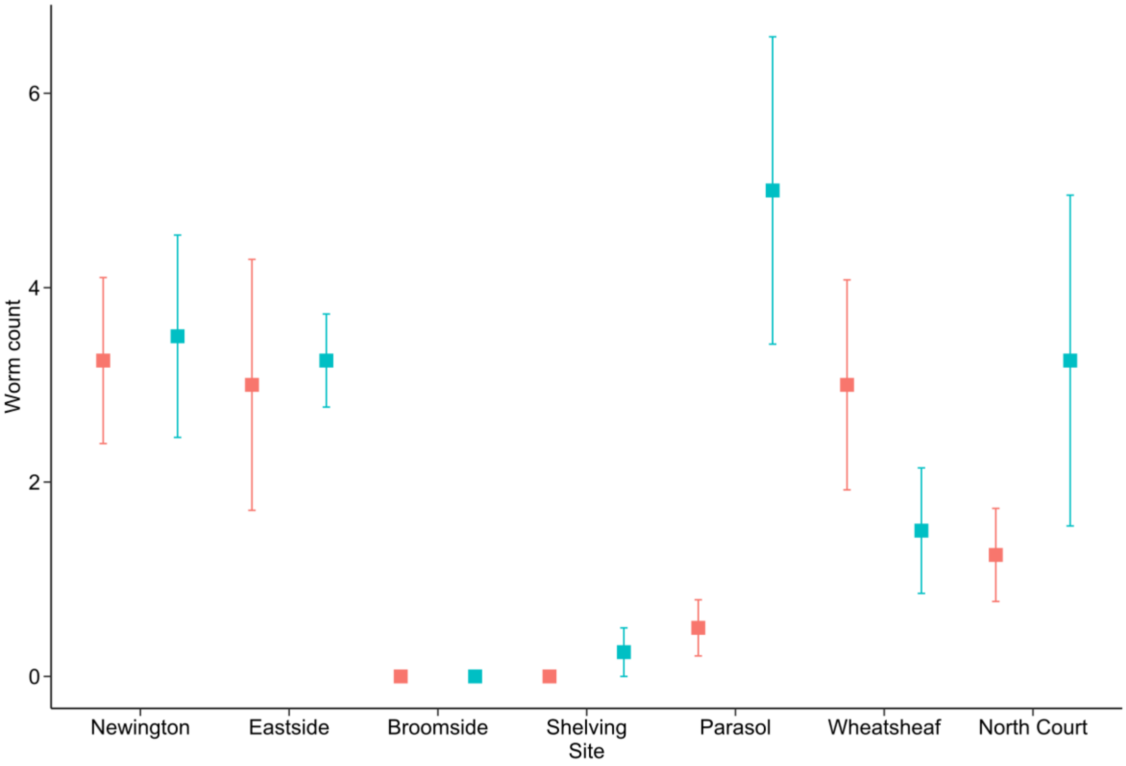

Figure 3.10 An error plot depicting worm counts from seven commercial orchard sites with conventional (C, red) and wildflower (WF, blue) strip managed alleyways. (N=4). ANOVA (P-value): Site P<0.001***; Plot P=0.09(NS); Site-plot P=0.051(NS).

3.3.11 Soil health indicator benchmark principal component analysis

The combined axes of the PCA explain 55.6% of the variation within the dataset. PC1 explained 34.5% and PC2 explained 21.1%. There are no distinct groupings of conventional or wildflower plots in figure 3.11. However, sites are distinguishable by factors such as C:N, pH, microbial biomass nitrogen, and organic matter at Broomside. Newington, Wheatsheaf and Parasol were distinguishable by bulk density and worm counts. Soil health characteristics were distinguishable by site rather than plot. This is evidenced by site specific clustering, although with some overlap amongst sites. Overall, soil health indicators were better characterized by site rather than wildflower or conventional plots. Loadings for worm count, bulk density, and organic matter strongly influence PC1, with worm count and bulk density negatively correlated with organic matter. Penetrometer resistance, plant available nitrogen and total nitrogen strongly influence PC2, with penetrometer resistance being negatively correlated with plant available nitrogen and total nitrogen. Loadings for C:N and pH displayed weak influence over either PC1 or PC2 but were positively correlated.

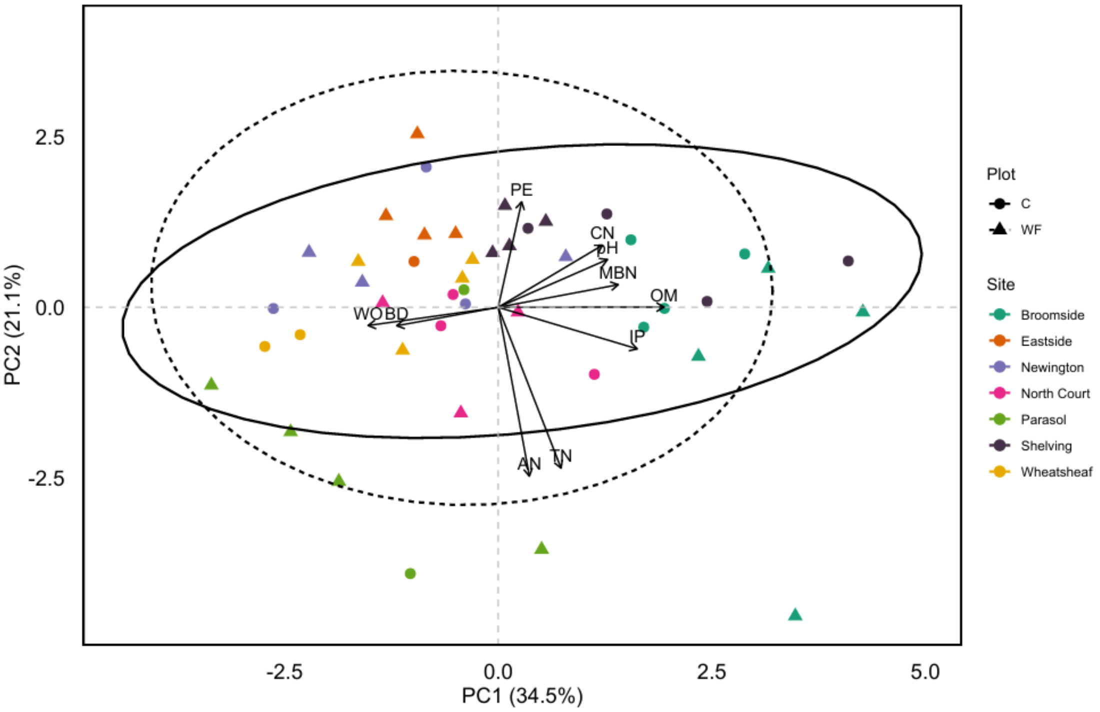

Figure 3.11. A principal components analysis of the relative abundance of all soil health indicators. PC1 explains 34.5% and PC2 explains 21.1% of variability of soil health indicators. Soil health indicators are denoted by bulk density (BD), worm counts (WO), penetrometer resistance (PE), C:N, MBN (microbial biomass nitrogen), OM (organic matter), IP (inorganic phosphorus), total nitrogen (TN), plant available nitrogen (AN), and pH. Triangles and circles display individual plots. Soil health indicators were not explained by wildflower or conventional management in plots.

Discussion

This study aimed to quantify potential benefits of sown wildflower strips in apple orchards compared to conventionally managed orchard alleyways. It was hypothesised that wildflower sown alleyways would significantly improve 1) physical, 2) chemical, and 3) biological measures of soil health. Results from this study indicate sown wildflower strips provide no consistent, significant improvement on soil health in apple orchards when compared to conventionally managed alleyways. Therefore, all three hypotheses were rejected. Significant differences were found in P, but the interaction at sites was inconsistent, with P increasing or decreasing in wildflower plots relative to conventional plots at different sites. No consistent significant differences were detected in physical (bulk density and penetrometer) or biological properties (worm counts). Similarly, to P, significant differences between wildflower and conventional plots were detected but the interactions were also significantly different. The inconsistent interactions are highlighted with resistance increasing in Eastside wildflower strips and decreasing in Parasol wildflower strips.

Insufficient replication between combinations of site characterises rendered us unable to compare between soil type and irrigation status. Soil types significantly influence soil characteristics and were consistent between plots at sites but not between all sites (see table 3.1).

Despite no consistent trend in wildflower strips improving soil health indicators, some significant differences were found a specific sites. Worm counts at Parasol were significantly higher under wildflower strips and penetrometer resistance was found to be significantly lower, indicating a possible interaction between soil structure and worm abundance. Studies have indicated that soil compaction can negatively affect worm abundance and reduce their associated benefits to soil function such as nutrient cycling and soil aggregation (Al-Maliki & Scullion, 2013; Blouin et al., 2013; Röhrig et al., 1998). However, bulk density values between Parasol plots were similar which may suggest differences in worm counts may have been mediated by soil moisture rather than soil compaction (Ruiz et al., 2016). Worm counts in these plots were still considered poor, even in a cropping context. This indicates current management approaches may not encourage healthy worm populations (see chapter 2).

Broomside displayed significant increases in plant available N and total N under wildflower strips. This difference was most likely attributed to the presence on rabbit faeces in this plot, rather than the presence of N-fixing legumes such as *T. pratense or L. corniculatus* included in the sown wildflower mixture (Bakker et al., 2004).

Studies attributing benefits of increased plant diversity on soil processes have found improvements when increasing from singular species to two or three and limited effects beyond that. Moreover, studies have indicated there are limited benefits of plant diversity on ecosystem stability and processes in response to perturbations, with specific plant identity and functional characteristics being the primary determinant (Bardgett & Wardle, 2010). Numerous studies have reported benefits of increased plant diversity on soil properties in agroecosystems, but these studies typically compare increases in diversity in monocultures or even bare soil, notably in Mediterranean orchard systems (Morugán-Coronado et al., 2020).

Similarities in soil properties between wildflower and conventional plots as highlighted by a PCA (see Figure 3.11) could be attributed to significant overlap in functional traits (Bender et al., 2016). Both plots were largely grass-dominated, likely due to regression in the wildflower plots related to orchard management and the time elapsed since their establishment (six years). Additionally, conventional plots contained species that may facilitate similar functions to sown wildflower species (Phoenix et al., 2008). Similarities between plant communities in conventional and wildflower sown plots at each site were highlighted with Sørensen index values ranging between 0.62 – 0.85 (see table 3.2). This may support the regression of species diversity overtime in wildflower sown alleyways, and similarity between plant community effects on soil health in conventional and wildflower managed alleyways. Few studies have compared the provision of soil processes by sown and spontaneous plant cover in an agricultural context. A study comparing the nitrogen dynamics (mineralisation, losses and uptake) between areas of set aside land sown with different species mixtures and spontaneous found negligible between covers (Eggenschwiler et al., 2009). Additionally, diverse wildflower strips may not be optimal for carbon sequestration in agroecosystems (Harbo et al., 2023).

A PCA of all soil health indicators across sites revealed site specific clustering, with some overlap between sites. This likely indicates spatial differences between sites rather than wildflower or conventional management in alleyways may have a greater influence on soil health indicators. Inherent soil properties, such as soil texture, or management practices carried out at these sites could explain the clustering presented in Figure 3.11. Bulk density was negatively correlated with C:N, pH and microbial biomass nitrogen, and penetrometer readings were negatively correlated with plant available nitrogen and total nitrogen. This may indicate compaction at sites negatively affects soil health in this study. Perturbation of orchard soil ecosystems, related to management, may attenuate benefits on soil processes related to plant diversity. Compaction was also evident at all sites in this study, primarily at the interface of vegetated alleyways and herbicide free strips. Penetrometer resistance above 1000kpa has been associated with deleterious effects on plant growth and soil function. All plots in this study were found to exceed this threshold. Compaction can have severe consequences on soil processes: it can increase bulk density and reduce soil porosity, thus reducing soil-atmosphere gas exchange, and water infiltration and storage (Whalley et al., 1995).

Application of agrochemicals (herbicides, pesticides, fungicides, and fertilisers) and frequent trafficking may interrupt or inhibit aboveground-belowground biotic linkages. Application of agrochemicals has been linked to decreased diversity and abundance of soil fauna and microbes (Beaumelle et al., 2023). This could have implications for soil processes as important functions such as litter decomposition are supported by soil fauna and microbes (Hättenschwiler & Gasser, 2005; van der Putten et al., 2013).

Studies pondering the benefits of diversity and whether it is desirable regardless of function, have highlighted diminishing gains in ecosystem functionality, even when increasing from a low number of functional types (two to three) (Bardgett & Wardle, 2010). However, studying these effects can be challenging to quantify when considering spatial and temporal factors. Broad ranges of species, up to 84% in experimental communities, provided benefits to ecosystem function when evaluated under different conditions (Isbell et al., 2011). This highlights the complex nature of above and belowground linkages and how optimising improvements in ecosystem functionality through increased biodiversity must consider other factors such as the practical feasibility of maintaining biodiversity and economic cost of doing so. Encouraging biodiversity in agroecosystems is an important aspect of sustainable soil management, but determining optimal diversity under different management intensities warrants further study (Hector, 2022).

Elucidating ecosystem function under current wildflower strip mixtures separate from management pressure could highlight soil health improvement thresholds (Fridley, 2002). Beyond this, studying vegetation mixtures under a gradient of management intensity in orchards could also provide insight into how different management practices disrupt or modify plant-soil interactions (Kleijn et al., 2019). Identifying these interactions could allow growers to integrate more economical and sustainable management practices in orchard systems and adapt to future stressors likely to occur under climate such as drought (Lehmann et al., 2020)

3.4 Conclusions

In conclusion, no significant changes were found in soil health indicators when comparing spontaneous alleyway vegetation to sown wildflower strips. This outcome may be attributed to two factors: the difference in vegetation composition, particularly after wildflower strip regression, may not have been sufficiently different over time to produce benefits. Management practices may mask or disrupt the way plant diversity can influence soil. Results from this study indicate more research is needed in relation to plant diversity and the gradients of management intensity and its outcomes for soil health.

4. Investigating the responses of conventional vs. wildflower-sown orchard alleyway soils to drought and rewetting

4.1 Introduction

There is a growing interest in soil health management of orchards in the UK, driven by climate change and the necessary resilience that will need to be developed to cope with these unprecedented challenges (DEFRA, 2023; Tahat et al., 2020) Drought arguably represents the greatest threat to cropping systems worldwide with the severity, frequency, and duration of drought episodes likely to increase in the future (Bodner et al., 2015b). An orchard fruit survey conducted by DEFRA in 2012 revealed 78% (13712 Ha) of fruit producing orchard land was situated in the Southeast & London (43%, 7501 Ha) or the West Midlands (35%, 6211 Ha). Studies projecting future drought scenarios in the UK have identified the Southeast and Central England as vulnerable to drought (Rahiz & New, 2013). Managing orchards to improve their soil health and develop greater resilience to drought is critical to the future of production but approaches to improve soil health require further research.

Management for increased plant diversity is a focus of contemporary research and demonstrates potential for improving soil health. More diverse plant cover has been linked to reduced soil erosion, greater conservation of soil moisture, improved soil carbon sequestration, and overall soil biodiversity (Sharma et al., 2018). Improved plant diversity in agroecosystems provides the opportunity to include a greater variety of functional plant traits, particularly below ground traits, that are linked to improved soil health (Griffiths et al., 2022; Zhang et al., 2022). Belowground functional plant traits influence soil ecosystems in several different ways. Architectural and morphological traits focus on aspects of roots such as rooting depth, branching, root diameter and dry biomass. Physiological and biotic traits concern kinetic processes and their associations with other living organisms such as mycorrhizal fungi (Bardgett et al., 2014). An assembly of functional traits can influence soil ecosystem processes. Orchards provide a good opportunity for introducing diverse plant cover in the form of wildflower strips in alleyways due to their relative permanence and the minimal intrusion required for establishment. Beyond benefits belowground, several advantages have already been prescribed to wildflower strips in orchards.

Plant diversity can also influence soil microbial communities and their responses to abiotic stressors, such as drought (de Vries et al., 2016; Li et al., 2022). Root exudates stimulate microbial communities and the breakdown organic matter (Bardgett et al., 2014). Some traits may facilitate a shift towards fungal dominated soils which typically enhance carbon sequestration (Laliberté, 2017). Larger, diverse microbial communities can also be more resilient and support ecosystem processes under drought (Williams & de Vries, 2020a). Greater plant diversity is linked to larger and more diverse microbial communities in soils (Zak et al., 2003). The effects of plant diversity and management on drought resistance and resilience of soils have been explored outside of an orchard context, often in experimental grassland systems or pot experiments. A study by Cole et al (2019) at Colt Park found increased plant diversity resulting from grassland restoration management, such as withdrawing fertiliser input, increased soil carbon flux resistance to drought. Bloor and Bardgett (2012) found no difference in CO_2 fluxes of model grassland ecosystems with varying species richness and N availability when exposed to drought or subsequent rewetting. Intensive management of grasslands has been linked to increases in drought resistant and resilient dominant bacteria, whereas the reverse effect was found in dominant fungi (Lavallee et al., 2024). Promotion of bacterial dominated systems may bolster soil resilience to drought but lead to decreased resistance to drought compared to fungal dominated systems. Furthermore, efficiency of C and N cycling in intensively managed grasslands may decrease where bacteria are dominant (Lavallee et al., 2024).

Drought and subsequent rewetting cycles can lead to dramatic shifts in soil biology and chemistry including respiration rates in the short term (Fierer & Schimel, 2002), and significant changes in carbon dynamics over longer time scales (Schimel et al., 1999). Many short term changes in soil carbon can be explained by the ‘Birch effect’ where respiration is elevated or ‘pulses’ to a higher rate than baseline (Birch, 1958; Schimel et al., 2007). This phenomenon can be driven by several mechanisms; expulsion of osmolytes accumulated by microbes to survive drought, metabolization of necromass, or mobilisation of stable soil carbon. Over longer timescales interactions between these mechanisms can lead to a loss of soil carbon from stable sources (Schimel et al., 2007). However, carbon loss can be mitigated by land management for diverse vegetation cover, such as wildflower strips, which can return carbon to the soil (Chen et al., 2018).

Studies investigating ecosystem responses to disturbances, such as drought, have utilised ecological terminologies where definitions have been inconsistent. Resistance is broadly recognised as the ‘ability to withstand disturbance’ (Nimmo et al., 2015). Whereas, resilience has been interpreted as either the ‘capacity to recover following disturbance’ (Nimmo et al., 2015) or ‘capacity of a system to persist or maintain function in the face of exogenous disturbance’(Hodgson et al., 2015). Such definitions are contradictory, and researchers have sought to unify terminology for resilience (Bardgett & Caruso, 2020). This study uses resistance as the capacity to withstand drought, as evidenced by stable rates of microbial respiration under drought. Resilience is considered as recovery, whereby microbial respiration returns to a similar rate before exposure to drought. This is similar to other studies assessing drought impacts on soils (Cole et al., 2019; de Vries et al., 2018; Wardle et al., 2000).

The responses of soil to drought and rewetting in agroecosystems are critical to understand the effects differing land management on ecosystems resistance and resilience to abiotic stress. The aim of this study was to compare responses of conventional (control) and wildflower sown orchard alleyway soils to drought and subsequent rewetting. It was hypothesised that:

i. the resistance of orchard soil respiration (CO_2) to drought would be greater in wildflower plots compared to conventional.
ii. the resilience of orchard soil respiration (CO_2) after drought would not be significantly different in wildflower and conventional plots.

To test these hypotheses an incubation experiment was established using soil cores from orchard alleyways with contrasting vegetation cover: conventionally managed alleyways, and alleyways sown with wildflower strips. These cores were subjected to severe drought, mild drought, and control treatments where soil respiration was monitored.

4.2 Methodology

4.2.1 Sampling site and preparation

Intact soil cores were collected in April 2021 from orchard alleyways at two commercial orchards in North Kent, United Kingdom. Both sites contained plots with and without managed wildflower strips. Newington was a rain fed apple orchard with wildflower strips established in 2013 as part of a pollinator ecosystem services experiment (McKerchar *et al*, 2020). The other site, Marshgate, was an irrigated pear orchard with wildflower strips that were established in 2019. Soil cores were collected on April 19th and 20th 2021 respectively.

Two treatments were considered at each site, established wildflower strips and conventional management which was dominated with grasses (90%+ cover) vegetation. Plots within each treatment were located in an orchard alleyway. Conventional plots at Newington were dominated by *Lolium perenne* and Marshgate was dominated by *L. perenne, Dactylis glomerata, Holcus lanatus,* and *Plantago lanceolata.* In the wildflower strip at Newington *L.perenne* also maintained significant cover but was accompanied by *Achillea millefolium, Galium verum, Trifolium repens, Veronica sp, Lotus corniculatus, Hypochaeris radicata,* and

Cerastium fontanum. Marshgate wildflower strips were composed of *Leucanthemum vulgare, A. millefolium, T. repens, Veronica sp, G. verum, S. dioica, T. officinale* and reduced cover of *L. perenne* and *D. glomerata*.

Species lists were recorded in plots at Newington and Marshgate to provide species richness. Species richness and common species between wildflower managed and conventionally managed alleyways were recorded. A Sørensen index analysis determined similarity between vegetation covers. Results indicated similarity between vegetation cover at Newington (0.86), whilst vegetation cover at Marshgate (0.31) was not similar.

Table 4.1. Species richness in two orchard sites with conventional and wildflower sown orchard alleyways. Similarity between plant communities in conventional and wildflower alleyways is expressed through Sørensen index analyses – A value of 0 indicates no overlap in species between plots and a value of 1 indicates complete overlap in species between plots.

Site	Conventional alleyway species richness	Wildflower alleyway species richness	Species in common	Sørensen index
Newington	9	12	9	0.86
Marshgate	4	9	2	0.31

Figure 4.1. A map showing the location of Newington and Marshgate plots/sites

A block design was used when sampling to reduce the effect of heterogeneity within plots. Areas for sampling in each plot were chosen to be representative of the sites by visual assessment for vegetation cover and overall condition of the soil surface. Turf was cutback in the chosen areas at each site before sampling commenced. Twenty cores were collected in each plot, 15 for the experiment and five supplementary cores to determine water holding capacity, totalling 80 cores overall. Cores were collected in 3 x 10 cm PVC tubing that was inserted into the soil and extracted. Extracted cores were sealed with Parafilm and electrical tape then transported to Lancaster University in a cool box, where they were deposited in a cold store at 4°C until the experiment was initiated. Soil auger and bulk density samples were also collected and soil properties (pH, plant available nitrogen, microbial biomass nitrogen), presented in Table 4.1.

Table 4.2. Key soil properties in conventional and wildflower-sown strips in two orchard sites (±SE, n=5). Marshgate ANOVA (P-value): (P<0.05). Newington ANOVA (P-value): (P>0.05)

	Marshgate Conventional	Marshgate Wildflower	Newington Conventional	Newington Wildflower
Bulk density	0.98 (±0.02) $g/cm^{3\ a}$	0.73 (±0.07)±$g/cm^{3\ b}$	0.83(±0.03)g/cm^3	0.83(±0.03)g/cm^3
pH	6.20 (±0.08)	6.51(±0.12)	6.99(±0.04)	6.37(±0.05)
Plant available N (NH_4^+/NO_3^-)	4.63(±3.25)µg/g	2.79(±1.94)µg/g	4.85(±3.01)µg/g	7.09(±4.21)µg/g
Microbial biomass N	59.58(±0.14)µg $N/g^{\ a}$	35.11(±4.85)µg $N/g^{\ b}$	40.98(0.22)µg N/g	66.96(0.12)µg N/g

Prior to the experiment, supplementary cores were used to determine the water holding capacity (WHC) in each plot. Cores were weighed then sealed with mesh and soaked overnight until saturated. The cores were elevated to allow excess water to drain and then weighed for 100% WHC. The cores were then dried in an oven at 105 °C to determine oven dry mass.

4.2.2 Incubation experimental design

Overall, 60 intact cores were used for the experiment with five replicates for each treatment. For the duration of incubation, cores were contained in 1 litre Kilner jars in a Sanyo MIR 553 incubator at 21 °C. Jars were sealed during gas sampling using adapted Kilner jar lids. Approximately, 1.3 cm holes were punctured in the centre of the lids to allow rubber butyl septa to be inserted and then secured with silicone sealant. Cores were also resealed at the bottom using Parafilm and electrical tape, with the top end kept open.

Cores were arranged in a randomized block design. Three treatments were randomly applied for each of the four plots – 75 %, 50 %, and 25 % WHC. A WHC of 75 % was maintained for all cores outside of the drought treatment period and represented conditions in the field at the time of sampling. A 'mild drought' treatment of 50% WHC was imposed to represent a gradual decline in soil moisture. Additionally, a ‘severe drought’ 75% WHC treatment was also used to represent a stronger drought effect. Mild and severe drought treatments were used to potentially identify differences in resistance and resilience thresholds to drought under orchard alleyways managed with wildflowers or managed conventionally. Cores were regularly watered via pipette using tap water to a pre-determined target mass. During the drought stage control cores (75% WHC) continued to be watered regularly, 50 % WHC cores were weighed each day and half of the water mass lost was replaced to produce a reduced rate of soil drying, and 25% WHC received no water until target treatments were met. The experiment took place over eleven weeks, with an initial three-week acclimation period for equalising WHC between plots, a four-week drought treatment, and four weeks between rewetting and the final sampling (see fig 4.2). Gas samples were taken at four different stages – an initial benchmark in May 2021 to provide baseline CO_2 emissions, at the end of drought treatment to gauge soil resistance to drought, after rewetting, and three weeks after rewetting to determine resilience to imposed drought treatments. Gas samples were collected 16 hours after watering, except at the rewetting stage. Gas samples were collected after 6 hours during rewetting to allow for sufficient time for soil cores to rehydrate due to their compact nature and for soil temperatures to stabilise. This was used to capture gas emissions under the representative conditions of the treatment, but also capture the large efflux of gases following initial rewetting.

Four samples were taken at each sampling period (see fig 4.2). Samples were taken immediately after jars were sealed providing a baseline for gas emissions. Further samples were taken after 24, 48, and 96 hours. This sampling regime was chosen to account for expected low rates of emission during the drought period and to monitor the anticipated decline in gas emissions following rewetting surge. Gas samples were taken using 20 ml syringes with 16 mm BD Microlance needles. The syringe was flushed with ambient air three times prior to taking gas samples, and then flushed three times when drawing the samples from the sealed jar to ensure sufficient mixing of gases. A 10 ml sample was drawn from each jar, with 1 ml expelled to account for gases held in the needle. The remaining 9 ml was immediately injected into a pre-evacuated 3 ml Labco exetainer. All samples were analysed on a Perkin Elmer Autosystem XL GC-FID using pre-made standards for 500 ppm, 1000ppm, and 4000ppm CO_2 (BOC, UK). Rate of CO_2 was calculated using the following formula:

CO_2-C mg L^{-1} g^{-1} h^{-1} = CO_2-C mg * headspace volume * CO_2 molecular weight (44.01)/ temperature (Kelvin) * gas constant (0.0821)/ soil mass (g)/ 1000/ 24

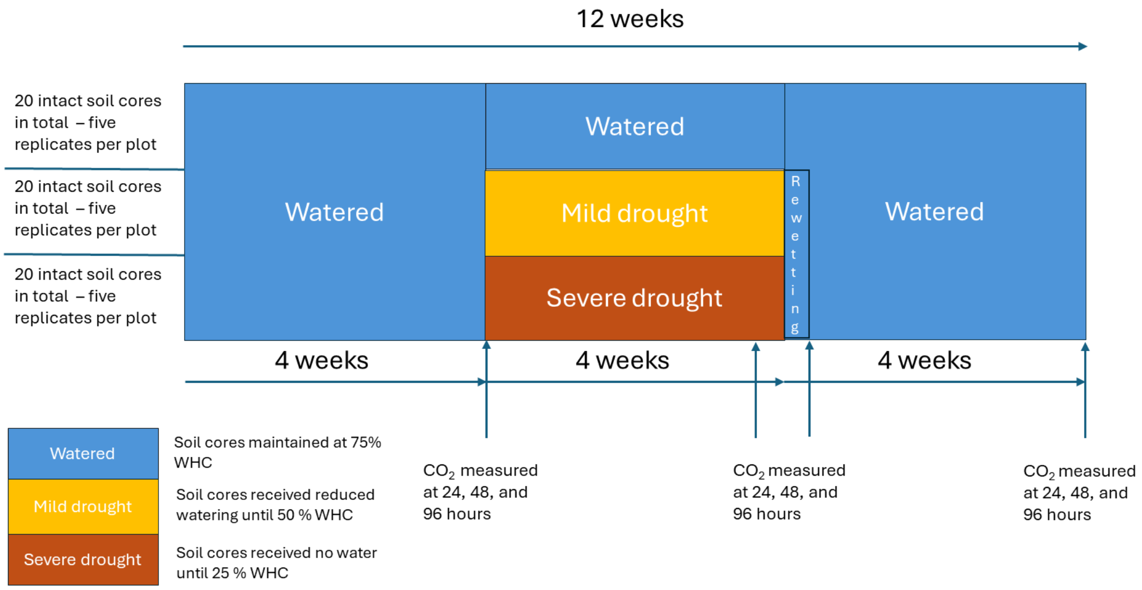

Figure 4.2 A conceptual methods figure displaying treatment intensity (WHC%) and timing (number of weeks), timing of sampling (indicated by arrows) with annotations, and number of soil cores from each site and plot used in total (n=60).

4.2.3 Soil property analyses

Bulk density was determined by drying five soil cores of known volume from each plot at 105 °C for 24 hours and removing any stones. The volume of removed stones was accounted for by water displacement and calculated by dividing the dry mass by its volume. Nitrate and ammonium were obtained by shaking 5 g of fresh soil in 25 ml of 1.0 M KCl for one hour on an orbital shaker, with the subsequent mixture being filtered through Whatmann No1 filter paper. Extracts were analysed on a Seal Auto-analyser 3 HR (Seal Analytical). Microbial biomass nitrogen was determined using chloroform fumigation as described by Vance et al (1987). Samples were fumigated for 24 hr and then extracted using 0.5 M K_2SO_4. Sample extracts were analysed using a TOC-L analyser (Shimadzu, Kyoto, Japan). To measure pH, 10 g of air-dried soil and 25 ml distilled water was added to a 50 ml corning tube. The soil and water were agitated using an orbital shaker for 30 minutes and allowed to settle. A pH reading was then taken at the soil-water interface.

4.2.4 Statistical analyses

Data were compiled in Microsoft Excel (Microsoft Corporation, 2018. Microsoft Excel) and analysed in R Studio (Version 3.3.0+, RStudio Team (2022); RStudio: Integrated Development for R. RStudio, Inc., Boston, MA, USA). Carbon dioxide emission data was tested against plot and drought treatment factors. Newington was analysed separately from Marshgate due to different management practices between sites. Plots at Marshgate were not able to be compared statistically due to differences in land use history influencing soil characteristics in the plots. The conventional orchard plot at Marshgate was previously grassland, whereas the wildflower plot at Marshgate was previously arable land. Data were tested for normality of distribution and homogeneity. Analysis of variance (ANOVA) was used for testing statistical significance, followed by a Tukey's post hoc test for pairwise comparisons.

4.3 Results

4.3.1 Marshgate Wildflower

At the initial sampling stage (see Fig 4.3) there was a disparity in mean (*M*) carbon dioxide emission rates at T=1 (C *M* = 2.12 *SD = 1.37*, MD *M*= 1.66 *SD*= 1.01, SeD *M*= 1.33 *SD* = 0.53) but no significant differences were detected (*P*<0.05). However, at T=2 emission rates decreased significantly in mild drought cores (*M*= 0.87, *SD* = 0.27) compared to control T=1 initial sampling (*P*<0.01), severe drought (*M*= 1.00, *SD* = 0.14) and control (*M*= 0.90, *SD* = 0.24) cores were not significantly lower despite recording similar rates of emission. Variation in emission rates were also markedly lower in all treatments. A significant decrease was observed in all cores at T=3 when compared to T=1 control and mild drought cores (*P*< 0.05).

Under drought conditions at T=1 mild (*M*= 0.47, *SD* = 0.28) and severe drought (*M*=0.25, *SD*=0.11) cores were not significantly lower than control cores (*M*= 1.37, *SD* = 1.15) despite 65.7% (*P*= 0.444) and 81.8% (*P*= 0.077) reductions in emission rates respectively. Control cores follow the same trend from initial sampling with a decrease in emissions of ~60% between T=1 and T=2 with variability also decreasing. Emission rates remained stable in severe drought cores at all three timepoints.

Rewetted severe drought cores at T=1 (*M*= 1.47, *SD*= 0.44) produced a significantly higher rate of carbon dioxide emissions compared to the same cores at drought T=1. Mildly droughted cores produce the lowest rate of emission at T=1. At T=3 all cores in their respective treatments showed decreased emission rates compared to T=1 but only severely droughted cores are significantly lower (*M*= 0.30, *SD*= 0.04, *P*= 0.047). A marginal increase of 10.5% in emission rates was detected after rewetting when compared the initial sampling of severely drought cores.At the last sampling stage emission rates were lower than their respective treatments when sampled initially, but only the 71% in mildly droughted cores was found to be significantly lower (*P*= 0.008).

4.3.2 Marshgate Conventional

Initial sampling at T=1 (see Fig 4.3) displays variable carbon dioxide emission rates ranging from 2.37 to 1.49. T=2 and T=3 in the initial phase become less variable, whilst also decreasing. Cores at T=3 all express similar rates of emission (0.53 – 0.49).

After drought treatment severely droughted cores (*M*= 0.60, *SD*= 0.15) were significantly lower than control cores at T=1 (*M*= 1.59, *SD*= 0.45). By T=2 they were no longer significantly different likely due to increased deviation in the treatments. Severely droughted cores (*M*= 0.60, *SD*= 0.15) under drought treatment at T=1 were significantly lower (P< 0.001 than at initial sampling a T=1 (*M*= 1.86, *SD*= 0.98). Control and mild drought cores were not significantly different under the same comparison. A similar trend was observed in drought cores at T=2 with a staggered decrease in emissions from control to severely droughted cores.

Following rewetting, Control (*M* = 1.22, *SD*= 0.47), mild drought (*M* = 1.06, *SD* = 0.42), and severe drought (*M* = 1.21, *SD* = 0.42) cores were all significantly lower than initial control cores at T=1 (P<0.05). However, no significant differences between treatments were detected at this stage.

At the last sampling stage, control and severe drought core carbon dioxide emissions significantly decreased by 54.4% and 50.5% relative to their initial sampling rates (P<0.05). However, while there was also a 50.3% carbon dioxide decrease in mild drought cores, this was not statistically significant (P=0.206). All cores showed a small decrease in emission rates from T=1 to T=2 at the last sampling stage (<15%).

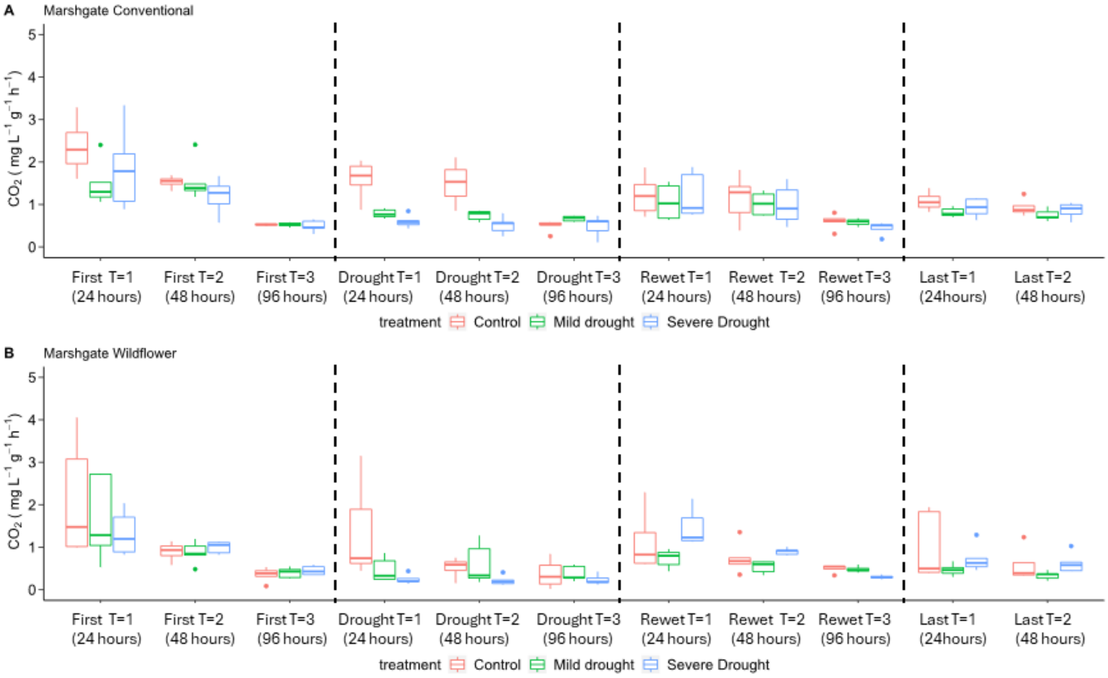

Figure 4.3. A boxplot showing carbon dioxide emissions rate of soil cores collected from A) conventionally managed B) Wildflower sown Kent orchard alleyways. Cores were exposed to three different drought treatments at 11 distinct time points. Each point displays median, 25% and 75% quartiles, and range. Outliers are shown by solid circles. T=1, T=2 and T=3 represent 24, 48, and 96 hours from baseline sampling respectively (N=5). With ANOVA (P-value) output: A) Timepoint P<0.001 *** , Treatment P<0.001 *** , Timepoint:Treatment P<0.05*. B) Timepoint P<0.001 *** , Treatment P<0.05 * , Timepoint:Treatment P= 0.107.

4.3.3 Newington Wildflower

Cores at initial T=1 (see Fig 4.4) sampling show similar rates of emission, but mild and severe drought were more variable. Further sampling at T=2 showed reduced variability. Emission rates at T=3 decline significantly from T=1 and T=2 across all cores (P= 0.05).

Under drought treatments at T=1 and T=2 cores showed a staggered range of emissions. Despite 58.8% (*P*= 0.08) and 57.3% (*P*= 0.06) lower emission rates between control cores and severe drought cores they were not found to be significant at T=1 or T=2 respectively. Severe and mild drought cores stayed relatively stable throughout the drought treatment phase (*SD Range*= 0.40-0.44, *M Range* = 0.54-0.63). Whereas control cores were stable a T=1 and T=2 and then dropped by 41.7%. Throughout the drought phase severely droughted cores had a consistent outlier.

After rewetting at T=1, severely droughted cores (*M*= 1.73, *SD*= 0.33) produced significantly higher (*P*<0.001) carbon dioxide emissions compared control cores (*M*= 0.81, *SD*= 0.23) at the same timepoint. The highest mean carbon dioxide emission rate was recorded in severely droughted cores after rewetting at T=1 but it was not significantly higher than any cores initially at T=1. The rate of emission at T=1 in severely droughted cores also significantly decreases by T=2

(*M*= 1.02, *SD* = 0.17). At T=3 under rewetting sampling cores exhibit a similar pattern to droughted cores.

At the last sampling stage emissions increased significantly (p<0.05) in each respective treatment between T=1 and T=2. A similar trend to rewetting T=1 in the last sampling stage was observed with staggered emission rates increasing from control to severe drought

4.3.4 Newington Conventional

When initially sampled at T=1 (see Fig 4.4) severe drought cores (*M*= 1.88, *SD*= 0.46) presented significantly higher emissions than control (*M*= 1.19, *SD*= 0.21) and mild drought (*M*= 1.29, *SD*= 0.33) with all cores stabilising to similar rates of emission at T=2.The final sampling point of initial sampling illustrated a decrease in emission consistent with that of other plots.

After drought treatment, severely droughted cores and mildly droughted cores emitted carbon dioxide at the same rate (T=1 =0.52, T=2 = 0.60, T=3 = 0.56-0.57). This was 40.2% and 43.47% lower respectively at T=1 and T=2 but was not deemed significantly different. Control emission dropped by T=3 which was consistent with other plots.

Rewetting resulted in a significantly higher (P= <0.01) rate of carbon dioxide emissions in severely droughted cores (*M* = 2.35, *SD* = 0.90) compared to control cores at T=1 (*M* = 1.30, *SD* = 0.15). However, mildly droughted (*M*= 0.83, *SD*= 0.20) cores produced only marginally higher rates of emission than control (*M*= 0.78, *SD*= 0.09) at T=1 and then same rate of emission at T=2 (0.74). By T=3 severely droughted cores had the lowest emission rate of all treatments, but no treatments were significantly different at this stage (*M*=0.39, *SD*= 0.05*).*

At the final sampling stage, carbon dioxide emissions increased for timepoint T=1 to T=2, similarly to the wildflower plot at the same site, and in contrast to both plots at Marshgate. Increases in the severely droughted cores from 0.96 to 1.85 (92.7%%), mildly droughted cores from 0.62 to 1.36 (119.4%) and control cores from 0.61 to 1.26 (106.6%) were deemed significant (P<0.01). Emission rates at late sampling T=1 were significantly lower than initial sampling, but then were not significantly different at their respective T=2 sampling stages (*P*<0.05).

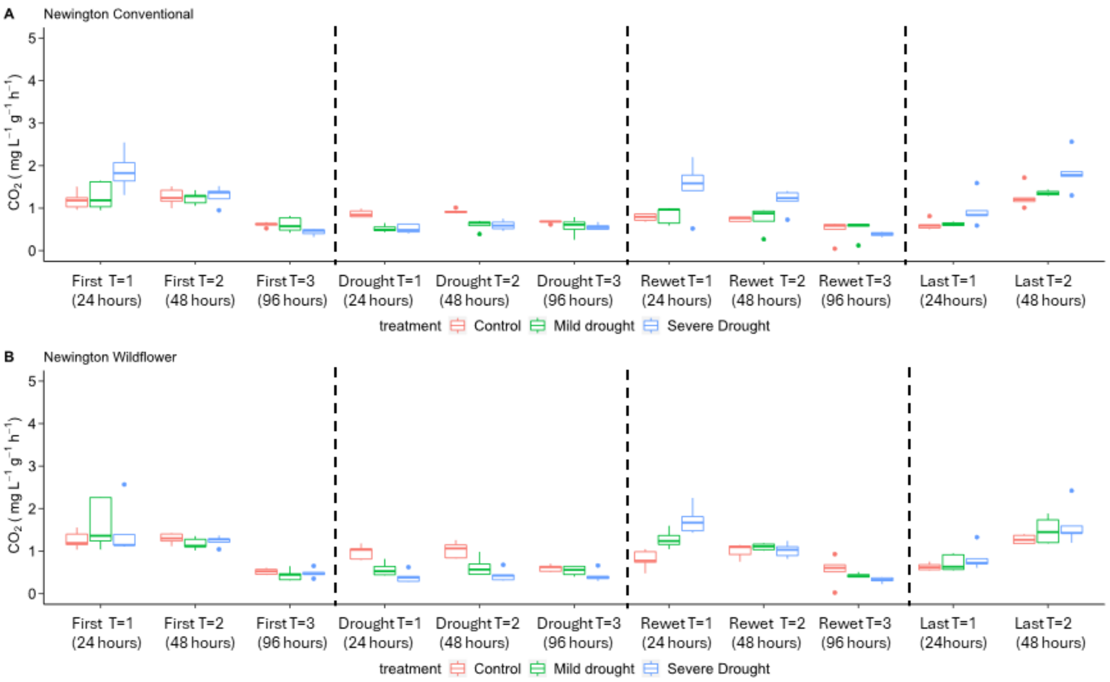

Figure 4.4. A boxplot showing carbon dioxide emissions rate of soil cores collected from A) conventionally managed B) Wildflower sown Kent orchard alleyways. Cores were exposed to three different drought treatments at 11 distinct time points. Each point displays median, 25% and 75% quartiles, and range. Outliers are shown by solid circles. T=1, T=2 and T=3 represent 24, 48, and 96 hours from baseline sampling respectively (N=5). With ANOVA (P-value) output: A) Timepoint P<0.001 *** , Treatment P<0.001 *** , Timepoint:Treatment P<0.001 ***. B) Timepoint P<0.001 *** , Treatment P=0.993 , Timepoint:Treatment P= 0.001*.

4.4 Discussion

This study aimed to determine the responses of conventional and wildflower sown orchard alleyway soils to drought and re-wetting. It was hypothesised that the resistance of orchard soil respiration (CO_2) to drought would be greater in wildflower plots compared to conventional. All plots showed effects of drought treatment with carbon dioxide emission reductions in severely droughted and mildly droughted cores when sampled at T=1 (24 hours). However, this was only significant under severe drought at Marshgate conventional with a 62.30% reduction in emissions (Figure 4.3). The absence of significant differences in other plots was likely attributed to variability between cores as similar or even greater reductions (MW= 85.8%, NC= 40.2%, NW= 58.8%) between control and severe drought cores were observed. Mild drought cores were also consistently lower than control cores at T=1 and T=2 except for Marshgate wildflower where they were marginally higher at T=2.

During drought treatment Newington conventional mild drought cores performed similarly to severe drought cores (Figure 4.4). This may have indicated greater microbial resistance to drought (Kakumanu et al., 2019). However, it is more likely due to variation in emission rates of cores. Severely droughted cores produced higher carbon dioxide emissions in the initial phase at T=1 and suffered a greater relative decrease in emissions at drought treatment. Greater drought stress in severely droughted treatment is also evidenced in the responses to rewetting (Rousk & C. Brangarí, 2022). Mildly droughted cores retained similar rates of emission to control cores whereas severely droughted cores produced a significant increase. Only conventional and wildflower plots at Newington were compared statistically in this study due to consistent management and establishment factors being found at this site. When compared, no significant differences were found between any timepoints. Results from this study indicate wildflowers had no effect on the resistance of orchard soil respiration (CO_2) to drought, therefore the first hypothesis was rejected.

Following rewetting severely droughted cores from both Newington plots and Marshgate wildflower produced significantly higher rates of carbon dioxide compared to control cores. Whilst Mildly droughted cores did not produce significant increases under rewetting. These results indicate microbes in severely droughted cores had undergone greater adaptations to tolerate drought stress (Fierer & Schimel, 2002; Kakumanu et al., 2019; Schimel et al., 2007). Newington wildflower was the only plot to show a significant decrease in emission rates between T=1 and T=2 after rewetting. Similarly, to drought, corresponding decreases in emissions after 24 hours were also found in Newington conventional and Marshgate conventional and a lack of significance in can likely be attributed to variation. In contrast to these plots, Marshgate conventional did not produce a comparable rewetting effect in severe or mild drought cores. Results from this study showed no difference in the resilience of wildflower and conventionally managed orchard alleyways after drought and subsequent rewetting, therefore we accept our second hypothesis.

The similarity in wildflower and conventional alleyway soils to drought and subsequent rewetting could be attributed to several reasons. Orchard soils may have had experienced multiple drought rewetting cycles previously (Zhang et al., 2020). Multiple or prior drought-rewetting cycles have been shown to alter responses over time. Soils exposed to multiple cycles have been shown to recover more quickly from drought and exhibit reduced responses to drought stress. This may be accommodated by shifts in community composition or previous adaptations to drought and rewetting (Fierer & Schimel, 2002).

Drought and subsequent rewetting cycles present possible changes in carbon dynamics in orchards. Under drought conditions gross primary productivity can reduce below total ecosystem respiration and cause ecosystems to become carbon sources (Van der Molen et al., 2011; Wang et al., 2014). Rewetting can further compound this effect by triggering bursts of microbial respiration increasing carbon dioxide emissions. However, the fate carbon accumulated for physiological adaptations in microbes is uncertain. Some studies have attempted to link a post rewetting carbon dioxide burst to carbon accumulated through osmolytes and other adaptations to drought. Although, conflicting responses other in studies may signal alternative pathways of carbon (Fierer & Schimel, 2002; Kakumanu et al., 2019; Schimel et al., 2007)

Soil ecosystem processes can occur at slower rates under drought, or their dynamics can change. Carbon and nitrogen accumulation in bacteria and fungi can increase dramatically to make physiological adaptations to drought. Amassing amino acids in bacteria and polyols in fungi, to act as osmolytes, is an energetically intensive process and is estimated to consume 3-6% of net primary productivity in grasslands (Kakumanu et al., 2019; Schimel et al., 2007). The fate of this of carbon and nitrogen following rewetting after drought is uncertain (Schimel et al., 2007). Diverse vegetation cover can be effective in improving certain soil health indicators related to drought resilience and resistance, such as soil structure and organic matter content, but site-specific management and soil characteristics such as soil texture can have a significant effect on efficacy (Blanco-Canqui & Ruis, 2020).

Vegetation cover in conventional and wildflower alleyways in Newington were similar, possibly explaining similarities in responses to drought and rewetting (also for soil health indicators in chapter 3). Marshgate wildflower plots had higher species richness and cover was dissimilar to the conventional plots. However, responses in Marshgate were likely confounded by differences in management history in plots (see table 4.2). Despite no significant evidence of increased resistance and resilience to drought in soils under differing vegetation cover throughout this experiment, other studies have identified improvements related to species composition (Craven et al., 2016; Li et al., 2022). The benefits of increased plant diversity may be maximised by tailoring root trait composition to select favourable characteristics for the desired soil improvements. Many traits to improve drought resistance and resilience are linked to improvements in soil structure. Total root length and root surface area have been linked with improvements to soil macroporosity and aggregate stability (Hudek et al., 2022). Certain root exudates profiles have also been identified as potentially providing beneficial enhancements under drought, although these areas require further research. Increasing biodiversity of vegetation cover is an effective way to utilise root traits for soil improvements as multiple traits can be incorporated simultaneously and work synergistically (Griffiths et al., 2022; Williams & de Vries, 2020b).

A study comparing experimental grasslands under drought highlighted the legacy effect of management intensity on aboveground and belowground carbon pools. Chomel et al (2022) found intensive grassland management impaired plant carbon delivery to soil-borne organisms when compared to extensively managed grasslands following drought. Aboveground biomass C assimilation increased but was retained in aboveground biomass, which destabilised microbial C pools. Whereas, no effect on C transfer to microbial pools was detected under extensive management. In contrast to these findings, a prior study found greater trophic complexity, often associated with extensive systems, had no effect on ecosystem stability under drought (Chomel et al., 2019). Photosynthate transfer from aboveground biomass was eliminated in this incubation study. Thus, the difference between soil responses under drought and control treatments may have been supressed. However, management intensities in this study were equal in sites which may have had a greater effect soil respiration.

When compared to other systems, higher species richness increased carbon dioxide emissions from soil, and is linked to more productivity of these communities in grassland experiments (Dias et al., 2010). Contrastingly, studies comparing soil respiration along gradients of management intensity found soil respiration increased under intensively managed, low diversity grasslands when compared to high diversity, extensively managed grasslands (Apostolakis et al., 2022). A study comparing drought resistance and resilience between intensively managed arable systems and extensive grassland discovered differences in soil food webs (de Vries et al., 2012). An extensive fungal based food web was more resistant and resilient to drought than a bacterially based intensive arable system (Bardgett & van der Putten, 2014; de Vries et al., 2012). The effects of increased plant diversity on grassland productivity have been observed in systems where nutrient additions or drought have been implemented and have been found to consistently improve productivity (Craven et al., 2016). Further study is warranted to quantify the effects of plant diversity on agricultural systems, particularly in a context of drought resilience and ecosystem stability in combination with multiple stressors (Yang et al., 2018)

4.5 Conclusion

Overall, this study provides evidence that there was limited effect of diverse plant cover in orchards on carbon dioxide emission responses to a soil drought-rewetting cycle. Results from this experiment demonstrate further insight in several areas regarding drought-rewetting cycles in soils, and management for diverse plant cover. Studies exploring the effects of plant diversity under different management regimes or soil textures are warranted. In field studies of drought resilience in apple and pear orchards planted with wildflower strips would also provide critical insight as there are discrepancies in soil respiration under lab and field treatments. Beyond this, studies quantifying total ecosystem respiration and orchard carbon fluxes would help illuminate potential changes in orchard ecosystem processes under various management regimes and climate change.

5. A comparison of plant functional group diversity effects on soil health outside of an intensive orchard system

5.1 Introduction

Interest in the sustainable management of soils in agricultural systems has accelerated (Lal, 2008). Wildflower strips have been targeted in apple and pear orchards, but research in an orchard context regarding benefits to soil health is limited (McKerchar et al., 2020). Many seed mixtures used to establish wildflower strips are used to provide floral resources or habitats for natural enemies of pests (Carvell et al., 2022; Mateos-Fierro et al., 2021; McKerchar et al., 2020). It is not clear whether commonly used wildflower strip species can also provide benefits to soil health, and which mixtures may be optimal.

Chapter 3 measured soil health indicators in commercial orchards to assess potential benefits to soil health from wildflower strips and found no consistent evidence compared to conventional grass dominated cover. Therefore, it is important to determine whether plant communities established through wildflower sowing provide demonstrable benefits outside of systems with external pressures and how these changes might relate to management. Additionally, studying potential soil health improvements from plant diversity in an applied context is challenging, particularly in intensive orchard systems, given other interacting factors (see chapter 2). Management interventions can substantially alter soil environments and change plant-soil interactions (Oelmann et al., 2015; Ward et al., 2016).

While there have not been any previous studies on soil health in wildflower sown orchard alleyways, many studies have explored the benefits of plant diversity for soil health; such benefits are more broadly prescribed to plant functional groups or even traits of plants (Cortois et al., 2016). The benefits of plant diversity are typically derived from the presence of functional traits and the productivity of species. The Mass Ratio Hypothesis dictates that the influence of plants on ecosystem processes can be related to its productivity and functional effect traits (Grime, 1998). While this has been shown to be an effective tool in some systems, certain circumstances have shown species can contribute minimally to productivity but can have disproportionately large effects on ecosystem processes, especially over time (Hector, 2022; Isbell et al., 2011). Furthermore, the influence of plant diversity can be attributed to niche complementarity, where temporal and spatial variance in resource acquisition strategies of plants can contribute to increased productivity and ecosystem process function (Sonkoly et al., 2019). Thus, predicting how different plant communities or plant functional groups will influence ecosystem processes and ultimately soil health remains difficult.

Nitrogen cycling in soil can be heavily influenced by the presence of specific functional groups (Abalos et al., 2019), particularly nitrogen fixing legumes. Forbs can also have a large control over inorganic nitrogen levels in soils (Craine et al., 2002; Hooper & Vitousek, 1998), for example, root exudate profiles can improve nitrogen availability through microbially driven organic matter breakdown or can immobilise nitrogen through increased microbial biomass (Moreau et al., 2019). Greater plant functional group diversity has been linked to improved carbon inputs to soils in natural and managed systems (Lange et al., 2019). Similarly to N, functional diversity can determine carbon input potential of plants and subsequent soil storage (Nielsen et al., 2011). Litter quality and root exudate profiles have been identified as major modifiers of organic components of soil carbon, although significant increases in soil carbon typically only occur after several years (Castellano et al., 2015; Huang et al., 2021).

Organic matter provides many benefits to soils such as reducing bulk density, increasing water retention and the mineralisation of organic matter provides soil nutrients such as inorganic forms of nitrogen for plants (Gregorich et al., 1994; Johnston et al., 2009). The ratio of C:N in soils can also indicate the state of certain soil processes. For instance a high C:N can indicate N immobilisation, whereas a low C:N can indicate the potential for N leaching (Cookson et al., 2007; Stockdale et al., 2002). C:N ratio can be influenced by the quantity and quality (C:N) of plant inputs such a litter or root exudates (Bardgett et al., 2014; Fisk et al., 2015; Zhang et al., 2023).

Increased plant functional group diversity, particularly with the inclusion of legumes is linked to increased P uptake and associated reductions in available P in soils during growing seasons. An experimental grassland indicated this could be enabled by plant driven increases in P bioavailability (Oelmann et al., 2011). Availability may be altered through the production of organic acids which can lower soil pH, as well as enzymes in plant rhizospheres such as phosphatases (Li et al., 2014). Changes in soil pH can also be caused by nitrification and litter decomposition (Haynes, 1983).

Decomposition of litter is affected by leaf quality and soil properties (Veen et al., 2015). Alleyway derived plant litter can be used as mulch to provide cover to stifle weed growth in the treeline, supply organic matter, and potentially accelerate decomposition of diseased pruning materials (Webber et al., 2022). Other mulching materials such as Straw and Grass and/or Legume mixes have been shown to provide benefits to orchard tree line soil as discussed in Chapter 2. Research has not yet considered litter from different mixtures of plant functional groups as mulch in an orchard context. The rate of litter decomposition can be increased or decreased depending on leaf traits such as litter quality and the microbial community composition of soil in which it is decomposing (Ball et al., 2014; Wardle et al., 2006). Studies have explored how functional traits effect rate of litter composition and found graminoids decomposed significantly more slowly than forbs, but N-fixing species only decomposed slightly faster than non-N fixing plants on average (Cornwell et al., 2008). The rate of mass loss between mixed litters has also been studied. Typically, mixtures show non-additive mass loss (~70% of mixtures), but synergistic and antagonistic relationships have also been reported (Handa et al., 2014; Porre et al., 2020).

This study aimed to assess the combined direct and indirect impacts of plant functional group diversity on soil health. Common soil health indicators were used to gauge soil health and ecosystem processes. Aboveground vegetation productivity was also determined and decomposition rate of litter from each vegetated treatment cover was measured. It was hypothesised that:

1. A higher number of functional groups in vegetation cover will significantly improve soil health indicators, namely lower bulk density and pH, and increase plant available nitrogen, microbial biomass nitrogen, total nitrogen, inorganic P, above ground primary productivity.
2. No significant differences will be found in total carbon, dissolved organic carbon, C:N or organic matter between treatments.
3. Two or more functional groups in litter will significantly increase rate of decomposition compared to grass.

5.2 Methods

Plant communities of five different ground covers were established in mesocosms (see appendix). Each community was representative of ground covers and species consistently observed in orchards from previous chapters (3 & 4). These communities provided four different functional group combinations: grass (G), grass and legume GL, forb and legume (FL), and grass, forb and legume (FLG). A bare ground treatment was also included. Four forbs, two legumes and one grass species were chosen due to their presence in experimental sites in chapters 3 & 4, they were comprised of: *Achillea millefolium, Centaurea nigra, Leucanthemum vulgare, Silene dioica, Trifolium pratense, Lotus corniculatus* and *Lolium perenne* (Table 5.1).

Table 5.1. Ground cover representative of wildflower mixes in orchard alleyways

Ground cover	Species composition (planting frequency)
Bare ground	N/A
Grass	*Lolium perenne* (25)
Grass and Legume	*Lolium perenne* (13), *Trifolium pratense* (6) and *Lotus corniculatus* (6)
Forb and Legume	*Achillea millefolium (5), Centaurea nigra* (5), *Leucanthemum vulgare* (5), *Silene dioica* (5) and *Trifolium pratense* (5)
Grass, Forb and Legume	*Lolium perenne* (5), *Achillea millefolium* (4), *Centaurea nigra* (4), *Leucanthemum vulgare* (4), *Silene dioica* (4), *Trifolium pratense* (4)

Each mesocosm was a 42 litre (38 x 38 cm, 40 cm depth) pot with a base of 10 cm of sharp sand topped with two layers of water permeable weed matting. This base was devised to aid drainage and restrict root growth beyond 35 cm replicating the typically shallow soils found in orchards. Above this, 35 cm of Norfolk Topsoil supplied by Baileys of Norfolk (Norfolk, UK) was used. The soil was a clay loam (pH 8.12:, C % 1.47, N % 0.12, and C:N ratio 11.97) that had been screened to 10 mm before packing.

Plants were grown from seed outdoors and transplanted into mesocosms. Each pot was planted with 25 individuals in a 5 x 5 grid with equal spacing (5.4cm). Planting frequencies are denoted in Table 5.1. Mesocosms were located at Hazelrigg Field Station, Lancaster University, UK for the duration of the experiment (July 2022 – July 2023). Pots were watered regularly throughout the experiment and vegetation was cut once in September with the clippings used in a decomposition assay.

After one year, mesocosms were destructively harvested. Aboveground vegetation from each treatment (see table 5.1) was cut to the soil surface and dried at 70 °C in a drying cabinet until mass was consistent. Soil was collected using an auger to 15 cm depth at five sampling points within each pot and mixed to form a composite sample. A bulk density sample was also collected from each pot. Soils were transported to a 4 °C cold store immediately after sampling. Composite samples were sieved to 2 mm in preparation for analysis.

5.2.1 Soil analyses

Dissolved organic carbon (DOC) was calculated by shaking 5 g of fresh in 35 ml Milli-Q water on orbital shaker for 10 minutes. The subsequent mixture was allowed to settle for 10 minutes and then filtered through Whatmann no.1 filter paper. DOC extracts were run on a Shimadzu TOC-L analyser (Shimadzu, Kyoto, Japan). Methods for determining soil moisture, organic matter, bulk density, Total C & N, C:N, Microbial N, Inorganic N &P, and pH can be found in chapter 3.

5.2.2 Decomposition assay

Clippings from the mesocosm experiment were collected in September 2022 and used in a decomposition assay. Fresh biomass rather than dried was used to simulate addition of clippings from alleyways to bare treeline soil. To calculate even application of biomass to each dish sub-samples of each species were dried at 70 °C for 48 hours and fresh/dry biomass ratios were calculated but these subsamples were not used in the decomposition assay. In each dish 1g of dry biomass was added in total. This 1g of dry biomass was divided equally between each species in a treatment (outlined in appendix in table 8.3).

Petri dishes were filled with 10 g of fresh treeline soil from a conventionally managed orchard site in Newington, Kent, UK (a study site from chapter 3 and 4). Plastic mesh of 1 mm^2 was placed over the soil with fresh biomass equivalent to 1 g of dry biomass from each cover placed on top (see table 8.3). The petri dish lid was secured with parafilm around ¾ of the circumference initially. Once fresh biomass had visibility dried, approximately one month after the experiment commenced, the entire circumference of the dish was secured with parafilm. Soil in the dishes was regularly watered using a pipette back to its initial mass. Assays were maintained in a Sanyo MIR 553 incubation cabinet at 21°C

Six months later remaining plant biomass fragments were retrieved from each dish. This was carefully collected from the mesh and soil using tweezers and dried at 70 °C for 48 hours before weigh to determine change in mass.

5.2.3 Statistical analyses

Data analyses were performed in R Studio (Version 3.3.0+, RStudio Team (2023); RStudio: Integrated Development for R. RStudio, Inc., Boston, MA, USA). Data were initially tested for normality of distribution and homogeneity. Any significant data sets were log transformed to conform to normality. An analysis of variance (ANOVA) was used for testing statistical significance, with independent variables represented by vegetation cover, and a dependent variable being a soil health indicator. This was followed by a Tukey's post hoc test for pairwise comparisons. A principal component analysis of all soil health indicators was produced to quantify the effects of vegetation cover treatments in mesocosms.

5.3 Results

5.3.1 Plant biomass

Total aboveground biomass (figure 5.1) was significantly different under treatments. In the grass only treatment aboveground biomass (38.5±4.45) was significantly lower compared to Grass + Legume (GL) (317±27.6), Forb + Legume (FL) (350±77.7), and Forb + Legume + Grass (FLG) (324±48.0). FL had a mean biomass around 10% larger than GL and FLG, but this was likely attributed to one outlier in GL.

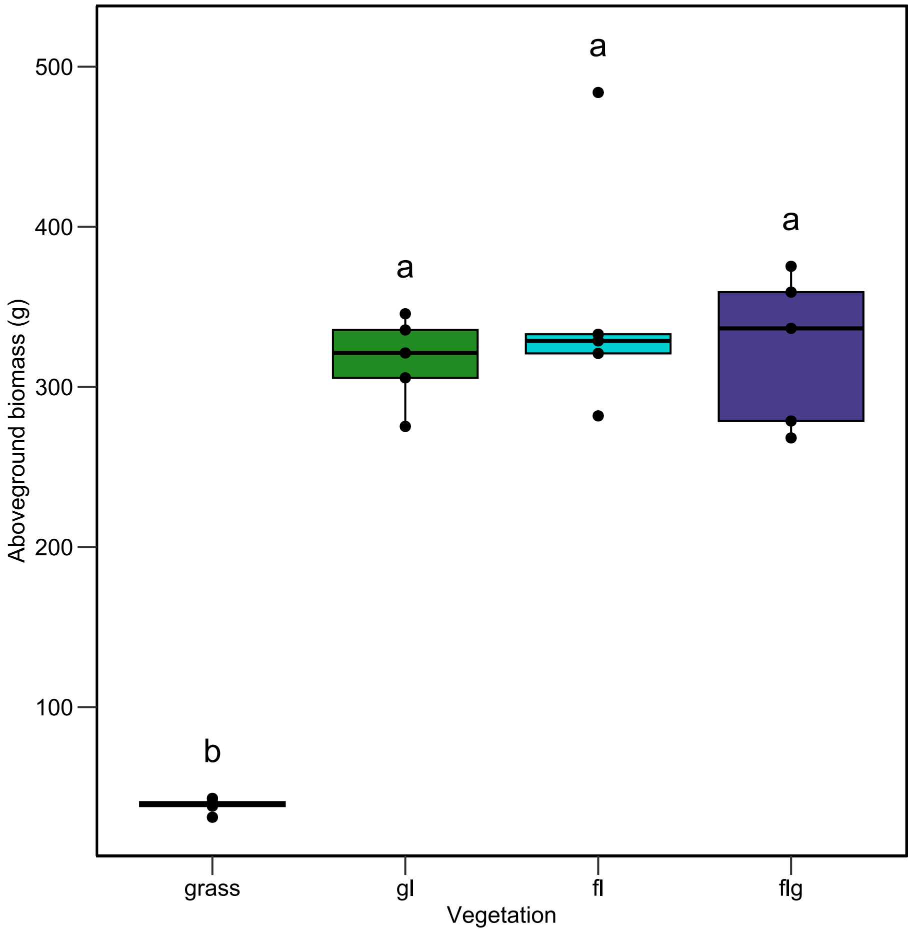

Figure 5.1. Box plots depicting Aboveground biomass of vegetation (g) from five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P>0.001*.

Separation of aboveground dry biomass revealed a significant decrease in Grass biomass as number of functional groups increased (see table 5.2). Grass only biomass was significantly lower in GL and FLG (P<0.05), and grass biomass in GL was significantly lower than FLG (P<0.05).

Table 5.2. A breakdown of dry masses of functional groups in each mesocosm treatment (N=5). ANOVA (P value): P<0.05.

Functional group	Grass dry mass (g)	Grass + Legume dry mass (g)	Forb + Legume dry mass (g)	Forb + Grass + Legume dry mass (g)
Grass	37.54 (± 4.45) ^{a}	18.48 (±6.61) ^{b}	-	4.202 (±2.07) ^{c}
Legume	-	307.55 (±18.34)	300.70 (±78.21)	268.92 (±60.39)
Forb	-	-	47.78 (±10.16)	50.62 (±31.27)

5.3.2 Soil chemistry

Plant available nitrogen was significantly different between vegetation covers (Figure 5.2) (P <0.001). Two covers, FL (0.50±0.39) and FLG (0.45± 0.35) were approximately 90% lower (P<0.001) in plant available nitrogen than bare soil (4.74±0.83), grass only (5.86±1.21) and GL (4.99±1.54) covers. Moreover, plots containing forbs were less variable compared to other covers. Grass only and GL both produced a non-significant (P>0.05) increase in plant available nitrogen compared to bare soil pots.

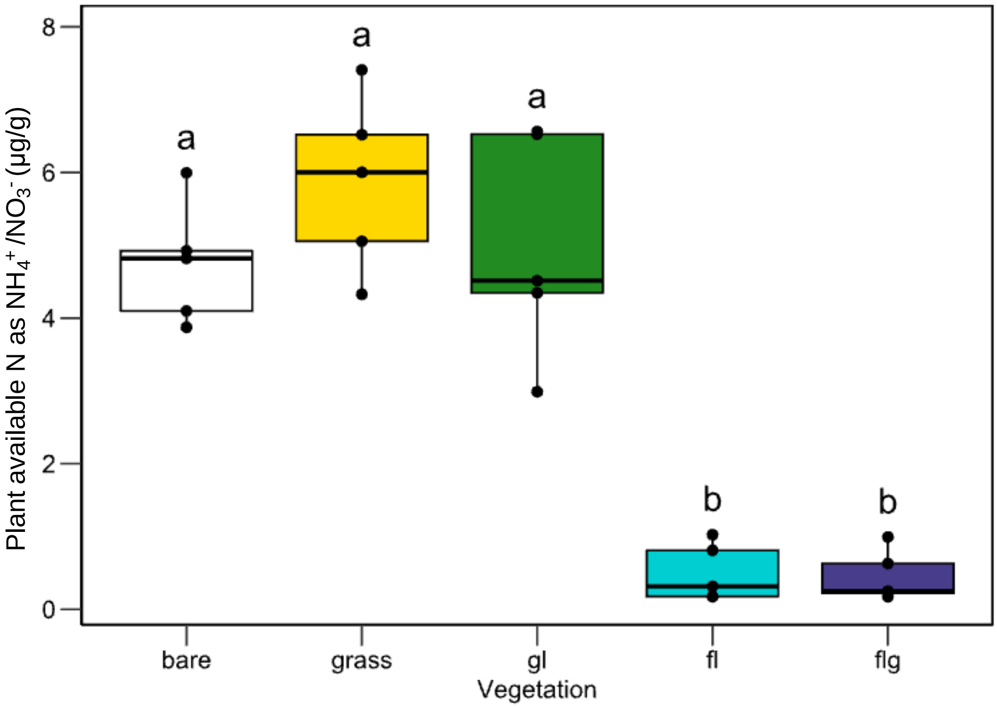

Figure 5.2. Box plots depicting plant available nitrogen (µg/g) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P<0.001***

Significant differences in total nitrogen (TN) were detected between vegetation covers (figure 5.3). Soil under grass only (21.2±1.11) cover had significantly higher TN than FL (16.2±3.31 P) and FLG (15.9±2.68). Bare soil (18.4±2.12) and GL (17.9±2.83) were not significantly lower than grass only or higher than FL and FLG.

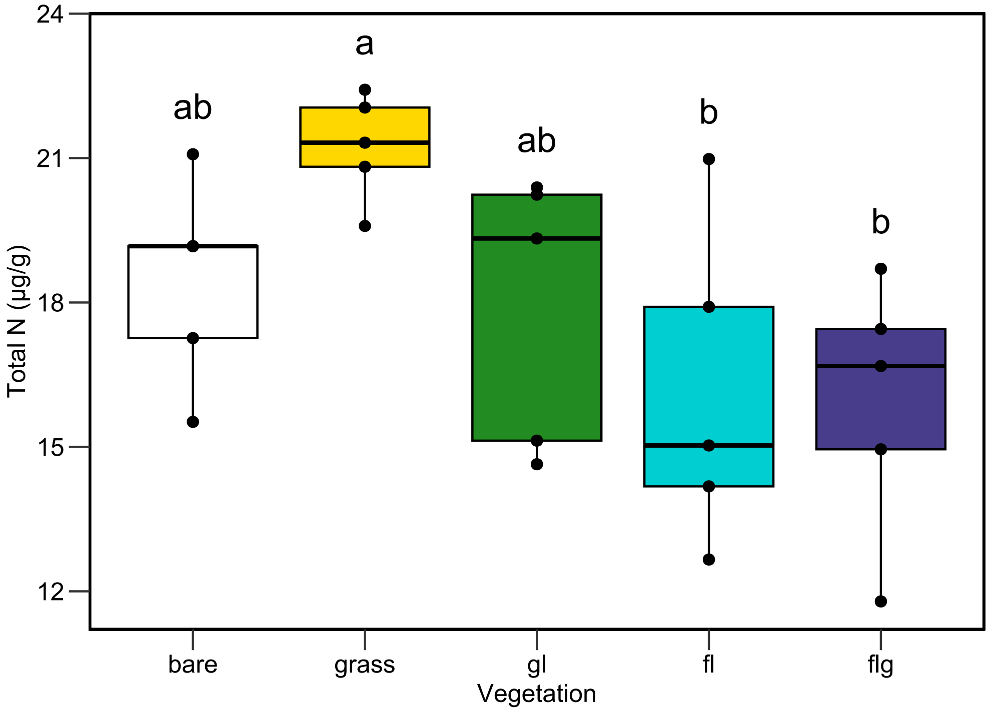

Figure 5.3. Box plots depicting Total nitrogen (µg/g) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P= 0.024*.

Significant differences (P<0.05) in microbial biomass nitrogen (MBN) were detected between vegetation covers (figure 5.4). Cover comprising two functional groups GL (24.8±4.47) and FL (24.4±5.95) were found to be significantly higher than bare soil (14.6±3.72) mesocosms. Grass only (16.8±4.41) and FLG (20.8±4.44) were not found to be significantly higher in MBN than the bare soil pot or significantly lower than GL or FL. Under FLG, MBN was equidistant between FL and Grass only.

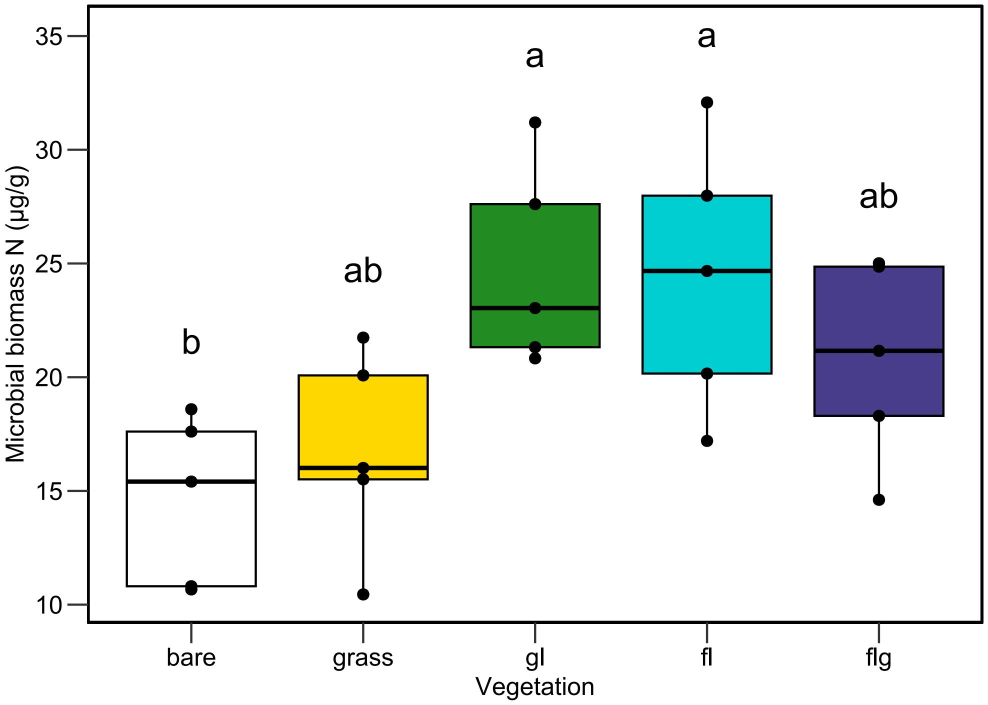

Figure 5.4. Box plots depicting Microbial biomass nitrogen (µg/g) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P= 0.007.

Dissolved organic carbon (DOC) was very variable under all covers and consequently, no significant differences were present (figure 5.5) (P>0.05).

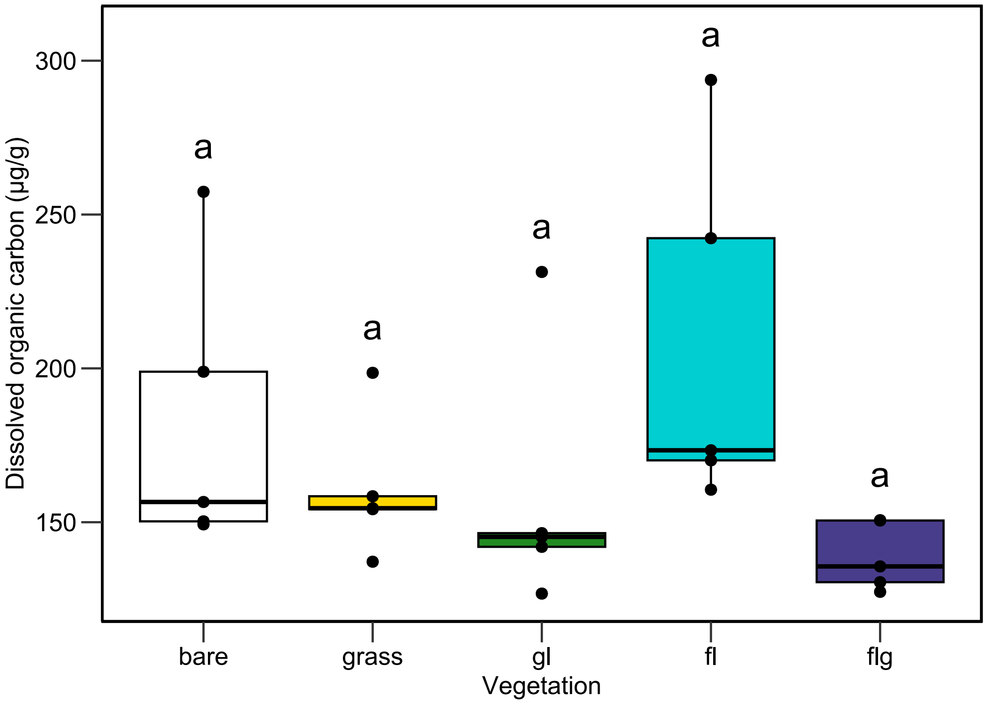

Figure 5.5. Box plots depicting Dissolved organic carbon (µg/g) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P= 0.07.

No significant differences were found in Total carbon (TC) under vegetation covers (figure 5.6). All covers were highly variable, with bare soil pots containing a considerable outlier.

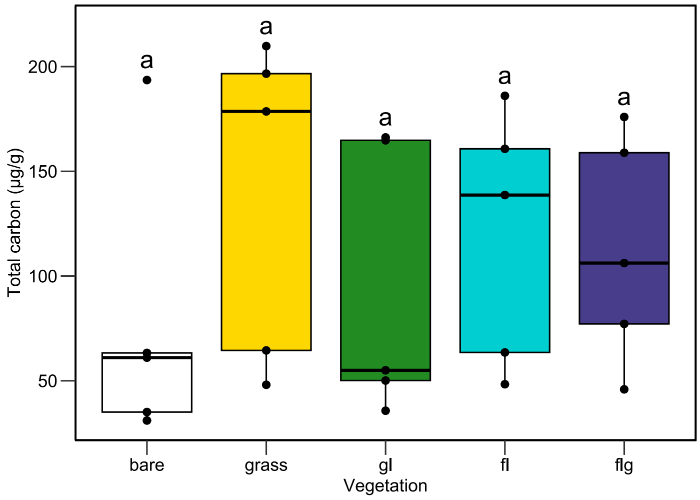

Figure 5.6. Box plots depicting Total Carbon (µg/g) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P= 0.519.

Under all treatments no significant differences were found in C:N (figure 5.7)

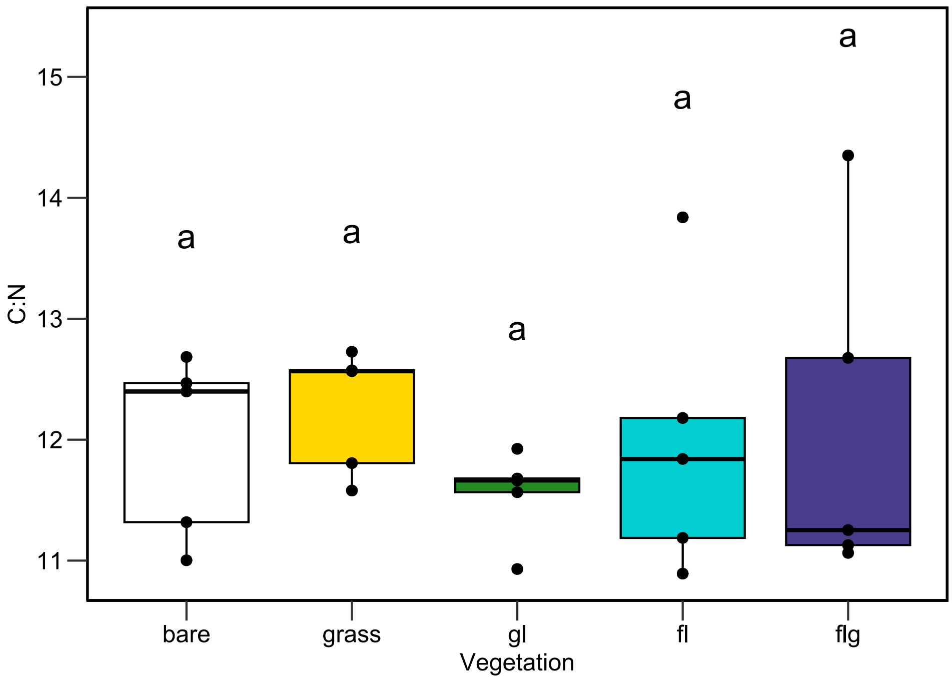

Figure 5.7. Box plots depicting C:N of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P= 0.816.

Organic matter (figure 5.8) was not significantly (P>0.05) different under any vegetation covers. The highest mean organic matter was recorded under FLG (4.64±0.73). Variability in organic matter increased with the number of functional groups present, except for bare plots. Grass only (4.16±0.15) and GL (4.08±0.29) had lower mean organic matter than bare soil (4.21±0.34) whereas FL (4.43±0.38) and FLG were higher.

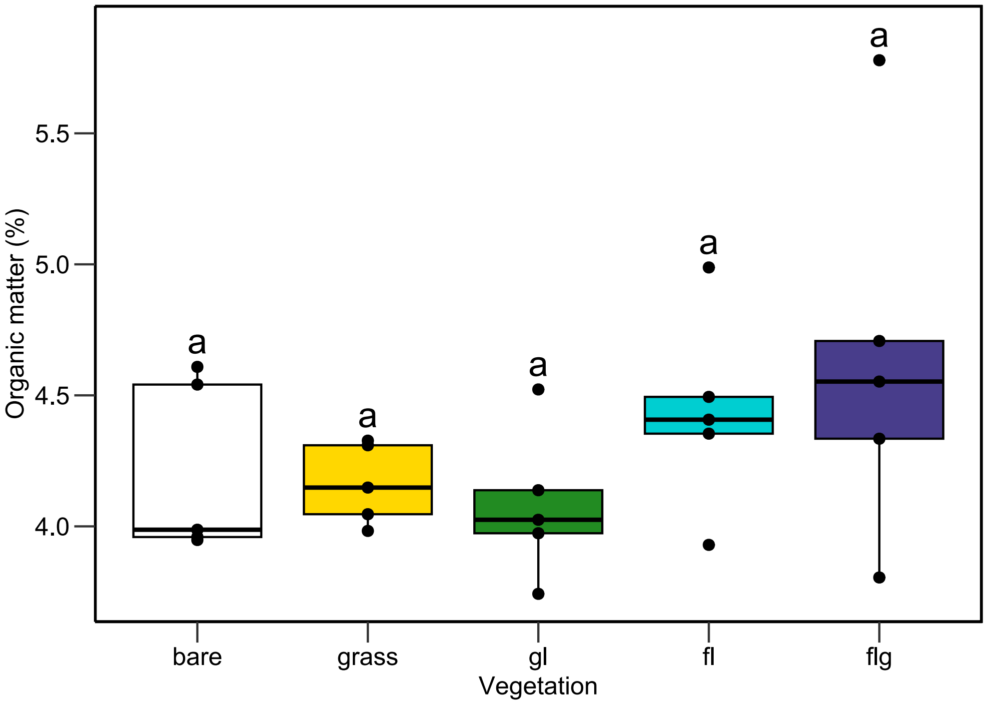

Figure 5.8. Box plots depicting Organic matter (%) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P= 0.253.

Pots with two or three functional groups had significantly lower pH (figure 5.9) than bare soil (8.12±0.05). Grass only cover mesocosms were also lower than bare soil (8.01±0.03), but not significantly so. Grass only cover mesocosm pH was not significantly higher than other vegetation covers.

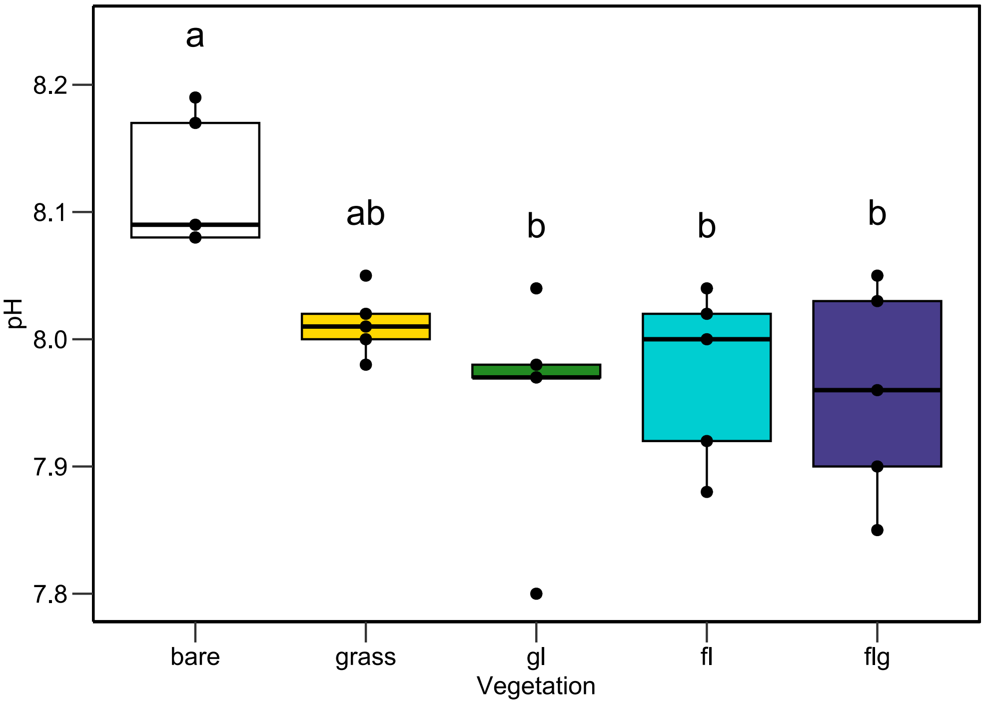

Figure 5.9. Box plots depicting pH of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): $P<0.05^{**}$

Inorganic P increased and decreased relative to bare soil plots (figure 5.10). However, no significant differences were found between all treatments. Plots containing forbs (FL 13.3±6.37 & FLG 10.9±2.56) increased inorganic P relative to Bare soil plots, whereas pots containing grass only without forbs (Grass 6.23±1.68 & GL 6.03±6.50) decreased inorganic P.

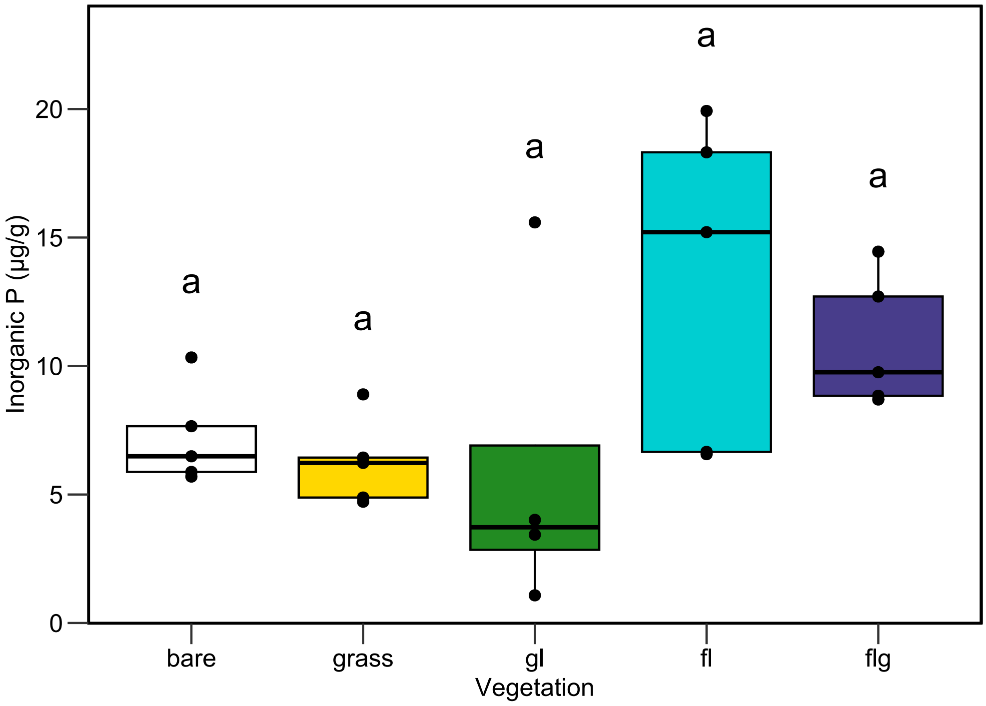

Figure 5.10. Box plots depicting Inorganic phosphorus (µg/g) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P>0.05 (ns)

5.3.3 Soil physical properties

Bulk density was significantly different under some covers. Soil under GL (1.18±0.07) had significantly lower bulk density than grass only (1.30±0.09) and Bare soil (1.33±0.04) plots (figure 5.11). Whilst GL had the lowest mean bulk density it was more variable than the other legume containing covers (FL 1.23±0.04 & FLG 1.25± 0.04). Bare soil plots had the highest mean bulk density although the magnitude of differences between treatments was small.

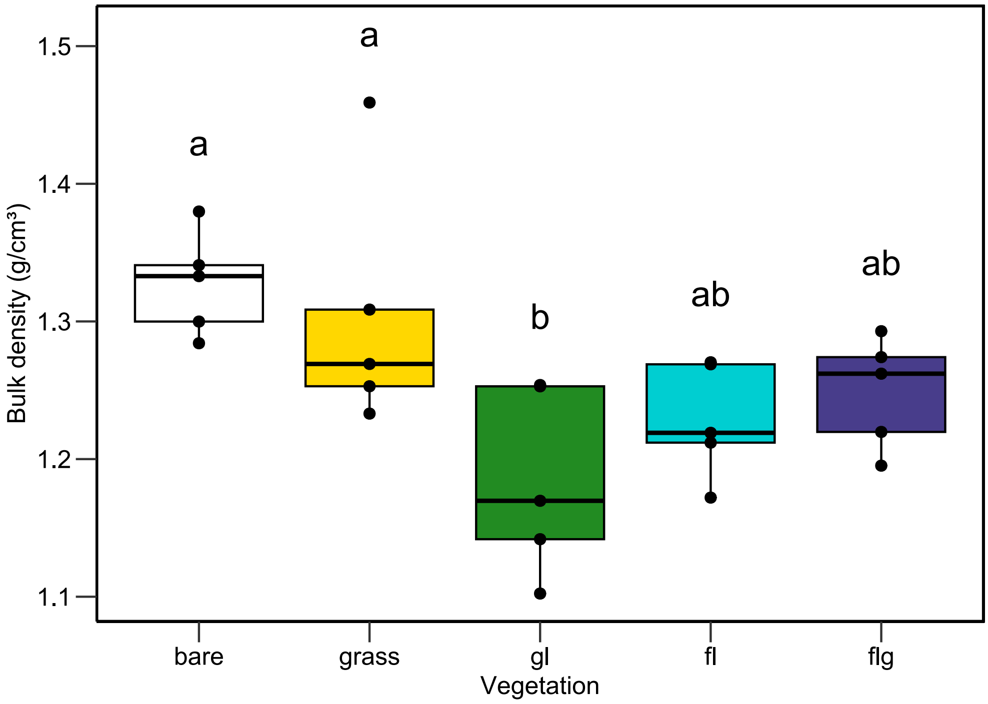

Figure 5.11. Box plots depicting Bulk density (g/cm^3) of soil under five different vegetation covers in mesocosms. Covers were: Bare soil, Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P= 0.007**.

5.4 Principal component analysis of soil health indicators

The combined axes of the PCA explain 47.6% of the variation within the dataset (figure 5.12). PC1 explained 31.5% and PC2 explained 16.1%. Treatments containing two or more plant functional groups are distinct from treatments containing one (Grass) and bare ground mesocosms by aboveground biomass, microbial biomass nitrogen, and organic matter. Bare ground plots were typified by pH and bulk density. Grass only mesocosms were associated with total nitrogen and available nitrogen. Overall, soil health indicators in mesocosms containing two or more functional groups were distinct from Bare and Grass mesocosms. Loadings revealed correlations between soil health indicators. Organic matter and inorganic phosphorus were negatively correlated with total nitrogen and plant available nitrogen. Also, bulk density and pH were negatively correlated to above ground biomass and microbial biomass nitrogen.

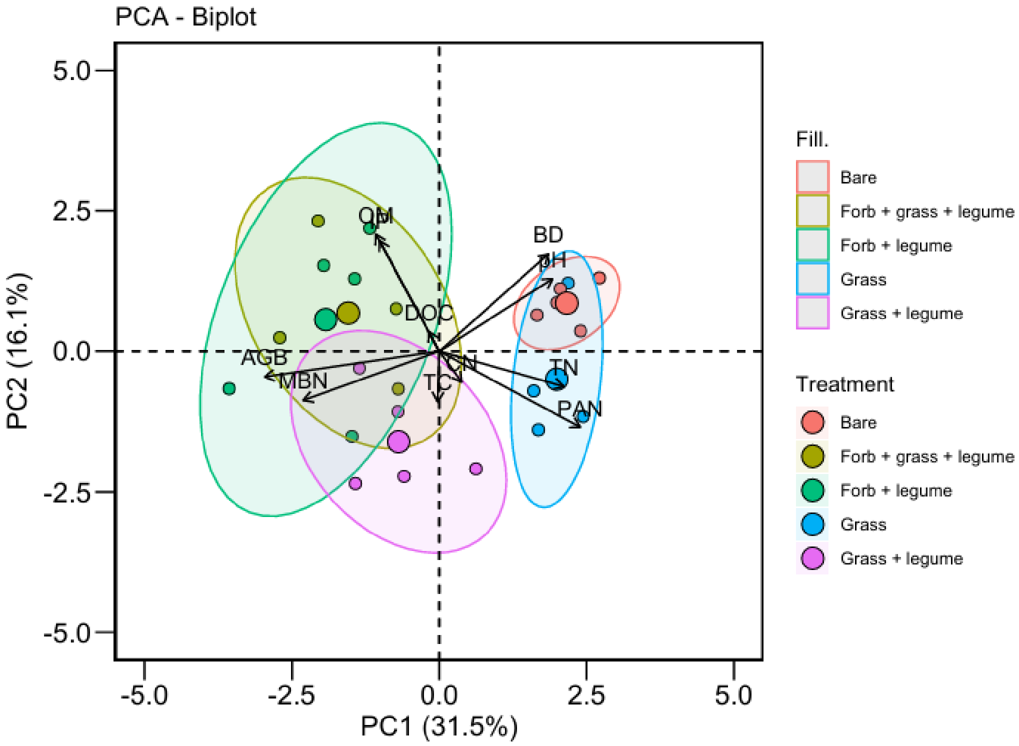

Figure 5.12. A principal components analysis of the relative abundance of all soil health indicators in mesocosms with five different plant cover treatments. PC1 explains 31.5% and PC2 explains 16.1% of variability of soil health indicators. Soil health indicators are denoted by bulk density (BD), CN (CN), MBN (microbial biomass nitrogen), OM (organic matter), IP (inorganic phosphorus), total nitrogen (TN), AGB (above ground biomass), available nitrogen (AN), pH, total carbon (TC), and dissolved organic carbon (DOC). Soil health indicators were explained by the presence of two or more plant functional groups in mesocosms.

5.4.1 Litter decomposition

Leaf litter decomposition produced significances between treatments (P>0.001) (figure 5.13). Mass of litter mixtures FL (0.67±0.06), GL (0.67±0.06) and FLG (0.66±0.06) decreased by two thirds compared to the Grass only (0.48±0.05) only mixture deceased by 48%. Litter loss was consistent across all pots with two or three functional groups.

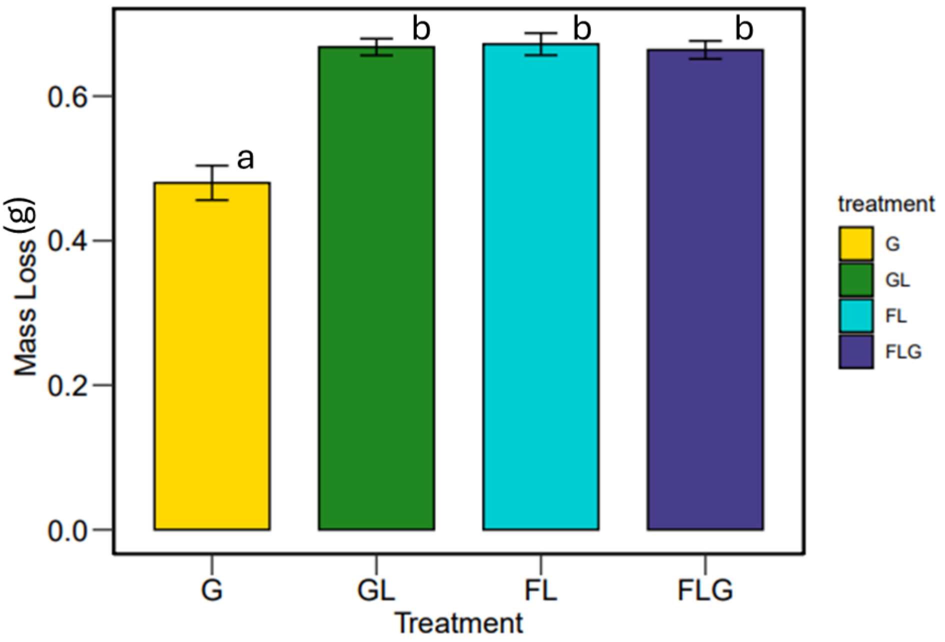

Figure 5.13. Bar chart depicting mass loss during leaf litter decomposition experiment. Initial leaf litter dry mass 1g. Leaf litter was derived from four covers: Grass only, Grass+ Legume (gl), Forb + Legume(fl), and Forb + Legume + Grass (flg). (N=5). ANOVA (P-value): P> 0.001***.

5.5 Discussion

This study aimed to assess the potential benefits of plant functional group diversity on soil health in the absence of external management pressures found in orchards. The expected benefits were outlined in two hypotheses, with an additional hypothesis included to outline my expectations for a related litter decomposition experiment.

Our first hypothesis, A higher number of functional groups in vegetation cover will significantly improve soil health indicators, namely lower bulk density and pH, and increase plant available nitrogen, microbial biomass nitrogen, total nitrogen, inorganic P, and also above ground primary productivity can be accepted with conditions.

Aboveground productivity was significantly higher in pots with two or more functional groups. Productivity in these pots was largely driven by *Trifolium pratense.* The inclusion of a third functional in FLG appeared to have no effect on aboveground biomass. Studies have found mixed results when comparing plant diversity and functional diversity to productivity. Some suggest an overyielding effect linked to greater resource acquisition potential of diverse communities (Grange et al., 2021). Whereas others suggest overall productivity is determined by specific highly productive species or certain combinations of species (Hooper & Dukes, 2004; Sanderson, 2010). In species richness-productivity experiments *T. pratense* has been identified as a species that exerts significant positive influence on productivity (Hector et al., 1999) .

Forbs and legumes exerted significant effects on soil N pools. Forbs (FL & FLG) significantly decreased plant available nitrogen. This relationship was likely driven by specific functional traits possessed by Forb species in mixtures as they were not the dominant functional group (Hooper & Vitousek, 1998). Other studies have highlighted high rates of N uptake where a specific combination of legume and grass/forb is present, rather than functional group or species richness (Pirhofer-Walzl et al., 2012; Rasmussen et al., 2013). Forbs have been linked with decreased plant available N, although with a difference in seasonality from this study (Hooper & Vitousek, 1998).

Grass only cover showed the highest mean soil TN. It was only significantly higher than forb containing covers which may further showcase the role of forbs in controlling inorganic N pools. High levels of TN in soil under grass only cover may be attributed to relatively poor levels of productivity and lower nitrogen uptake in comparison to other covers. This effect was likely compounded by removal of litter, where nitrogen would have been returned through decomposition(Abalos et al., 2019; Bonanomi et al., 2006). Litter removal has been shown to decrease N pools in other systems (Xu et al., 2021).

Legumes significantly increased microbial biomass N in GL & FL relative to bare plots. The reduction in microbial biomass N in FLG (4) could be attributed to a reduced number of legume individuals in the cover established compared to FL (5) and GL (6) rather than the exclusion of *L. corniculatus.* Cover containing all functional groups (FLG) may have exhibited improved N uptake through increased complementarity (De Deyn et al., 2009; Hooper & Vitousek, 1998), allowing for more effective uptake of available soil N resources. Although this was not reflected in aboveground biomass, which could be associated to an overall reduction in nitrogen use efficiency (Lü et al., 2014). Legumes were linked to a significant decline in soil pH in pots with two or more functional groups. An unbalanced uptake of cations in comparison anions in studies have been linked legumes with a greater soil acidification potential than other functional groups, along with organic acid exudation (Haynes, 1983). A reduction of pH in this study under cover with two or three functional groups could indicate functional group diversity may increase availability of essential plant nutrients in soil such as potassium, calcium, and magnesium (Furey & Tilman, 2021).

Unlike soil N indicators, inorganic P was not significantly different under any treatments. Functional group richness and species richness have been linked to increased soil P availability due to increases in phosphatase enzyme activity (L. Li et al., 2014). Studies have shown that this increase in enzyme activity can counteract plant uptake to maintain inorganic P in soil (Menezes-Blackburn et al., 2018; Oelmann et al., 2011). Furthermore, litter returns plant bio-available sources of P to soil and could act an effective source of inorganic P as a mulch (Oelmann et al., 2011) but these changes were not observed here. Inorganic P was likely contained in plant biomass as mesocosms were harvested during the growing season.

Bulk density significantly decreased under GL compared to bare and grass only plots, whereas FL and FLG were similar to GL. Similarly, a study by Gastine *et al* (2003) found that soil bulk density was lowest under cover containing two functional groups (GL). However, the same study also found covers containing three functional groups (FLG) had significantly higher bulk density, whereas higher bulk density was found under grass only cover in this study. Contrary findings of bulk density under grasses in these studies could be related to differences in root traits of grass species used by Gastine *et al*, such as lower root tissue density (Freschet et al., 2017).

No additional benefits to soil health in the previous indicators were found comparing between two (GL and FL) and three functional (FLG) group treatments. This may be because there is an overlap between the functional roles of functional groups and their effects on soil processes or the species do not significantly affect soil processes (Hooper et al., 2005). Similarities in mesocosms with two or more functional groups and the effects on soil health were highlighted in figure 5.12. These treatments were distinct from Bare and Grass plots. However, without Grass + Forb or Forb only treatments, whether this effect is driven by functional group number of the presence of legumes cannot be revealed. However, wildflower alleyway cover without legumes was not observed at any sites in chapters 3 & 4.

In this chapter bulk density and pH are positively correlated, whereas in chapter 3 they are negatively correlated possibly indicating an effect of orchard management which was not present in this study, such as compaction from trafficking. Other relationships between soil health indicators in chapters 3 & 5 were similar.

The second hypothesis, no significant differences will be found in total carbon, dissolved organic carbon, C:N or organic matter between treatments can be accepted. Across carbon related measures (DOC, TC, OM, C:N) no significant differences were found. Studies reporting increases in soil carbon measures associated with plant diversity typically find increases over several years. Multiple biodiversity ecosystem function experiments have found carbon storage relationships strengthen overtime (Cong et al., 2014; Fornara & Tilman, 2008). Mechanisms such as complementarity of plant communities increasing over time have been suggested, whereas other studies have indicated improvements in soil biota community growth such as decomposers and arbuscular mycorrhizal fungi (Eisenhauer et al., 2012). Removing clippings from pots may have also restricted any potential increases in soil carbon. Plant litter is an important carbon input to soil. A meta-analysis investigating litter removal across ecosystems found soil carbon in topsoil (0cm-20cm) typically decreases where litter is removed. However, no effect was found on soil C content and stocks in mown grasslands (Xu et al., 2021).

The third and final hypothesis predicted that two or more functional groups in litter will significantly increase rate of decomposition compared to grass. Mixed litters showed increased rate of decomposition compared to grass monocultures in this study. Other studies have shown similar increases with increasing plant functional diversity (Scherer-Lorenzen, 2008; Ward et al., 2015). It is likely legume litter was accelerating overall decomposition through higher quality/lower C:N litter (Cornwell et al., 2008; Handa et al., 2014). Very similar loss of litter mass across FL, GL, FLG despite increasingly diluted masses of legume material indicated legume litter has a strong effect on decomposition rates of other litters. In this study aboveground productivity and litter decomposition rate can be linked with rate of both being consistent where *T. pratense* was present. Increased rate of decomposition may be less effective for supressing weeds in tree rows, as evidenced by Webber et al (2022) where grass + legume mulch was found not supress weeds more than a control treatment (no mulch) and significantly less than straw. However, it was suggested application of grass + legume muscles could accumulate over several years for a similar suppression effect as straw.

5.6 Conclusions

In conclusion, some significant differences were detected in soil health indicators under covers with two or three functional groups. This indicates that under the appropriate management intensity, wildflower strips have the potential enhance soil health. Despite the relatively short period of time elapsed from establishment to harvest in this study, evidence of the controls certain functional groups or increase in functional groups exert upon soil health indicators is displayed. Predictably, changes in N cycling were more pronounced, consistent with other studies of similar timescales. Despite the dominant effect of *T. pratense* aboveground primary productivity and litter decomposition, the importance of non-dominant biomass was highlighted such as the control of inorganic nitrogen in forbs and shows how diverse species cover can influence soil health.

6. General Discussion

This thesis investigated the effects of wildflowers strips on soil health in intensive apple and pear orchards. Following an initial literature review, chapter 2 reviewed the effects of intensive orchard management on soil health. Chapter 3 benchmarked soil health in sites planted with and without wildflower strips seven years after establishment. Following this, Chapter 4 explored responses (CO_2) of orchard soils under conventional and wildflower strip alleyway cover to drought and rewetting. Finally, chapter 5 studied the effects of representative orchard ground covers in mesocosms on soil health and how plant functional diversity affected rates of litter decomposition.

This chapter aims to summarise key findings across data and review chapters and discusses interactions between their results. Next, findings are related to the future of apple and pear orchards in Great Britain with a focus on climate change and the prospects for soil health. Areas for future studies of plant diversity and management intensity in orchards are then highlighted.

6.1 Direct effects of species composition on soil health

Wildflower strips are sown in orchards with the intention of increasing plant diversity. More specifically, the intention is to increase the number of functional traits to provide or enhance ecosystem processes/services. In chapter 3, only a small difference in species composition was observed, with a clear dominance of *L. perenne* in both wildflower sown and conventional orchard alleyways.

Results from the benchmarking study (chapter 3) highlighted limited differences in soil health indicators under conventional and wildflower strip sown alleyways. The lack of differences detected between covers could be to several reasons. One explanation could arise from the mass ratio hypothesis which dictates ecosystem function is largely determined by the traits of the dominant biomass (Grime, 1998). Under both covers grass had become the dominant biomass and could have had largely determined ecosystem function and subsequently soil health. Alternatively, the difference between species compositions may not have been sufficient to evoke any differences. Increases in plant productivity and the associated benefits to soil function require sustained diversity and increasing species richness has a decreasing benefit from even two or three species (Fridley, 2002; Hooper & Dukes, 2004). Grass with the inclusion of some weed species in the conventional (spontaneous) cover may be sufficient diversity for a comparable community effect on soil processes. Finally, functional traits in plant species found in conventional and wildflower strip alleyways could provide the same effects on ecosystem function (Grime, 2006). However, this is unlikely, as evidenced in the mesocosm study using the same plant species (chapter 5), where functional groups were found to effect soil processes. Legumes significantly increased microbial biomass nitrogen and forbs significantly decreased plant available inorganic nitrogen sources in soil.

Wildflower establishment method at sites used in chapter 3 (benchmark) were not dissimilar to approaches suggested in chapter 2 by Carvell *et al*. Although, the wildflower strips were considerably older when sampled in this study and there was not any vegetation cover data available to see if establishment over three years was comparable to Carvell *et al*. Additionally, wildflower strips in chapter 3 were not reseeded following initial establishment, which studies have suggested may be essential for maintaining wildflower strip diversity over time (Schmidt et al., 2020; Schmied et al., 2023). This would be important to reap potential benefits to soil health as many species' composition derived enhancements to soil processes are found to increase over time (Hector, 2022; Isbell et al., 2011).

Management of alleyway vegetation can directly affect species composition. Mowing can be beneficial or detrimental to species diversity depending on frequency and species sensitivity to mowing (Valkó et al., 2012; Yang et al., 2019). Mowing frequency at sites may not have been optimised for species diversity. Adopting more conservative mowing regimes and reseeding highly productive species such as *Trifolium pratense* may sustain wildflower strip diversity beyond initial establishment (chapter 5).

6.2 Indirect effects of species composition on soil health

Earthworms are broadly recognised for their capacity to provide vital soil processes (Lemtiri et al., 2014; Medina-Sauza et al., 2019). In chapter 3 no differences in worm count were found between spontaneous cover and wildflower strips. At some sites no worms were found during sampling indicating deleterious effects of management on earthworms, particularly functional groups commonly found in upper soil horizons such as Epigeic and Endogeic (Huang et al., 2020). In another study, plant species identity rather than diversity appears to control earthworm diversity, with legume *Trifolium repens* a notable species for supporting diverse earthworm communities (Piotrowska et al., 2013). Higher N inputs however are linked with decreased earthworm biomass and abundance even where favourable plant species are present (Piotrowska et al., 2013). Herbicide application and compaction also effect earthworm abundance and biomass, similarly, as described chapter 2 (management). Microbial biomass nitrogen also showed no differences under wildflower or conventional cover, which may also be linked to high N inputs (Treseder, 2008).

In my mesocosm experiment (chapter 5) *T. pratense* was found to exert significant effects on aboveground primary productivity, pH, bulk density, and litter mass loss. *T. pratense* has been found significantly contribute to ecosystem processes in other studies (Hector et al., 1999). Under intensive management (high N) the competitive advantages of *T. pratense,* N fixation, may be reduced as its ability to forage for N resources is inferior to other functional groups (non-legume) with more fine root mass. Additionally, in mown areas, grasses typically exhibit a superior rate of regrowth which may further diminish the competitiveness of legumes like *T. pratense* and reduce the influence of their functional traits (Oelmann et al., 2015).

Orchard management perturbs vegetation covers and selects for species adapted to management interventions. Furthermore, high input of agrochemicals is not synonymous with high species richness (Isbell et al., 2017). Compaction from trafficking in orchards may also negate benefits provided by combining functional groups in mesocosms (chapter 5). Compaction of soil can inhibit root growth and architecture which can affect root exploration and scavenging which may increase competition in accessible soil. This may reduce potential niche complementarity benefits associated with increased plant diversity (Griffiths et al., 2022; Mahaut et al., 2020).

6.3 UK orchard soil resilience in the face of climate change

Climate change represents a major threat to UK orchards as many are situated in areas predicted to be heavily affected by future drought, such as Kent (see chapter 4). Soil structure and organic matter are major controls of soil drought resistance. Good soil structure enables water infiltration and, along with organic matter, retention of water (Bot & Benites, 2005). Rooting depth is also improved in soils with more open pore structures (Gao et al., 2016). Increased productivity of aboveground biomass could increase transpiration losses under diverse covers, but soil water scavenging could be superior (Chen & Coughenour, 2004; O'Keefe et al., 2019).

Evidence suggests more diverse vegetation cover should support resilience. However, in chapter 4, no significant differences were detected in the responses (CO_2) between covers under drought and rewetting. Similarly to the benchmark chapter, potential differences between species compositions of conventional and wildflower sown alleyways may not have been substantial enough. Studies (Isbell et al., 2011; Tilman et al., 2014) point to large relative increases in species richness required to evoke desired impacts upon ecosystem function. Increasing diversity in orchard alleyways to achieve a desired effect may require sustained species richness improvements. Mixed effects of plant diversity have been found on improving drought resistance and resilience of soil microbes (Li et al., 2022; Orwin & Wardle, 2005).

Orchard soil capacity to show drought resilience may have been compromised by disturbance of cumulative selection pressures such as compaction or mowing frequency (see chapter 2). Soil fungal communities in apple orchards under integrated pest management and organic management have been shown to be more vulnerable to agricultural management such as herbicide and fertiliser inputs than bacterial networks when compared in diversity indices, community composition, and functional groups (Hulsmans et al., 2022). Bacterial networks have been observed to be less stable under drought stress (de Vries et al., 2018). Management intensity reductions alongside sustaining increased plant diversity may be necessary to improve the resistance and resilience of soil alleyway microbial communities to drought.

Under prolonged or severe drought conditions, there is a limit to how much effects of drought can be limited without water supplementation, especially in tree rows where bare soil is typically maintained. Irrigation is an effective strategy to deal with low available soil moisture, but water availability cannot be guaranteed under future scenarios where demands are increased across multiple sectors (Harrison et al., 2015). Irrigation methods such as high frequency drip line irrigation likely provide the most efficient use of water resources and are adopted in most newly planted orchard systems (Manning., 2021).

Mediterranean orchards can provide a looking glass into the future for effects of climate change in the UK. Drought currently effects Mediterranean fruit crops with yields impacted in recent years (Fraga et al., 2020; Lagacherie et al., 2018). The most viable strategies for adapting to drought have been through irrigation management (Jovanovic & Stikic, 2018). Research approaches from areas where drought is currently prevalent will likely inform increasingly more drought prone areas of orchard production.

6.4 The Future of UK Orchards

The short-term outlook for apple and pear orchards in the UK is poor. Prices offered for British apples and pears are too low relative to production costs, as reported by grower-funded organisation British Apple and Pears (Manning., 2021). This can be evidenced by recent widespread grubbing even in areas renowned for production such as Kent. This fits within a trend of orchard decline in the UK over the last few decades (see chapter 2)

This poses the question of whether growers will invest and adopt management innovations and accept financial risks or transition away from orchards. It makes reductions in management intensity, possibly at the cost of yields, to improve sustainability difficult to accept. High management intensity and the provision of soil ecosystem services by wildflower strips are currently not compatible (see chapter 3). If management intensity is reduced to improve sustainability and increase viability of wildflower strips potential decreases in yield may not be accepted by growers. Payments for Sustainable Farming Incentives through the Environmental Land Management schemes (ELMs) whereby growers may receive payment for in-field flower rich strips (£673 per ha) in permanent crops and other sustainable management changes such as conversion to organic productions methods is likely to provide the best incentive for implementing wildflower strips and reductions in management intensity in orchards (DEFRA, 2023). With lifetime production costs estimated by growers between £45,600-£86,450/tonne (Manning, 2021), it remains uncertain whether grower payments under new ELMs, including interventions such as in-field wildflower strips, will provide sufficient compensation to encourage uptake.

6.5 Potential for improving soil health in UK orchards

There are reasonable prospects for improving soil health as current orchard management is not optimised for soil management. Soil health needs to be documented under practical application of different management strategies to ascertain the most beneficial changes that can be made improve soil health (Rinot et al., 2019). Further research is needed to provide an evidence base to support decision making.

Understanding which practices are most damaging to soil health, and which practices can be altered within viable management strategies is important. Trade-offs between yield and the cost associated with management, particularly with agrochemical inputs appears to be the most likely direction for improving soil health (Mandal et al., 2020; Tahat et al., 2020). Maximising ecosystem services to produce fruit, but also other ecosystem services such as carbon sequestration that can be financially rewarded through schemes such as ELMs may enable a transition to less intensive management (DEFRA, 2023; Demestihas et al., 2017a).

Delivery of inputs on a more prescriptive basis with growers able to be more reactionary than blanket with their input approach. Many examples point towards precision application of pesticides after detection of pathogens with novel biosensors, but a similar method can be adopted for soil moisture (Zhang et al., 2021). Tree row soil can be irrigated to critical values of soil moisture to enable an optimised application of water (Jovanovic & Stikic, 2018).

Reduction of trafficking through more precision approaches offers an opportunity to reduce the impact of intensive management on soil health. New technologies such as multi row sprayers with precision application can reduce trafficking in rows by concentrating trafficking into 1/3 of alleyways rather every alleyway and overall inputs can be reduced by decreasing drift of pesticides (Grella et al., 2021; McCoy et al., 2021).

6.6 Further work

Effects of management intensity on apple orchard soil health is an area of research that requires further attention. Comparisons exist between conventional and organic orchard systems but there is a large disparity in management methods and inputs (Cook, Magan, Robinson-Boyer, et al., 2023; Hulsmans et al., 2022). Looking at management practices that target an integrated approach could offer examples of systems that target high productivity, but with a less intensive inputs. Measuring a concise suite of soil health indicators (such as organic matter, bulk density, and pH) across a broad range of sites (soil type, climates, and land use history) from different regions in the UK could elucidate the impacts of different management strategies on soil health (Feeney et al., 2023). Additionally, a similar approach to chapter 3, where wildflower and conventional covers were maintained within the same orchard site could provide valuable insights to effects of management intensity. This approach could identify where gains in soil health can be made in relation to management. Quantifying the effects of management intensity along a gradient may reveal thresholds for which increased plant diversity may be able to be sustained and/or enhance soil ecosystem processes (Martin & Sprunger, 2022; Ward et al., 2016).

Further assessments of diversity beyond vegetation may reveal functional changes in soil ecosystems under different management (Chen et al., 2019; Nielsen et al., 2011). For example, monitoring of microbial and faunal diversity in orchard soil may highlight appropriate levels of management intensity required to sustain robust soil food webs with high levels of functional redundancy (Daelemans et al., 2022), potentially a critical development for orchard systems under future stressors such as increased intensity and duration of drought (Cook, Magan, Robinson-Boyer, et al., 2023).

As discussed in chapter 4 climate change is projected to impact many orchard growing regions in the UK, such as Kent where water shortages already occur. Field drought studies may provide a view of orchard soil responses to real drought events. Whilst field conditions lack precision of microcosms regarding treatment (e.g. water holding capacity control), they may allow more representative responses to drought, especially where vegetation is maintained during treatment (Canarini et al., 2017).

Investment into methods to establish and maintain wildflower strips in orchards at meaningful timescales for soil health improvement would be valuable (Carvell et al., 2022). Approaches have been outlined but not necessarily at the scale beyond three years. Difficulties in establishing and maintaining wildflower strips has been documented in literature, and in chapters 3 & 4 (Schmidt et al., 2020; Schmied et al., 2023). Studies to determine optimal methods for establishing diverse vegetation covers across a range of soil types would be incredibly valuable in efforts to supply more ecological management solutions. Associated management strategies to maintain diversity once established would be equally valuable (Schmidt et al., 2020; Schmied et al., 2023).

6.7 Conclusions

This thesis explored the potential benefits of wildflower strips to improve soil health in commercial orchards, alongside current management practices. I provided evidence that wildflower strips alone are not sufficient to improve soil health in intensive orchard systems. Experiments benchmarking soil health indicators in several sites and responses of orchard soils to drought produced limited differences between conventional and wildflower strip managed alleyways. Subsequently, I designed a study comparing representative covers from orchard sites at mesocosm scale. Results suggested an improvement in some soil processes, particularly nitrogen cycling, indicating that positive benefits to soil health under wildflower strips may be stifled by current intensive management practices. This claim was supported by a review of orchard management where the current strategies were associated with negative effects on soil processes and highlighted where direct and indirect effects of species composition in wildflower strips could be supressed.

Moving forwards, further research should target where critical thresholds exist along gradients of management intensity. The role of biodiversity beyond wildflower strips, such as soil food webs may also improve understanding of the effects of plant diversity and management intensity in an orchard context, and how soil health may be improved in apple and pear orchards.

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8. Appendices

8.1 Appendix

Table 8.1. Species list for plots sampled in Chapter 3.

Site	Plot	Species
Newington	Conventional	*Lolium perenne* *Taraxacum officinale* *Galium sp* *Rumex obtusifolius* *Trifolium pratense* *Veronica sp* *Lotus corniculatus* *Cerastium fontanum* *Leontodon hispidus*
Newington	Wildlflower	*Lolium perenne* *Dactylis glomerata* *Taraxacum officinale* *Achillea millefolium* *Galium sp* *Rumex obtusifolius* *Trifolium pratense* *Veronica sp* *Silene dioica* *Lotus corniculatus* *Centaurea nigra* *Cerastium fontanum* *Leontodon hispidus*
Eastside	Conventional	*Lolium perenne* *Taraxacum officinale* *Rumex obtusifolius* *Trifolium pratense* *Veronica sp* *Lotus corniculatus* *Cerastium fontanum* *Leontodon hispidus*
Eastside	Wildlflower	*Lolium perenne* *Dactylis glomerata* *Taraxacum officinale*

		Achillea millefolium *Rumex obtusifolius* *Trifolium pratense* *Silene dioica* *Lotus corniculatus* *Centaurea nigra* *Cerastium fontanum* *Leontodon hispidus*
Broomside	Conventional	*Lolium perenne* *Taraxacum officinale*
Broomside	Wildlflower	*Lolium perenne* *Taraxacum officinale* *Trifolium pratense* *Leontodon hispidus*
Shelving	Conventional	*Lolium perenne* *Taraxacum officinale*
Shelving	Wildlflower	*Lolium perenne* *Taraxacum officinale* *Trifolium pratense* *Leontodon hispidus*
Parasol	Conventional	*Lolium perenne* *Dactylis glomerata* *Taraxacum officinale* *Rumex obtusifolius* *Trifolium pratense* *Leontodon hispidus*
Parasol	Wildlflower	*Lolium perenne* *Dactylis glomerata* *Taraxacum officinale* *Achillea millefolium* *Rumex obtusifolius* *Trifolium pratense* *Silene dioica* *Leontodon hispidus*
Wheatsheaf	Conventional	*Lolium perenne* *Taraxacum officinale* *Rumex obtusifolius* *Trifolium pratense* *Leontodon hispidus*
Wheatsheaf	Wildlflower	*Lolium perenne* *Dactylis glomerata* *Taraxacum officinale*

		Achillea millefolium *Rumex obtusifolius* *Trifolium pratense* *Silene dioica* *Leontodon hispidus*
North Court	Conventional	*Lolium perenne* *Taraxacum officinale* *Galium sp* *Veronica sp* *Cerastium fontanum* *Leontodon hispidus*
North Court	Wildlflower	*Lolium perenne* *Dactylis glomerata* *Taraxacum officinale* *Achillea millefolium* *Galium sp* *Rumex obtusifolius* *Trifolium pratense* *Veronica sp* *Silene dioica* *Lotus corniculatus* *Centaurea nigra* *Cerastium fontanum* *Leontodon hispidus*

Table 8.2. Species list from plots sampled in Chapter 4.

Site	Plot	Species
Marshgate	Conventional	*Lolium perenne* *Dactylis glomerata* *Holcus Lanatus* *Plantago lanceolata*
Marshgate	Wildflower	*Lolium perenne* *Dactylis glomerata* *Leucanthemum vulgare* *Achillea millefolium* *Trifolium pratense* *Veronica sp* *Taraxacum officinale* *Galium sp* *Silene dioica*
Newington	Conventional	*Lolium perenne* *Taraxacum officinale* *Rumex obtusifolius* *Galium sp* *Trifolium pratense* *Veronica sp* *Cerastium fontanum* *Leontodon hispidus*
Newington	Wildflower	*Lolium perenne* *Dactylis glomerata* *Taraxacum officinale* *Achillea millefolium* *Rumex obtusifolius* *Galium sp* *Trifolium pratense* *Veronica sp* *Lotus corniculatus* *Cerastium fontanum* *Silene dioica* *Leontodon hispidus*

Figure 8.1: Example Bare only treatment from chapter 5

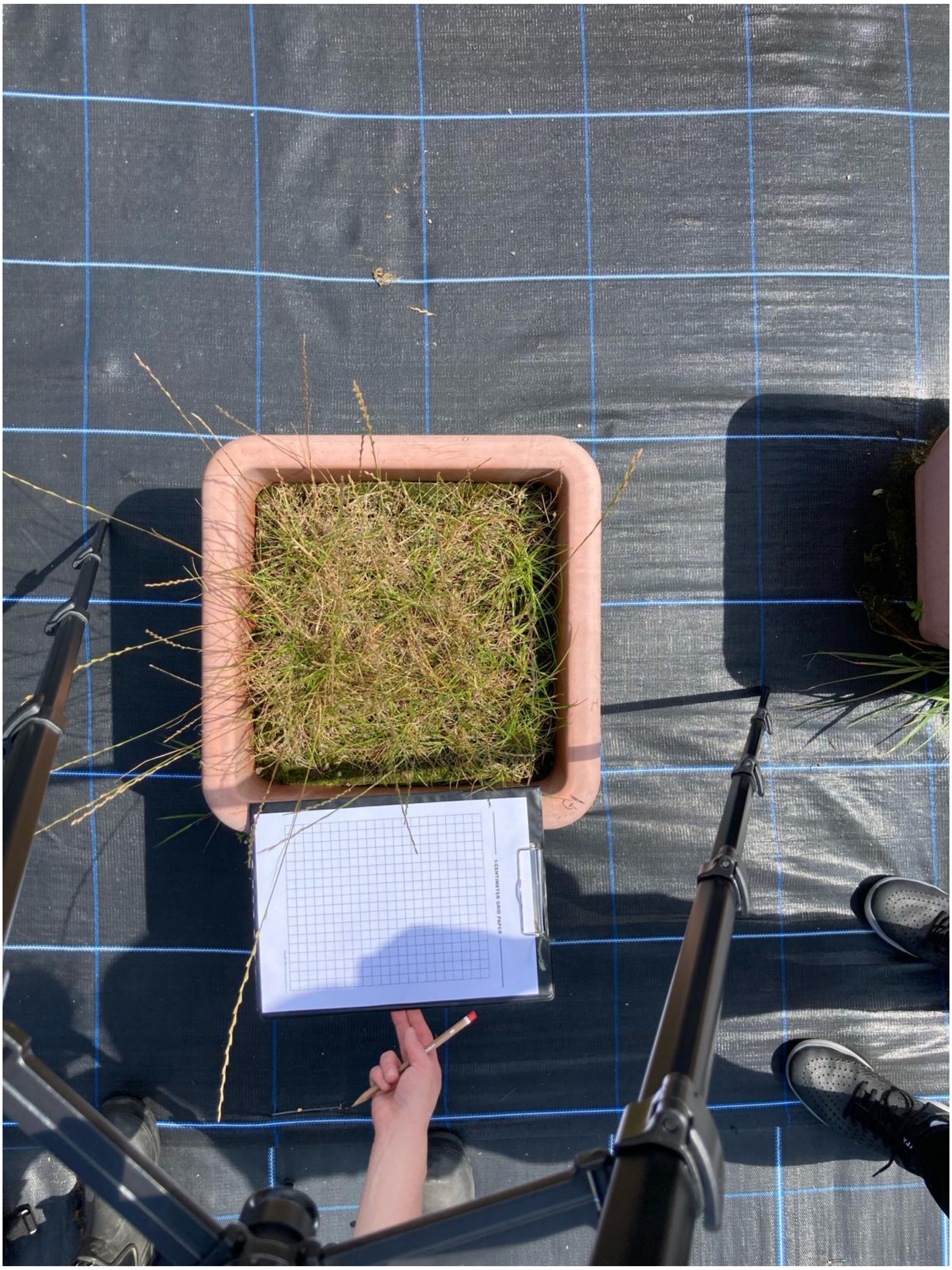

Figure 8.2: Example Grass only treatment from chapter 5

Figure 8.3: Example FL treatment from chapter 5

Figure 8.4: Example GL treatment from chapter 5

Figure 8.5: Example FLG treatment from chapter 5