Spatial benefits from temperate agroforestry

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Declaration and author contributions

I confirm that the work submitted is my own, except where work which has formed part of jointly authored publications has been included. My contribution and contributions of other authors to this work has been explicitly indicated below. I confirm that appropriate credit has been given within the thesis where reference has been made to the work of others.

Chapter 1 includes information that has appeared in publication in *Encyclopaedia of Soils in the Environment (Second Edition)* as:

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Thesis by alternative format rationale

This thesis follows the University of Leeds Faculty of Environment protocol for the format and presentation of an alternative style of doctoral thesis including published material. The research questions of the project were investigated using a range of approaches, which made the presentation of the data chapters as three individual manuscripts appropriate.

One of the manuscripts has been published, one has been resubmitted for publication following favourable review comments, and the final manuscript has recently been submitted for publication. The main body of the thesis therefore consists of the published and prepared manuscripts.

These are preceded by an introduction, which provides background to the thesis, reviews relevant literature, and outlines the aims and objectives of the work. A synthesis chapter brings together findings from the three chapters/manuscripts and discusses them in the context of the research questions, concluding the thesis.

Copyright declaration

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You will go out in joy and be led forth in peace; the mountains and hills will burst into song before you, and all the trees of the field will clap their hands.

– Isaiah 55:12 (NIV)

Abstract

Agroforestry has grown in popularity as a land-sharing strategy capable of sustaining food production while supporting other terrestrial ecosystem benefits. However, in temperate zones, widespread adoption of agroforestry is hindered by limited understanding of its relevance and benefits across contexts. Using field sampling and hydrological modelling approaches at agroforestry sites in the UK, this thesis addresses context-specific knowledge gaps in support of three beneficial outcomes: soil carbon storage, soil nutrient cycling and soil hydrological functioning. Four practitioner design and context factors are assessed for their influence over these outcomes: planting trees (i.e. an indiscriminate tree area focus), tree species selection, tree placement and landscape influence.

Minimal differences in soil organic carbon (SOC) stock to 50 cm depth were found overall for areas planted with trees compared to control sites. In 12 m-spaced agroforestry alleys a small difference of 700 kg ha year^{-1} in SOC was observed compared with a treeless control. Tree species was shown to influence SOC storage in forest floor material, and hillslope factors mediated the potential for agroforestry to mitigate soil and SOC loss to erosion. Understanding soil nutrient dynamics required a spatial focus, with narrower 12 m tree alleys having higher nutrient stocks than 24 m alleys due to greater resource circularity, and competition for nutrients (and water) at the tree/alley boundary. Natural flood management potential was strongly determined by tree placement at lower storm intensities, with 40 ha indiscriminate planting delivering only 1.6 times the flood peak reduction of just 1 ha of riparian tree planting. A spatial focus was therefore essential for understanding benefits and trade-offs across soil carbon, soil nutrient and soil water domains, moderated by landscape and tree-species factors. Support which recognises this heterogeneity is critical for agroforestry to become an attractive option in adapting agroecosystems to a changing climate.

Table of Contents

List of Figures

List of Tables

Chapter 1. Introduction

This thesis investigates how key soil and ecosystem benefits from agroforestry are distributed in the landscape in the vicinity of trees, with a focus on contextual controls on these outcomes, and how benefits accrue at different scales of observation both laterally within the landscape and with soil depth. Considerable research, particularly focussed on the tropics, has uncovered mechanistic linkages between contextual factors such as climate, tree or soil management and tree-crop associations; and common outcomes such as soil organic carbon (SOC) storage, soil nutrient and soil water cycling in agroforestry systems. Yet despite this research, many practitioner-relevant questions concerning agroforestry outcomes remain unaddressed, such as how to choose alley width, which tree species to plant and how to situate systems within landscape context. Moreover, in contrast with tropical agroforestry which is much better-studied, evidence for agroforestry outcomes in temperate zones remains much more limited, and the importance of observation scale in assessing stakeholder value of agroforestry is only rarely considered. This thesis therefore targets these practitioner questions, asking how controls such as tree species, hillslope position, and tree placement influence the capacity for agroforestry systems to support ecosystem service provision in temperate regions, and on which spatial and soil depth scales outcomes are observed.

This introductory chapter sets the scene for three research chapters which follow, before the thesis concludes with a synthesis in Chapter 5. The next section of the introduction (1.1) provides a detailed background to the study, considering wider challenges for land use in the context of resource constraints and changing climate, and the potential role of agroforestry. Section 1.2 provides a background to agroforestry itself and defines some system types, before section 1.3 outlines opportunities and challenges for scaling agroforestry using the United Kingdom as an example. Section 1.4 reviews current literature and identifies knowledge gaps related to factors controlling the outcomes of temperate agroforestry, before sections 1.5 – 1.8 outline research questions and methodology for the remainder of the thesis considering evidence needs.

1.1. Background

Terrestrial ecosystems regulate the natural environment and facilitate provisioning of food and essential materials for life (Millennium Ecosystem Assessment, 2005; Díaz et al., 2018; Smith et al., 2021). These ‘ecosystem services’ have been divided into four categories: *supporting* services such as nutrient cycling and primary production, *provisioning* services including food and fuel production, *regulating* services including flood risk and climate change mitigation, and *cultural* services such as education and recreation (Millennium Ecosystem Assessment, 2005); all of which are essential for human health and societal wellbeing. However, land management decisions focussed on food production, technological expansion and urban development have prioritised provisioning services such that much food production now occurs in competition with other terrestrial services (Daily, 1997; Müller et al., 2016). In the United Kingdom, a fourfold increase in yields in the post-war period driven by government subsidy, chemical inputs, mechanisation and efficiency savings has increased food production, albeit at significant environmental cost (Robinson and Sutherland, 2002; Eichhorn et al., 2006; Hodge, 2016) prompting questions about tipping points in ecosystem service delivery from agricultural landscapes (Watson et al., 2021).

Disregard for complexity is often a root cause of environmental degradation. In an agricultural context, intensification of production of cereals and other annual crops uses soil resources and reduces the input of carbon (C) and nutrients from plant litter to soil due to crop harvesting. The result is a well-documented decline in SOC (Lal, 2006; Sanderman et al., 2017) and nutrients (Lindstrom, 1986; Tan et al., 2005) over time, that impairs hydrological functions and degrades structure, and leads to a weakening of soil microbial communities and decline in soil functions (Trivedi et al., 2016; Gupta et al., 2022; Judson et al., 2023). Similarly, as demand for food and raw materials has intensified on land, economic pressures have either forced or incentivized farmers to bring new or more marginal land into production, a process which has necessitated removal of natural habitat including woodlands (Eichhorn et al., 2006), wetlands (Verhoeven and Setter, 2010) and more recently, hedges (Petit et al., 2003; Tresise et al., 2021), all of which support soil and wider biosphere functioning. Removing perennial vegetation from landscapes for intensive crop production interrupts feedbacks on which soil functioning relies, with consequences for terrestrial ecosystem service provision.

The significance of damage caused by removal of trees and hedges from the landscape is gaining prominence among both the public and policymakers. Additionally, the importance of soil systems for wholesale sustainability is increasingly recognised (European Commission, 2021; Smith et al., 2021), along with the unsustainability of many examples of land-use change that have occurred over the last two centuries (Robinson and Sutherland, 2002). The popular environmental movement which began in the 1960s brought the urgency of ecosystem service decline into modern public consciousness (e.g. Carson, 1965; Meadows et al., 1972), but it is only more recently that negative effects of land-use decisions have begun to force shifts in policy (Batáry et al., 2015; Defra, 2018b; Defra, 2018a; Defra, 2020). Of particular interest is the potential for the reintroduction of semi-natural habitats, such as trees and hedges, into the farmed landscape (Burgess and Graves, 2022; Biffi et al., 2023; Judson et al., 2023). The benefit of trees and hedges in agriculturally intensive environments is an increase in multifunctionality – for example, hedges are not only a stock-proof barrier, but also promote better water infiltration in soils, sequester C and provide both above and below-ground habitats for beneficial organisms (e.g. Holden et al., 2019; Biffi et al., 2022). Similarly, trees offer shelter for both livestock and crop systems, the potential for diversification of income and restoration of mutually beneficial below-ground symbioses with soil microorganisms.

There has been growing interest in the potential for agroforestry to contribute to climate change mitigation and national carbon inventories in the lead up to net zero (IPCC, 2014; CCC, 2020; Woodland Trust, 2022). Agroforestry is frequently included as a Nationally Determined Contribution in support of the Paris Agreement (Seddon et al., 2019) and several European nations have set targets for agroforestry and hedgerow planting (Biffi et al., 2023). It is thus timely to consider the extent to which strategies such as agroforestry can contribute to balancing provisioning of food, fuel and other goods with regulating ecosystem services such as flood risk and climate change mitigation which support life. Ecosystem impacts cannot continue to be externalised by policymakers, agri-businesses, and land managers, not least because they threaten the sustainability of food production itself. The challenge, therefore, is to consider how agricultural landscapes might be bolstered in their capacity to support natural systems whilst simultaneously supplying the needs of a growing and changing population. Recent discussion at policy level has sought to address this, focussing on how goals of agricultural environmental schemes (AES) in the UK might pivot away from subsidies based on farm area towards payments for the provision of public goods (Defra, 2018b). Agroforestry is one such strategy, reintroducing trees to agroecosystems to boost their complexity and strengthen their contribution to terrestrial ecosystem service provision (Jose, 2009). Policy in the UK and European Union has sought to support expansion of agroforestry with financial support for planting and management of trees (Mosquera-Losada et al., 2023; Defra, 2025), and a target of 10 % of land under agroforestry has been included in contributions to net-zero emissions in the UK by 2050 (CCC, 2020). However, adoption of agroforestry in temperate areas is hampered by misunderstanding of its benefits, cost of conversion, understanding of appropriate tree management and a lack of relevance to specific contextual constraints (Westaway and Smith, 2019; Sollen-Norrlin et al., 2020). This thesis will make the case for a spatially-explicit, multiple-benefit focus on agroforestry, such that it can become a stronger value proposition for practitioners across a diverse range of scales and temperate contexts.

1.2. Agroforestry

1.2.1. Origins

Agroforestry is the cultivation of two or more species of plants (or plants and animals) within the same space, at least one of which is a woody perennial (Nair, 2005). It thus offers many of the same benefits as hedges, such as shelter and protection of soil, while substantially increasing the density of space for nature in agricultural land. Consequently, agroforestry systems are more complex, both ecologically and economically, than conventional agricultural systems. Within agroforestry systems, trees may be planted both on the edges of fields and in amongst productive areas normally used solely for livestock or annual crops.

The first written use of the term *agroforestry* was in the late 1970s, defined as *'a sustainable land management system which increases the yield of the land, combines the production of crops (including tree crops) and forest plants and/or animals simultaneously or sequentially on the same unit of land'* (Bene et al., 1977). The focus was explicitly on provisioning benefits of incorporating trees into farming systems, only alluding to sustainability as a secondary benefit in service of provisioning. However, such a focus is not unusual; practices resembling agroforestry are ancient, and the productivity benefits of farming with trees have been known for millennia. Throughout history, crops and trees have been deliberately cultivated together, and like other regenerative or agroecological practices, agroforestry can be found in ancient indigenous practice (Clarke and Thaman, 1993; Nair, 1993). Practitioners of agroforestry often have no word for it in their language, with some referring to it simply as 'traditional practice' (Hoffner, 2019). The Yoruba of Nigeria cite soil fertility and avoidance of nutrient leaching and erosion as principal benefits of combining herbaceous, shrub and tree crops (Nair, 1993). Shade-grown coffee and cocoa comprise a considerable area of global agroforestry, and for many years communities have identified better food security and economic returns as reasons for combining the two crops (Hoffner, 2019). Some of the most successful examples of agroecological farming today revive ancient technologies, incorporating trees to promote productivity and simultaneously conserve resources, with potential to reclaim function within partially or wholly degraded soils otherwise unsuited to sustainable production (Reij et al., 1996; Nicholls and Altieri, 2018).

More recently, agroforestry has been defined simply as 'farming with trees' (Soil Association, 2019). Particularly in the industrialized world, the stated environmental benefits of agroforestry have grown in prominence as an alternative to conventional, intensive land management systems. Overreliance on agrochemicals such as fertilizers and pesticides and the systematic conversion of natural habitat to cultivation have detached agricultural practice from ecological cycles which sustain ecosystem service provision. Yield and efficiency gains through the 20th Century, although economically significant, are not sustainable so long as natural systems simultaneously sustained by land are neglected. In contrast, the presence (or reintroduction) of trees on farms offers potential to restore and take advantage of ecological complexity for the benefit of provisioning and ecosystem regulation. In this sense, agroforestry is typical of agroecological methods, applying ecological principles to the design of food systems (Gliessman, 2014) and utilizing circularity to generate multiple benefits and simultaneously address challenges to soil functioning, food security and environmental sustainability.

For most of history, food production has been the justification for incorporating trees on farms (King, 1987), with ecological gains serving that purpose. The potential for agroforestry to deliver global goods beyond provisioning, in particular carbon sequestration or resilience against climate change, is comparatively modern and part of the growing trend in its popularity and promotion. Of particular relevance is the extent to which agroforestry can deliver benefits in temperate regions to mirror its success elsewhere (e.g., tropics) for both people and environment. Historic agroforestry practices are found in temperate areas – the concept of 'dehesa' (Fig. 1.1), a Spanish system ('montado' in Portugal) in which dispersed pastureland is interspersed with holm and cork oaks, has origins in the early Middle Ages, however the practice is almost certainly much older than the word (Álvarez, 2016).

Figure 1.1 – Examples of agroforestry. Clockwise from top left: a) Cattle graze in a silvopastoral dehesa system; b) silvoarable alleys within poplar in Bedfordshire, UK; c) shelterbelt planted for protection of bulb crops, Isles of Scilly, UK. Panel a) Toros en la dehesa by Fernando Cuenca Romero is licensed under CC BY-NC-SA 2.0. b) Silvoarable agroforestry, Silsoe 2002 by Paul Burgess is licensed under CC BY-NC-SA 2.0. Panel c) Bulb fields and shelter belts from the air, St Mary's by Mary Gillham Archive Project is licensed under CC BY 2.0.

Similarly, trees planted within slash and burn systems in Germany date back to the Middle Ages (King, 1987). Lines of trees have often historically been introduced as shelterbelts, and some modern agricultural landscapes in Europe resemble agroforestry simply as a result of remnant trees surviving land-use change (den Herder et al., 2017). What distinguishes modern agroforestry from the simple presence of trees on farms is its intentionality and focus on people and management as integral components of the system (Nair, 2005).

1.2.2. Agroforestry system types

Agroforestry includes several subdivisions depending on system design (Lawson et al., 2016; Augere-Granier, 2020) (Fig. 1.1). *Silvopasture* describes integration of trees with livestock such as ruminants, pigs or poultry, an example being the *dehesa* or *montado* systems mentioned above. *Silvoarable* agroforestry is the integration of trees and crops in the same field. Within silvoarable systems, trees are normally planted in rows to allow access for machinery, with crops in the 'alleys' in-between; as such silvoarable agroforestry is often referred to as 'alley cropping' (Lawson et al., 2016; Soil Association, 2019). Additionally, all three elements (trees, crops, animals) may be combined into an *agro-silvopastoral* system. Additional systems that are commonly classed as agroforestry include: shelterbelts, windbreaks, hedgerows and riparian buffer strips, which are examples of trees on farms for the promotion of ecosystem and economic benefit, designed to protect livestock, crops or soil and water quality. Smaller-scale applications such as kitchen gardening make use of trees to contribute to vegetable production.

The reintroduction of trees to farmed landscapes in more recent times is thus with the explicit focus of strengthening soil function and wider ecosystem goods lost through land-use conversion to conventional agriculture. Trees provide ecological corridors within the farmed landscape, while offering many of the benefits for soil function that come with increased diversity of vegetation and the ability to host higher levels of above and below ground biodiversity.

1.3. Agroforestry in the UK: opportunities and challenges

Trees on farms are still a relatively common sight in temperate areas, and the United Kingdom is a good example. An inventory in 2000 suggested there were 122 million trees in the UK outside of woodlands, with particularly high density in England and Wales (6.8-7.4 trees ha^{-1}) compared with Scotland (2.5 trees ha^{-1}) (Burgess, 2011). More recently, 565,000 ha (4.3% of total land area) of trees outside National Forest Inventory woodland were mapped in England alone (Brewer et al., 2017). Yet despite the presence of trees on farms, agroforestry as a deliberate practice is rare in the UK, struggling to resonate with UK farmers and consumers (Burgess, 2017) and falling between regulatory cracks that limit its economic potential for land managers (Soil Association, 2018). This contrasts with the relative popularity of agroforestry elsewhere in Europe, particularly in Iberia (den Herder et al., 2017). Capital investment requirements and tenancy issues are significant barriers for UK practitioners, as are a limited understanding of what agroforestry itself involves, management complexity, potential benefits and lack of market incentives (Sollen-Norrlin et al., 2020; Tosh, 2021). Moreover, at typical planting densities (50-200 stems ha^{-1}) agroforestry failed to qualify for direct payments under the EU Common Agricultural Policy, which imposed a maximum planting density of 50 stems ha^{-1} until 2013, increasing to 100 stems ha^{-1} up to 2020 (Mosquera-Losada et al., 2023). This was a missed opportunity for environmental and economic reasons: the UK has among the highest percentage area of farmland in Europe (73 percent) with one of the lowest proportions of woodland (15 percent) (Burgess, 2017). The country is also significantly exposed to strong winds and adverse temperatures, affecting yields, soil and nutrient retention and livestock performance in the absence of shelterbelts, particularly in upland regions (Gregory, 1995). Moreover, droughts and flooding are predicted to intensify by the end of the century under climate change (IPCC, 2013). The absence of trees further weakens biosphere feedbacks responsible for sustaining regulating ecosystem benefits such as flood management and climate resilience.

Recent policy such as the Sustainable Farming Incentive (SFI), part of the Environmental Land Management scheme (ELMs) in England has sought to address this, with support for maintenance of in-field agroforestry (AGF1 and AGF2) and planting new trees (AF1 and AF2) (Defra, 2025). Meanwhile the EU has removed planting density limits for support in the CAP 2021-27 period (Mosquera-Losada et al., 2023). The Woodland Trust (2022) and Climate Change Committee (CCC, 2020) have recommended that 10% of UK arable land converted to silvoarable, and 30% pasture converted to silvopasture in order to contribute to C emissions reductions and net-zero by 2050. Yet for uptake at these levels to become a realistic prospect, practitioners need better understanding of how agroforestry in their own context will translate into diverse benefits beyond C storage alone, including economic benefit from increased soil fertility or reduction in soil loss to erosion. Current guidelines for agroforestry design focus not so much on predicting benefits of given systems and contexts but on adaptive management, in which designed-in capacity for *post hoc* adjustments is prioritised over predictability of spatial outcomes (e.g. Soil Association, 2019). As in any sustainability context, delivering benefits from agroforestry is a wicked problem, in which stakeholders have differing reference frames and starting goals against which outcomes are judged (Rittel and Webber, 1973; Bouma et al., 2011), making the idea of 'success' contested in terms of agroforestry design required to achieve it in the landscape. For example, arable farmers needing fertility for crop production (e.g. Donn et al., 2014) will prioritise local (1 - 10 m scale), topsoil-focussed soil benefits, whereas stakeholders interested in landscape C inventories or natural flood management, such as local authorities or government, will focus on landscape (1 - 1000 km) and whole-soil-column responses to tree planting (e.g. Rogger et al., 2017; De Stefano and Jacobson, 2018). Scale is, therefore, a critical concept in differentiating goals for planting agroforestry, as are the multiple benefits it has potential to provide.

Combining these ideas, delivering agroforestry at scale can be thought of as a system of interdependencies (Fig. 1.2, system variables italicised in following text). Soil and ecosystem benefits of agroforestry are here labelled *agroforestry outcomes*. This thesis focusses on three outcomes (i.e. ecosystem services): a. C storage and climate change mitigation, b. soil fertility and nutrient cycling, and c. water retention and natural flood management. These are assessed using soil organic carbon (SOC), macronutrient availability and soil moisture/infiltration indicators, respectively, as proxies. Impacting delivery of these outcomes is agroforestry *system design* – choices within practitioner control which affect how trees interact with the pedosphere, to deliver ecosystem benefit. This interaction is iterative, with *agroforestry outcomes* in-turn influencing subsequent *system design* alterations in a process of adaptive management. Practitioner *system design* choices are influenced by *system goals*, with stakeholders assigning variable weighting to different outcomes depending on need, and designing agroforestry systems accordingly. This process will also be iterative, in that new goals can emerge on shorter timescales during design, and on longer timescales following analysis of outcomes. On a longer timescale still, driving the development of system goals and system design is *policy*, in the form of financial, legal or knowledge-exchange incentives which can influence both *system goals* and *system design*, which in-turn should be designed in response to outcomes. Two important external controls play a role: *local context* and *scale*, with the former including geographic surrounding and environmental factors (Jose, 2009), and the latter including lateral (e.g. tree, farm, landscape) and soil depth dimensions. Finally, a degree of unknown *random variation* will influence *agroforestry outcomes* independently of *system design*.

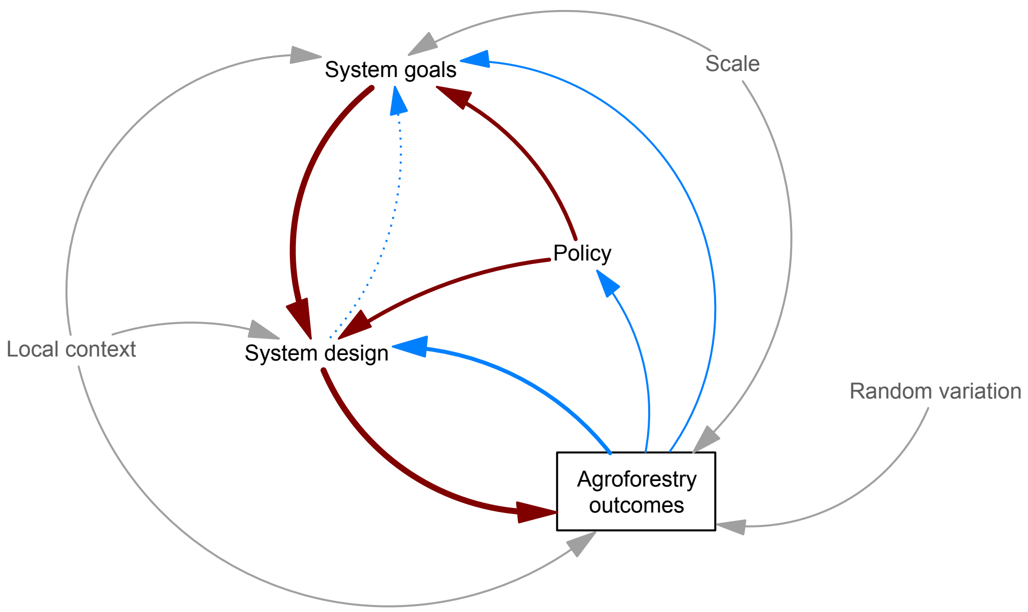

Figure 1.2 – Interdependencies associated with scaling agroforestry for multiple benefits. Driving effects on agroforestry outcomes (direct and indirect) are shown in red, and feedback mechanisms are shown in blue. Grey arrows represent external drivers. Arrow weight corresponds to strength of forcing. A temporal dimension is not indicated, although interdependencies will operate on quite different timescales.

Scaling agroforestry depends on well-researched linkages between each of these elements. To name a few examples: the effect of specific design choices on agroforestry outcomes must be well-understood, as should the influence of contextual factors outside practitioner control. Appreciating scale is critical in considering multiple benefits and differences between stakeholder objectives (Reith et al., 2022). Goals and systems need to be supported by policy informed by evidence. Given the high capital cost to practitioners of redesigning agricultural systems around a new management model, robust linkages between input and outcomes allow agroforestry to become a better value proposition. The rest of this introductory chapter examines the strength of these linkages in terms of current literature evidence for each of the three outcomes: C storage, soil fertility and soil hydrology. Four *design* and *context* factors are considered – tree-planting on arable soils (hereafter simply 'trees'), tree species choice, tree layout/placement and slope/topography.

1.4. Literature evidence for soil benefits

1.4.1. SOC and C storage

At planting densities typical of agroforestry systems (50-200 stems ha^{1}) an increase in SOC stocks following planting is generally observed (Upson and Burgess, 2013; Cardinael et al., 2018; De Stefano and Jacobson, 2018; Shi et al., 2018; Mayer et al., 2022). In their meta-analysis of 53 published studies De Stefano and Jacobson (2018) demonstrate how measurement depth and prior land use control the magnitude of SOC changes. At 0-30 cm depth arable to silvoarable agroforestry produced the largest increase in SOC stocks (+40%), with a smaller increase for pasture/grassland to silvopasture (+9%). At 0-100 cm depth, although an increase in SOC stocks of +34% was observed for arable to silvoarable conversion, there was no significant difference in SOC stock for pasture/grassland to silvopasture, demonstrating the importance of sample depth and prior land use in determining additional C storage potential. SOC content typically decreases exponentially with depth (Rolo et al., 2023) yet as shown in Fig. 1.3, increases in SOC in agroforestry systems compared to a control site are possible in deeper horizons, as well as the commonly reported topsoil (Cardinael et al., 2018). Fine root systems under agroforestry are typically highly modified compared with continuous woodland, often diverting downwards to access areas beneath the plough layer and shallower-rooting adjacent grassland or cropland such that C-dense exudates can be delivered to deeper in the soil profile (Cardinael et al., 2015b). Nonetheless differences in SOC stocks in deeper horizons between agroforestry and pasture systems are often harder to resolve (Upson et al., 2016), and sampling below the topsoil is urgently needed, particularly in temperate systems, to quantify the C sink potential of agroforestry more effectively (Upson and Burgess, 2013).

There are several other factors that influence SOC stocks under agroforestry, adding nuance to more general meta-analysis findings such as that by De Stefano and Jacobson (2018) summarised above. One of these is climate, to which SOC stock changes are known to be sensitive (Ma et al., 2020). Faster SOC accumulation is generally observed in tropical areas, albeit with higher eventual C stocks in temperate areas despite slower accumulation. Mayer et al. (2022) conducted a meta-analysis focussed on C storage rates in the temperate area, compiling studies comparing agroforestry treatments with treeless controls (both diachronic, in which control is baseline-sampled on the same site before agroforestry establishment; and synchronic, in which control is a treeless neighbouring plot with similar soil characteristics). Their results confirm that temperate agroforestry systems normally store more C than control areas; that accumulation rates are faster in the topsoil (0.21 ± 0.79 t C ha^{-1} $year^{-1}$, 0-20 cm) than subsoil (0.15 ± 0.26 t C ha^{-1} $year^{-1}$, 20-40 cm); that hedgerows accumulate SOC most rapidly, followed by silvoarable systems, with changes in SOC in silvopasture being slightly negative; and that broadleaf systems accumulate C more rapidly than conifer systems. However, as their accumulation rates by depth show, uncertainty values can exceed the magnitude of storage rates themselves, complicating determination of the overall effect of agroforestry on SOC storage.

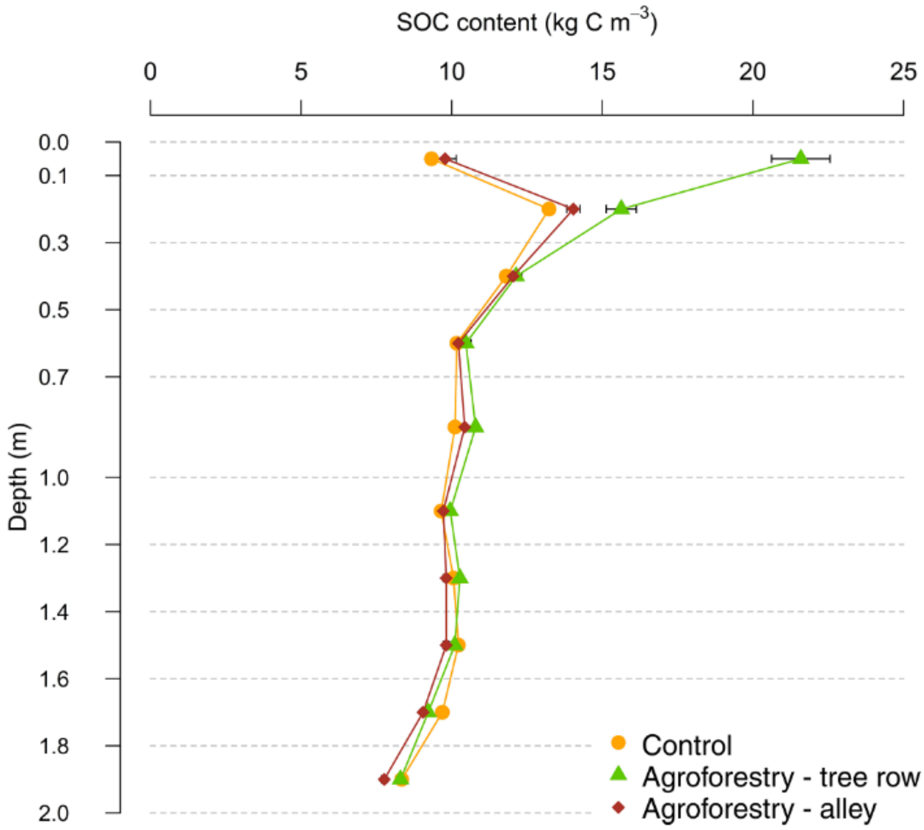

Figure 1.3 –SOC within a silvoarable treatment, showing content variation with depth within tree row (green), arable alley (red) and control plot. The majority of SOC additions are in the top 30 cm, however significant (small) differences exist down to 2 m. N=40 in tree row, N=60 in alley and N=93 in control. From Cardinael et al. (2018).

These uncertainties demonstrate the high degree of variability in outcomes among temperate agroforestry studies due to context, and the need for investigation focussed on controls over which farmers and other land managers have influence, such as tree species choice, planting geometries and siting within the landscape. Targeting factors is constrained by the availability of mature temperate agroforestry study sites (Wotherspoon et al., 2014), particularly given that a decade or more is often needed to see changes in outcomes such as SOC stock (Smith, 2004). The precise mechanisms by which agroforestry produces increases in SOC are also not well known (Cardinael et al., 2018), and little is known of how tree-derived C additions are transferred laterally from tree rows into intercrop alley or pasture areas, and how SOC stocks vary within agriculturally used components of agroforestry systems (Mayer et al., 2022). Nonetheless, inputs of C to the soil can be characterized into three principal pathways: addition of leaf litter (or pruning residue) to the soil surface, addition of fine root litter turnover in the soil and the production of organic carbon exudates within the rhizosphere (Jobbágy et al., 2001; Haichar et al., 2014). SOC stocks are in a dynamic equilibrium between gains from organic material incorporation and exudates, and losses to a) the atmosphere as CO_2 as a result of organic matter decomposition by microorganisms, and b) the aquatic ecosystem via leaching of dissolved organic carbon (DOC) or erosion of particulate organic carbon (POC). The presence of root matter in soil helps to stabilize soil aggregates, which in turn protects organic molecules from exposure to microbial respiration (Dijkstra et al., 2021), and at larger scale, trees interact with landscape to prevent erosive soil loss of POC, in addition to soil nutrients such as mineral nitrogen (N) and phosphorous (P).

Given these known interactions between physical, chemical and biological components of agroforestry systems, it follows that an increase in ecological complexity is generally correlated with greater SOC stocks (De Stefano and Jacobson, 2018). This has important implications for practitioner design of systems. A number of studies confirm the importance of understorey vegetation for long-term SOC storage. Upson et al. (2016) examined a silvopastoral system in the UK fourteen years after planting and found higher concentrations of SOC in pasture soils (6.0 %) than beneath agroforestry (5.3 %) or woodland (4.6 %) at a depth of 0-10 cm. They speculate this difference to be the result of reduced understory beneath woodland and the silvoarable plot, leading to reduced fine root density and thus lower rates of rhizodeposition and fine root matter incorporation into the soil. Cardinael et al. (2018) found that the largest above-ground OC inputs in a silvoarable system came from herbaceous vegetation, and not from trees. Shan et al. (2001) assessed the effect of understory-elimination in pine plantations in the south-eastern US, finding that SOC storage was lower when the understory was removed.

The importance of complexity for SOC stocks implies that plot-scale factors such as tree spacing and silvoarable alley width are likely to control overall C dynamics, given the ecological connectivity that closer-spaced trees and understorey vegetation generate. This, along with the influence of soil depth on C accumulation mentioned above, highlights how scale issues are critical in determining SOC differences, both in terms of depth (Fig. 1.3) (Upson et al., 2016; Cardinael et al., 2018) and lateral distance from trees (Oelbermann et al., 2004; Cardinael et al., 2019). More rigour is needed in measuring spatial differences in outcomes at the plot scale (Minarsch et al., 2024), and alley width itself has not been examined as a control on C storage due to the difficulty of finding appropriate temperate sites at which it can be isolated as a controlling factor. At smaller, single-tree scale, C additions to soil under agroforestry are likely to be species-dependent, as trees produce litter of very different chemical and physical properties (Hoosbeek et al., 2018; Partey et al., 2018). The rate at which litterfall C is incorporated into soil also depends on soil and litter stoichiometry (e.g. C:N) controlling rates of microbial decomposition (Liang et al., 2017; Zhou et al., 2019), demonstrating how soil outcomes themselves are interlinked. Although better addressed in tropical zones, controls such as these have received minimal attention in temperate areas for their effects on SOC and long-term C storage in soil.

1.4.2. Macronutrients and soil fertility

Describing an overall effect of agroforestry on soil fertility is counterintuitive, as combining trees with crops or grassland gives rise to both competition and facilitation with respect to macronutrient concentrations (Dollinger and Jose, 2018). However, specific controls on how nutrients are cycled within agroforestry systems have received considerable attention, particularly in tropical areas (Cardinael et al., 2020). The effect of nitrogen-fixing (leguminous) trees within silvoarable systems is well known, including their influence over yields and biological nitrogen fixation (e.g. Querné et al., 2017). Using stable ^{15}N isotope tracers within KNO_3 and $(NH_4)_2SO_4$ fertiliser, trees (Hog plum – *Choerospondias axillaris*) and crops (peanut – *Archis hypogaea*) were shown to compete for N in the surface soil within a subtropical alley cropping system in China (Zhang et al., 2008). However, tree roots accessed leached N from deeper soil horizons otherwise inaccessible to crops and their more superficial rooting systems in a process known as nutrient ‘pumping’. The result was better N use efficiency overall when compared with the peanut monocrop system, demonstrating potential for trees to limit N loss through leaching and illustrating the presence of both synergies and trade-offs within the same system. Querné et al. (2017) looked to investigate how light and water effects might influence levels of biological N-fixation for alfafa (*Medicago sativa L.*), a N-fixing intercrop, grown in alleys of 17-year-old walnut trees (*Juglans nigra x Juglans regia L.*) in a Mediterranean site. Transect (N-S) measurements of $\delta^{15}N$, a proxy for the fraction fixed from the air, were taken from alfafa shoots at varying distances from the tree row. Alfafa yield losses in shaded areas close to tree rows were buffered by facilitative mechanisms between plants and trees which increased light use efficiency (LUE) and $\delta^{15}N$ in the intercrop, countering the notion that shade competition reduces N-fixation. In each of these examples nutrient dynamics in the alley vary with distance from the tree row, illustrating the importance of tree row spacing for nutrient use efficiency particularly in silvoarable systems.

As in the case of SOC, agroforestry systems add nutrients to surface soil directly through litterfall, and fertility differences are likely to be species-controlled, in addition to influence from other factors such as management and climate. This is evidenced in several tropical studies. Isolated trees in a Nicaraguan silvopastoral system increased C, N and P concentrations in the topsoil, with *Crescentia alata* Kunth improving N stocks and *Guazuma ulmifolia* Lam. favouring P stocks (Hoosbeek et al., 2018). Nitrogen mineralization patterns varied among ten plant species considered by Partey et al. (2018), with greatest N-supply (up to 93 mg N kg^{-1}) coming from two of them – *Tithonia diversifolia* (Hemsl.) A.Gray and *Gliricidia sepium* (Jacq.) Steud. In Zambia, Yengwe et al. (2018) demonstrate potential for *Faidherbia albida* A.Chev., an N-fixing tree, to improve soil fertility in the short term. Soils amended with *F. albida* leaf litter demonstrated higher rates of N mineralization than control soil samples, supplying an extra 18 kg N ha^{-1} $year^{-1}$. The extent of the litterfall effect may be moderated by tree age and, in turn, litter quantity – a study in Mexico revealed greater supply of P, Ca, Mg, Fe and Cu from younger cacao trees in an agroforestry system as they were supplying more litter (Pérez-Flores et al., 2018). Irrespective of age, litterfall supply of each of N, P, K, Ca and Mg was sufficient to recover nutrient concentration lost to the main cacao crop, again demonstrating the increase in nutrient use efficiency derived from a more complex agroecosystem such as agroforestry.

However, as in the case of SOC storage, contextual controls are much better evidenced in tropical zones, and knowledge of nutrient dynamics in temperate areas is much more limited (Cardinael et al., 2020). Some temperate controls are known. At tree scale, roots with larger N and P concentrations, such as those from legume crops, promote a smaller soil C:N ratio which facilitates faster decomposition of organic matter; furthermore, additive effects between litter and crop residues enhance efficiency of uptake and grow long term fertility (Zeng *et al.*, 2010). At plot scale, trees are known to compete with alley crops for nutrients, and growth phase effects between each are critical for overall efficiency (Van Noordwijk and Purnomosidhi, 1995). As if to demonstrate this, yields and N use efficiency for agroforestry systems may be influenced by the presence or absence of artificial barriers between tree and crop. In a US Midwestern maize system combined with black walnut trees (*Juglans nigra L.*), tree roots took up soil water and N before the maize was planted, reducing supply for the crop. With a 1.2 m soil barrier in place the competition effect was minimized (Jose *et al.*, 2000). These alley edge competition effects are likely to be relevant in a wide range of simultaneous agroforestry systems, in contrast with sequential systems in which tree and crop annual growth pattens are offset from one another and root capture of nutrients and water is more facilitative between species (Van Noordwijk and Purnomosidhi, 1995). Other competition mitigation effects have been tested, such as root pruning, which was found by Inurreta-Aguirre et al. (2022) to have no appreciable effect on yield, despite increasing water availability to crops. What is not known, however, is how competition effects relate to specific alley width choices, and the extent to which space available for agriculture in the inter-row trades off against enhanced nutrient cycling with greater tree density. More detail is needed on how contextual factors such as these control nutrient cycling in temperate agroforestry systems.

1.4.3. Soil hydrology and natural flood management

A number of plot-scale benefits of agroforestry for water cycling are well known. Benefits such as infiltration, *in situ* storage and reduced runoff are normally enhanced in agroforestry compared with conventional agriculture due to higher spatial and ecological complexity (Rowe et al., 1999; Cardinael et al., 2020). Complementary root distributions between trees and crops improve water use both by enhancing water distribution across shallow and deep soil horizons and through shared mycorrhizal networks (Cardinael et al., 2015b; Bayala and Prieto, 2019; Cardinael et al., 2020). Although significant competition effects between trees and crops are observed for water (Jose et al., 2000a), field studies have shown that these interactions can be both adaptive and facilitative. For example, jujube (*Ziziphus jujuba* Mill.) trees in semiarid China selectively sourced water from either shallow or deep soil horizons depending on levels of demand from adjacent crops of canola (*Brassica napus* L.) and daylily (*Hemerocallis fulva* L.) (Huo et al., 2020). Tree roots and litterfall increase surface roughness, promote aggregate stability and increase soil porosity (Liu et al., 2016), promoting infiltration, slowing surface flow and reducing soil erosion and sediment transport during storm events (Battany and Grismer, 2000). Management of the understorey beneath trees may further assist in increasing roughness and promoting infiltration (Dabney, Delgado & Reeves, 2001). However, as is the case for SOC and soil nutrient benefits above, the generality of these effects and how they are controlled by contextual factors such as tree spacing and layout are not well known (Cardinael et al., 2020).

Tree planting is known to influence hydrological functioning in upland areas, both from empirical (Dadson et al., 2017; Monger et al., 2022) and modelling studies (Gao et al., 2015; Bond et al., 2022; Kingsbury-Smith et al., 2023), albeit at larger scale and higher planting densities than agroforestry. Natural flood management (NFM) can be defined as interventions distributed throughout a landscape which restore or enhance catchment processes to reduce flood risk, while fostering environmental benefits such as biodiversity, soil and water improvement, carbon sequestration, better agricultural productivity and greater public enjoyment of the countryside (Dadson et al., 2017). Four physical processes aid NFM (Cumbria Strategic Flood Partnership, 2018): slowing water, storing water, increasing soil infiltration and intercepting rainfall. Catchment afforestation methods, such as agroforestry act on each of these components both directly and indirectly. Trees intercept rainfall directly in the canopy, preventing as much as 20-30 % from reaching the ground in the case of broadleaf woodland (Herbst et al., 2008). On the ground, trees or hedges increase surface roughness, which in turn leads to reduced runoff and better infiltration (Environment Agency, 2017; Battany & Grismer, 2000). In upland areas, modelling results suggest that increasing roughness in the riparian zone or areas of shallower gradient carries the largest NFM benefit for an equivalent area of land-use change (Gao et al., 2016).

Agroforestry is likely to impact NFM through indirect effects on soil. Agricultural activities frequently lead to surface and sub-surface soil compaction, the result of which is poor infiltration and increased surface runoff of water (Dadson *et al.*, 2017). Practices such as agroforestry which are capable of promoting lower soil bulk densities and higher infiltration may offer NFM as a benefit, provided soil permeability does not increase to such an extent that water reaches the stream very quickly by subsurface processes (Whipkey, 1965). Infiltration rates beneath UK hedgerows can be more than an order of magnitude greater (100 mm hr^{-1}) compared with arable (3 mm hr^{-1}) soils (Holden et al., 2019). Nonetheless, assessment of the potential for lowland agroforestry to provide hydrological benefits, including optimal placement of trees, has not been studied. Isolated studies have considered the role of woodlands for NFM in temperate farmed landscapes (e.g. Marshall et al., 2009), yet evidence for effects at the catchment scale is minimal, in part due to the logistical challenge of undertaking large, long-term studies and also transferability of results between catchments with varying geographies. Anecdotal evidence exists for positive catchment-scale effects of NFM interventions (Nisbet, 2017), however, little is known about underlying mechanisms or complex interactions of summing local effects over whole landscapes. As Rogger *et al.* (2017) stated in a review on catchment effects of land-use change, spatial connectivity of flow processes is crucial in determining effects of interventions on flood severity. Upscaling the effects of soil compaction to catchment scale depends on an understanding of how flowpaths and flow timings are impacted by linear features such as agroforestry alleys, hedges and ditches and their respective spatiotemporal interactions.

1.4.4. Summary and research approach

In light of the preceding literature review, Table 1.1 illustrates evidence in terms of linkages between design/context and outcomes (Fig. 1.2). The same four factors introduced in Section 1.3 are presented (trees, tree species choice, tree placement and landscape) along with the same three outcome categories (soil carbon, soil nutrients and soil water).

Table 1.1 – Interactions between context and design factors, and agroforestry outcomes

Context (Ct) or design (Dn) factor	SOC and C storage	Macronutrients and soil fertility	Hydrology and natural flood management (NFM)
Trees (Dn)	• Agroforestry systems generally produce higher long-term SOC stocks in soils to 30 cm depth (De Stefano and Jacobson, 2018) • C additions decrease exponentially with depth (Rolo et al., 2023) • Hedgerows store C fastest, followed by silvoarable and silvopasture, resp.	• Adding trees on farmland produces both competition and facilitation with respect to macronutrient concentrations (Dollinger and Jose, 2018) • Difficult therefore to generalise woodland effect on soil fertility.	• Soils under hedges and agroforestry demonstrate higher levels of infiltration, allowing water to be stored in the landscape during storm events, mitigating downstream flood peaks at local scales. • Tree and hedge canopies intercept water, creating above ground storage reducing flood peaks
Tree species (Dn)	• Broadleaf systems produce higher C inputs than conifer (Mayer et al., 2022) • Varying litter quality and quantity likely to produce varying C additions to topsoil	• Leguminous trees and shrubs are capable of fixing N from the atmosphere in soils. • Known differences between other species in terms of nutrient delivery through litterfall (Hoosbeek et al., 2018; Partey et al., 2018). Weaker temperate evidence	• Conifers are known to intercept higher levels of rainfall than broadleaf woodland (Calder et al., 2003) • Minimal knowledge of tree-species differences for NFM in lowlands
Tree placement (Dn)	• OM loss to erosion can be mitigated by planting. • Generally cross-slope woodland is best for achieving this (Narain et al., 1997). Weak temperate evidence. • Lower density planting typically produces higher C storage per unit area of trees in pasture due to competition and understorey effects (Upson et al., 2016)	• Litterfall additions of nutrients to soil, as well as tree root and mycorrhizal fungal extent diminish with increasing distance from tree, • Better nutrient circularity when trees are more closely spaced. • Weak evidence in temperate zones for nutrient effects, although some competition observed (Jose et al., 2000b)	• Positioning of woodland and other roughness-generating vegetation e.g. *Sphagnum* variably affects flood peaks in upland areas (Maske and Jain, 2013; Gao et al., 2016; Bond et al., 2022) • However, little is known about tree placement effect on NFM potential in lowland areas
Landscape (Ct)	• Surface soil organic matter can be lost to erosion on steeper slopes, with evidence that trees can mitigate loss (Narain et al., 1997).	• N and P in soil susceptible to loss through leaching and erosion, with higher risk on steeper slopes	• Upland evidence for shallower slope and riparian areas being most significant for NFM (Gao et al., 2016) • Minimal lowland evidence

Several specific knowledge gaps emerge. Firstly, the need for better evaluation of the impact of different agroforestry systems on soil health indicators and respective soil functions over deeper soil depths and lateral scale dimensions (Cardinael et al., 2020). The spatial extent to which any demonstrated benefit of agroforestry is realised is critical for determining its relevance to different stakeholders and their goals for planting agroforestry. Secondly, linkages between agroforestry benefits and factors over which planters of agroforestry have control, such as tree species or tree spacing choice, are not well characterised in temperate regions, and a more integrated approach is needed between outcomes such as water, nutrients and SOC (Cardinael et al., 2020). Linkages between some factors and outcomes is temperate regions is particularly weak, such as spatial evaluation of competition for nutrients between the cultivated and non-cultivated components of agroforestry systems, and between lowland agroforestry placement and potential for catchment-scale NFM. Finally, better understanding is needed of trade-offs between multiple outcomes related to both context and design choices for agroforestry. A number of important studies consider multiple outcomes of agroforestry at landscape scale (e.g. Fagerholm et al., 2016; Torralba et al., 2016), yet with only a few exceptions (e.g. Cardinael et al., 2015a; Holden et al., 2019) limited research has its focus on smaller scale spatial interactions between outcomes. Plot-scale factors such as tree spacing and placement are likely to influence all the outcomes introduced so far, and as alluded to by Beillouin et al. (2022), studies at small scale are critical as they enable practitioners to maximise ecosystem benefit from farmland over which they have an influence.

The aim of this thesis is to determine spatial effects of agroforestry design and context factors on soil outcomes in temperate settings. Through this I will demonstrate distribution and superposition of benefits and trade-offs in the landscape at scales ranging from plot to catchment. The aim will be achieved through the research questions below (RQ 1-3), moving from effects at small, 'single tree' scale, through plot-scale, before examining effects at catchment scale. This means that linkages can be identified with relevance for different stakeholders: tree- and plot-scale factors with greater relevance to landowners and farmers managing individual plots, and catchment-scale factors for policymakers and planners responsible for ecosystem delivery from whole landscapes. Different outcomes lend themselves more easily to different scales of observation: for example, C storage and fertility effects are more pertinent local to trees and within-field, whereas effects such as hydrology and NFM occur across much larger scales, from farm to catchment to landscape.

1.5. Research questions

RQ 1. What is the relative influence of trees, tree species choice and random pre-planting soil variability on common indicators of soil function at plot scale?

RQ 2. What is the relative influence of trees, hillslope and alley width on common indicators of soil function at field scale?

RQ 3. What is the relative influence of trees and on-farm tree placement on soil hydrology and natural flood management potential at catchment scale?

The purpose of these research questions is to leverage appropriate research sites in the UK to isolate and strengthen important, unaddressed, linkages between design/context and outcomes, at three different scales of observation.

1.6. Research sites

Three sites were chosen to investigate the above questions, with each corresponding to an increasing scale of outcomes. The first is the University of Leeds research farm, Tadcaster, UK, at which an agroforestry experiment has been underway since 1988 (Fig. 1.4 a). This site was chosen to examine effects at plot scale. Uniquely at this site, woodland blocks of four individual high-value timber tree species have been planted on previously arable land, with each precisely replicated four times in close vicinity to each other. Using a field measurement, soil sampling and laboratory analysis approach this allows us to investigate the influence of both trees and specific species choice on soil outcomes, and to consider the soil depths at which each factor has a controlling influence over outcomes. Moreover, the replicated nature of the experiment allows us to consider the effect of random variability between each of the four plots, and the extent to which unknown controlling factors outweigh tree-planting and species-related differences between woodland and arable control areas on outcomes.

The second site – Barton Farm, Shillingford Abbot, UK – is a working organic silvohorticulture farm and was not set up as an agroforestry experiment. However, its age and unique layout facilitate consideration of effects on similar soil outcomes to the University of Leeds site at larger, field scale. Trees were first established at the site in 2002, in silvoarable rows with 12 m spacing (Fig. 1.4 b). In 2012, trees were planted in an adjacent field at 24 m spacing (Fig. 1.4 c), and to date the two areas have maintained almost exactly the same arable rotation in the cropped alleys, are on a similar slope with equivalent aspect, and share the same soil type. Using a field measurement, soil sampling and laboratory analysis approach I use this site to consider the effect of a single design factor – alley width – and a single contextual factor – hillslope position – on soil outcomes at row, alley and field scales.

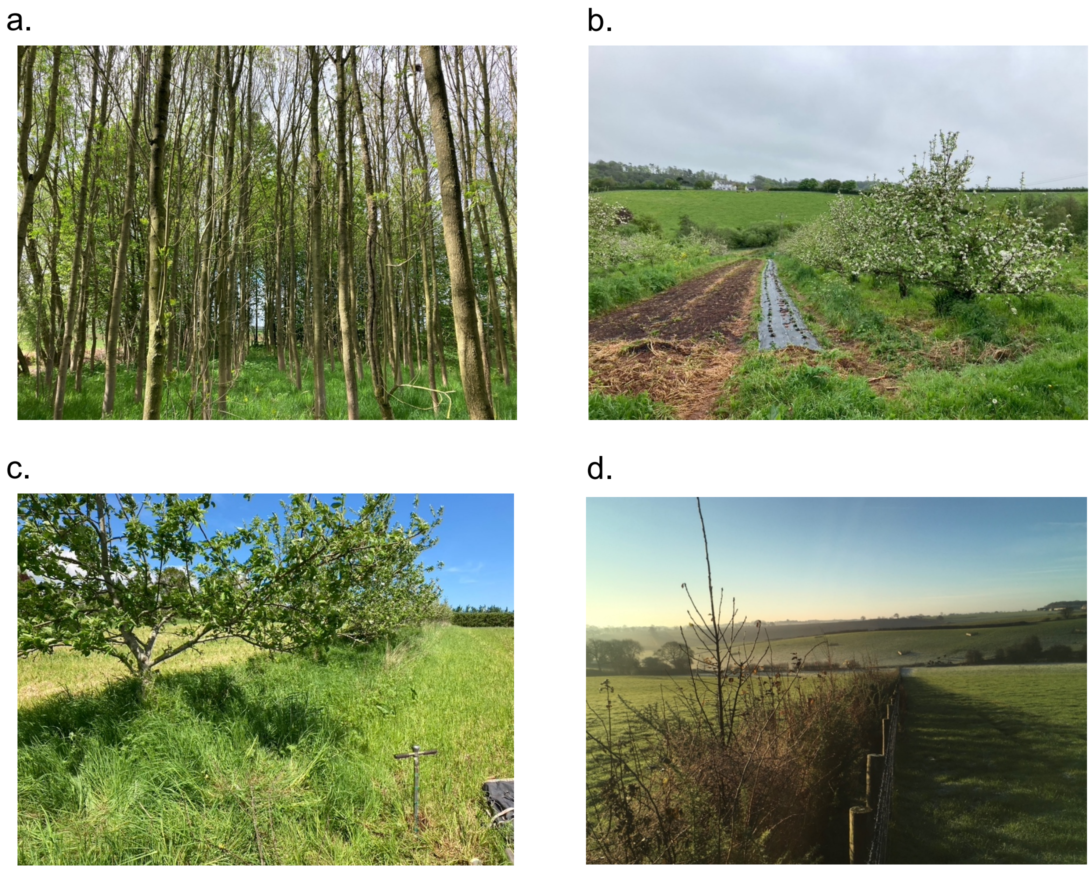

Figure 1.4 – a. Site 1 - high-value timber agroforestry at Leeds University Farm, Yorkshire, UK (continuous woodland blocks) for plot-scale study; b. Site 2a - silvoarable agroforestry alleys (12 m spacing) at Barton Farm, Devon for field-scale study; c. Site 2b - silvoarable agroforestry alleys (24 m spacing) at Barton Farm, Devon for field-scale study; d. Site 3 - hedgerows with agroforestry copses across valley, planted at Kiddal Quarry Farm, Yorks, UK for catchment-scale study.

The third site – Kiddal Quarry Farm, Leeds, UK (Fig. 1.4 d) – represents the largest of three scales studied, namely, small-catchment. Soil outcomes of agroforestry such as C storage and fertility can be scaled from the two preceding, smaller-scale approaches, such that the outcome on which this experiment focusses is hydrological functioning alone. Using a flood modelling approach validated by *in situ* field measurements, I addressed the unanswered question of whether lowland agroforestry can contribute to NFM, and which placement strategies of trees within a typical pasture farm can lead to the greatest flood risk reduction for an equivalent area of planting.

This approach carries a number of benefits despite being spread across three quite different geographies. Considering multiple soil outcomes and design/contextual controls at once allows us to examine co-benefits and trade-offs in agroforestry systems, a key research need in temperate zones. Secondly, comparing different scales of observation allows us to distinguish between stakeholder objectives and priorities for where benefits are generated in the landscape.

1.7. Research methodology

The outcomes addressed by RQ 1 and RQ 2 are at plot and field scale, respectively, and therefore a methodology focussed on *in situ* soil sampling was deemed most appropriate. For RQ1, at the University of Leeds study site, I considered agroforestry outcomes in the forest floor, topsoil and subsoil horizons, and soil sampling was therefore undertaken across all three horizons to 50 cm depth (at 10 cm depth intervals) for each treatment and replicate, with samples subsequently processed in the laboratory for determination of relevant soil indicator values. Soil samples were supplemented by infiltration measurements at the soil surface for determination of saturated hydraulic conductivity.

For RQ2, addressed at Barton Farm, Shillingford Abbot, soil sampling was undertaken to the same sampling depth as at the first site and the same soil indicators analysed. However, as the focus was on spatial differences with hillslope and across tree rows, spatial coverage was prioritised over vertical resolution, and the subsoil samples (20-50 cm) were combined into a single sample. Forest floor material was not considered, as the focus was no longer on inter-species differences in litter additions. Instead, field measurements of tree height and diameter at breast height (DBH) were taken for the purpose of estimating above- and below-ground biomass. Field measurements of surface infiltration were also collected.

Addressing RQ 3 required a catchment-scale view of outcomes in response to spatially-explicit land-use changes. A rainfall-runoff modelling approach was therefore chosen, and SD-TOPMODEL (Gao et al., 2015) deemed ideal given its capacity to predict runoff according to spatially-variable input parameters. Land-use scenarios corresponding to different tree placement strategies were analysed for their effect on flood peak size and timing following arrival of synthetic storm events at Kiddal Quarry Farm, Leeds. Validation of model parameters was undertaken using rainfall and catchment outlet stream level data collected at the study site.

1.8. Thesis outline

The subsequent research chapters (Ch. 2 – 4) of this thesis have been prepared, reviewed or published as complete manuscripts, after which a synthesis of findings is presented in Chapter 5. The three research chapters are as follows:

Chapter 2 – *Impacts of arable reforestation on soil carbon and nutrients are dependent upon interactions between soil depth and tree species*. This chapter begins the research by considering small, 'single tree' effects at the plot scale. It assesses the influence of three context/design factors – tree-planting, tree species and random between-plot variability – on three soil outcomes – SOC storage, nutrient availability and soil physical properties – within different components of the soil column, asking which factors have the largest influence at different depth scales. It has been published as: Judson, J.B., Chapman, P.J., Holden, J., Galdos, M.V., 2024. Impacts of arable reforestation on soil carbon and nutrients are dependent upon interactions between soil depth and tree species. *Catena*, 247, 108465.

Chapter 3 – *Alley width and slope position influence soil carbon storage, nutrient dynamics and hydrology at a mature silvoarable site, SW England*. This chapter increases scale to 'single-rows' of trees at the field scale. It considers the influence of two more factors, hillslope and alley width, on each of the soil outcomes already described, and how agroforestry outcomes are distributed laterally from tree rows into cropped alleys, with hillslope and with depth in the soil column. It has been published as: Judson, J.B., Chapman, P.J., Holden, J., Galdos, M.V., 2025. Alley width and slope position influence soil carbon storage, nutrient dynamics and hydrology at a mature silvoarable site, SW England. *Catena*, 260, 109439.

Chapter 4 - *Spatial layout of trees on farms influences magnitude and timing of river flow peak*. This chapter concludes the research at catchment scale, asking whether agroforestry planted on-farm can have a meaningful influence on flood peak size and timing and contribute to NFM, and which spatial planting strategies generate the most significant flood risk reduction benefit for an equivalent area of planting.

Chapter 5 is a synthesis chapter which summarises findings of chapters 2 – 4 in the context of themes discussed in Chapter 1. It considers wider implications of findings for agroforestry and environmental policy and considers limitations and further work arising from this thesis.

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Chapter 2. Impacts of arable reforestation on soil carbon and nutrients are dependent upon interactions between soil depth and tree species

Abstract

Recent interest in temperate farm woodland has focussed on strengthening delivery of ecological and economic benefits from land. However, impacts of temperate farm woodland on soil properties and carbon inventories are poorly studied. With field samples and measurements taken at 35-year-old agroforestry experiment we determine how functioning in three components of the soil column (forest floor, topsoil (0-30cm) and subsoil (>30 cm)) respond to land-use change, tree species choice and small-scale random variability in soil properties. Tree species influenced soil nutrient dynamics in the forest floor and topsoil, with Hazel forest floor material 27% less concentrated in phosphorus (P) but containing 50% more soil organic carbon (SOC) stock than Cherry or Sycamore. Change in land use from arable to woodland controlled soil bulk density, organic matter content and C storage in topsoil, with 15% (11.8 t ha^{-1}) more SOC stock in 0-30 cm soil beneath woodland compared with arable. In subsoil, tree species and land cover influence over soil functioning was insignificant. Notably, no net difference between arable and woodland soil C storage was found when the 0-50 cm part of the profile was considered as a whole, although net C storage was highly variable by plot. 35 years following planting, soil structure and SOC storage were only different in the forest floor and topsoil compared to the adjacent arable system. Each soil component therefore has its own functioning 'signature' in response to afforestation. Future policy support for farm woodland must account for this complexity.

2.1. Introduction

Competition in temperate landscapes between economic production and ecosystem service delivery has prompted significant interest in reforestation (Ashwood et al., 2019). This is particularly the case on farmland, where agroecological practices such as agroforestry may simultaneously be capable of delivering ecological and economic benefits (Burgess, 1999; Araujo et al., 2012; Torralba et al., 2016; Judson et al., 2023). However, success in benefit delivery from afforestation requires thorough understanding of system interdependencies: between inputs, such as species choice or system design, and outputs such as soil function. These are less well studied in temperate areas: the complexity of tree-soil-atmosphere interactions makes it challenging to isolate specific inputs from other confounding variables, and interdependencies are less well-characterised.

Incorporating trees into farm systems has been cited by bodies such as the IPCC (2014) and UK Climate Change Committee (CCC, 2020a) as an essential component of future climate resilience, and many national (and supra-national) carbon inventories rely on carbon dioxide (CO_2) drawdown in woodland and soils to meet emissions reduction obligations (Smith et al., 2022). The UK's Climate Change Committee recommends that woodland be incorporated on a minimum of 10% of farmland by 2050 in order to reach net-zero emissions under their balanced pathway (CCC, 2020b). Yet scaling up farm woodland creation in temperate areas is limited by lack of understanding of benefits, lack of market incentives and cost (Sollen-Norrlin et al., 2020). Directing financial incentives towards best land-use change is always challenging at policy level, and in the case of afforestation, land managers lack contextually specific information on planting design for the successful delivery of multiple benefits. Many temperate studies on farm woodland have focussed on soil C sequestration (De Stefano and Jacobson, 2018; Mayer et al., 2022), with others considering outcomes such as nutrient dynamics (Oelbermann and Voroney, 2007), hydrological functioning (Marshall et al., 2014; Monger et al., 2022) and biodiversity improvements (Varah et al., 2013) or combinations of variables and drivers (Amorim et al., 2022).

There has been less focus on specific drivers of soil function in the context of arable land afforestation – such as soil type, tree species, climate, topography, historic management – nor on depths within the soil column at which they have strongest influence. As with many sustainability challenges, delivering ecosystem benefit from farmland soils is a wicked problem in which stakeholders have differing reference frames and objectives from which to judge outcomes (Rittel and Webber, 1973; Bouma et al., 2011). In the case of afforestation, different stakeholders may place emphasis on soil functions in different depth horizons. For example, practitioners requiring nutrient availability for crop production are looking for benefits in topsoil (e.g. Donn et al., 2014), whereas stakeholders interested in landscape soil carbon inventories or flood management need to consider the response of subsoil in addition to shallower horizons to afforestation (e.g. Rogger et al., 2017; De Stefano and Jacobson, 2018).

In light of this, we sample a carefully-replicated 35-year-old farm woodland experiment in Yorkshire, UK to consider the response of three soil depth horizons – forest floor material, topsoil (0-30 cm) and subsoil (>30 cm) – to three drivers of soil function, in order to assess their relevance to different stakeholders. The first driver is land-use change: we consider how different horizons of the soil profile respond to woodland planted on previously arable land, and the effect of land-use change on properties such as soil organic carbon (SOC) stock, hydrological functioning and nutrient availability. The second driver is tree species: we consider how key soil properties are controlled by three broadleaf timber species commonly planted for agroforestry. These are sycamore (*Acer pseudoplanatus* L.), cherry (*Prunus avium* L.) and hazel (*Corylus maxima* Miller, cv. Kentish Cob). The third is small-scale random variability in soil properties (pre-planting) between experimental replicates, whereby we demonstrate the extent to which outcomes are controlled by pre-existing conditions, irrespective of treatment. The aim of the study is to determine how each of the three depth horizons respond to drivers in terms of delivery of ecosystem services from arable soil following afforestation.

2.2. Methods

2.2.1. Study site

The University of Leeds research farm is a commercially-run mixed arable and pasture farm 5 km west of Tadcaster, northern England. Mean annual precipitation between 1992 and 2021 was 639 mm (Met Office, 2006) and mean annual temperature was 9.5°C (Met Office, 2019). The soils in this study are Calcaric Endoleptic Cambisols from the Aberford Series: well-drained calcareous brown earths extensively found on low-dipping Permian and Jurassic limestones in both mid- and northern England (Cranfield University, 2022). Aberford soils in the UK are commonly under arable cultivation with much more limited areas of pasture. Soil depths on the University Farm range between 30 cm and 70 cm depth, broadly in line with depths found elsewhere for the soil series.

Figure 2.1 – Map of experimental plots used in this study at University of Leeds research farm showing sampling locations by species (coloured crosses) and plot number (white figures). Sampled woodland areas shown as dotted boxes, with ash areas shown in grey. Upper right inset shows site location.

An agroforestry experiment was established at the farm in 1988 (Fig. 2.1). Four replicate plots of approximately 110 x 110 m area were planted close to one another, each containing a silvoarable agroforestry plot and adjacent woodland areas. Each site included a 48 x 30 m arable 'control' area in an adjacent, conventionally cropped field. The arable control areas have been under continuous, intensive management from 1994 and since 2009 have been in a four-year rotation of winter wheat, oilseed rape, potatoes or vining peas, and winter or spring barley. They are ploughed annually and fertilised with 140-150 kg N ha^{-1}, 70-86 kg K ha^{-1}, 23 kg P ha^{-1} and 22 kg S ha^{-1}, in addition to 8 t ha^{-1} pig manure in 2018 (Guest et al., 2022). Today, two of the arable control areas remain, with two (corresponding to plots 1 and 2) unusable having been converted for separate experimental trials. Plot 1 and 2 control areas were therefore replaced with two areas to the west of Plot 1 for this study (Fig. 2.1).

The woodland areas of the agroforestry experiment were each planted with three furniture timber species – ash (*Fraxinus excelsior* L.), sycamore (*Acer pseudoplanatus* L.) and wild cherry (*Prunus avium* L.) – in adjacent 26 x 20 m rectangular blocks. These were established with 130 trees per block at a 2 x 2 m spacing, a density typical of conventional forestry. A fourth species – hazel (*Corylus maxima* Miller, cv. Kentish Cob) – was planted as an orchard in 4 x 4 m spacing. Surrounding each plot and enclosing the silvoarable and forestry areas is a windbreak of 11 cultivars of poplar (*Populus alba* L.) and four of willow (*Salix sp*). The woodland areas represent a unique opportunity for study as they are carefully replicated, subdivided by species and analogous to mature areas of farm woodland which might be found elsewhere in either arable or pastoral contexts.

2.2.2. Sampling strategy

Soil samples were first collected in May 2022 beneath three of the four species in the woodland areas – hazel, wild cherry and sycamore – and in arable control areas (Fig. 2.1). Within the woodland areas, ash blocks were omitted from our study as the spread of ash dieback has meant farms are no longer planting ash. Sampling sites beneath each of the three woodland species were replicated across the four plots, producing 12 woodland sample locations in total (Fig. 2.1). A location was chosen near the centre of each block, at the midpoint of the shortest distance between two adjacent trees. In the case of the hazel orchard this was 2 m from the nearest tree; for the sycamore and wild cherry plots this was 1 m from the nearest tree. Four control sample locations (arable, which were all in winter wheat in 2022) were sampled, with two adjacent to Plot 1 and one adjacent to each of Plots 3 and 4. Control samples were taken within the wheat crop itself, away from tramlines. The entire sampling procedure was repeated in February 2023.

Soil samples were extracted from five depth intervals between 0 and 50 cm. At each sampling location a 5 cm diameter ring corer (Eijkellkamp, Holland) was used to extract intact 100 cm^3 soil cores at 2.5-7.5, 12.5-17.5, 22.5-27.5, 32.5-37.5 and 42.5-47.5 cm (representing 0-10, 10-20, 20-30, 30-40 and 40-50 cm, respectively) below the surface for the determination of bulk density and moisture content. For the remainder of the study, we refer to 'surface soil' as 0-10 cm depth, 'topsoil' as 0-30 cm depth and 'subsoil' as >30 cm depth. At three woodland sample locations (within plots 1 and 4) limestone bedrock was reached at depths shallower than 50 cm, such that deeper layers could not be sampled. Separate, loose, soil samples from the same sample locations and depth intervals were collected for determination of SOC, nitrogen (N) and phosphorus (P) concentrations.

L (litter), F (fermented) and H (humic) organic horizons (hereafter 'forest floor') were sampled from the forest floor at each of the woodland sample locations. A 50 cm^2 quadrat was laid at the midpoint of the shortest distance between two trees within each of the woodland blocks. From the quadrat, the L horizon was collected as loose leaf litter and twigs and placed in plastic bags. Removal of the litter exposed the F and H horizons beneath. The F and H horizon was subsequently sampled using a single 5 cm diameter (100 cm^3) bulk density ring gently hammered into the underlying soil surface, with contents retained intact and returned to the laboratory for analysis.

Tension infiltrometers were used in February 2023 to measure infiltration at a constant tension of -2 cm. Eight measurements were taken for each tree species and arable control across all plots. Infiltration estimates were combined with constants for clay loam soil derived from the method of van Genuchten (van Genuchten and Nielsen, 1985) to estimate near-surface saturated hydraulic conductivity (K_s). This was carried out at the same distance to tree used for soil sample locations across the woodland and arable sites.

A space-for-time substitution approach was used throughout, in which a point-in-time soil sample from beneath woodland species was compared against arable soil samples, with arable areas assumed to represent baseline ($t = 0$) state before afforestation. This widely used approach (Cardinael et al., 2015; Biffi et al., 2022) assumes that the field and woodland sites were equivalent prior to land-use change – a reasonable assumption given the woodland plots were situated in the corner of arable fields.

2.2.3. Laboratory analysis

2.2.3.1. Bulk density and organic matter

Soil samples taken intact with the 100 cm^3 ring corer were weighed and subsequently oven dried at 105°C for 12 hours before being weighed again for the determination of bulk density. Moisture content was determined for each of these samples by comparing soil mass before and after oven drying at 105ºC. Roots and stones were extracted and weighed in order to correct for their presence in the soil. Bulk density was calculated by subtracting the root and stone fraction from the final mass. Finally, oven-dry samples were heated to 550°C for 12 hours in order to determine soil organic matter content (%) by the loss on ignition method. These were subsequently weighed, with loss on ignition determined as the change in mass between 105°C (oven-drying) and 550°C (ignition), divided by oven-dry mass.

Loose, field-moist L horizon material from the forest floor was weighed and oven dried at 65°C for five days before being weighed again to determine moisture content. Field-moist ring samples of combined F and H (hereafter 'FH') horizons and underlying mineral soil were returned intact to the laboratory. Colour change was used to determine the boundary between F and H horizons and underlying mineral soil, and combined FH thickness was measured and recorded using electronic Vernier callipers. F and H horizons were combined due to difficulty in distinguishing and separating them. For each sample, the FH horizons were cut off horizontally at the measured depth and weighed, before being oven dried at 105°C for 12 hours and then reweighed to determine moisture content. Uncorrected bulk density of the FH-horizon was determined using oven-dry mass in combination with the ring diameter and measured thickness of the FH horizon.

2.2.3.2. Soil nutrients and pH

Plant-available N and Olsen's phosphorous (Olsen, 1954) were determined using field-moist soil samples from surface soil (0-10 cm). Loose samples were returned to the laboratory and immediately homogenised by passing through a 5 mm sieve. For determination of N concentration, approximately 10 g field-moist sample was combined with 50 mL 1M KCl solution and shaken for 1 hour at 150 cycles min^{-1} using a shaker table. These were subsequently passed through Whatman 42 filter paper into centrifuge tubes, with NO_3-N and NH_4-N content determined using a Skalar San ++ (Skalar Analytical B.V., Netherlands) continuous flow auto-analyser. Following available N determination, remaining field-moist sample was dried at room temperature and passed through a 2 mm sieve. Approximately 2.5 g air-dried soil was weighed into a shaker bottle, combined with 50 mL of 0.5M $NaHCO_3$ solution and mixed for 1 hour at 150 cycles min^{-1} using a shaker table. These were subsequently passed through Whatman 42 filter paper into centrifuge tubes, with Olsen's phosphorous (PO_4-P) content determined using the same continuous flow auto-analyser.

For L- and FH-horizon samples determination of Olsen's phosphorous was not possible and total P was instead determined using a digestion method. Approximately 0.4g of air-dried and shredded (L) or <0.5 mm sieved (FH) sample was combined with 4.4 mL digestion reagent consisting of 30% hydrogen peroxide (87.5 mL), selenium (0.105 g), sulphuric acid (105 mL) and lithium sulphate (3.5 g). Digestion of plant material was carried out at slowly increasing temperature up to 300°C until the digest became colourless, with total P in the extract determined colorimetrically using a Skalar San ++ continuous flow auto-analyser.

Following nutrient determinations, soil pH was measured using the <2 mm fraction of air-dry soil. Approximately 20 g air dry soil was combined with 40 mL deionised water and stirred for 15 minutes. The pH was measured in the deionised water only, before being measured again after the addition of 250 μL $CaCl_2$.

2.2.3.3. Total C, total N and SOC

Total N content of mineral soil and FH samples was determined using loose samples dried at room temperature. Following nutrient determination, dried samples were ground to <150 μm using a Retsch MM400 ball mill (RETSCH GmbH, Germany). Approximately 4 mg of <150 μm sample was weighed using a six-figure balance into tin capsules, crushed into a small cube to remove air from the sample, and subsequently introduced into an Elemental Vario EL cube (Elementar Analysensysteme GmbH, Germany) combustion analyser to determine concentration of total carbon and nitrogen. To determine organic C content of mineral soil and FH samples the procedure was repeated, with the exception that all samples were weighed into silver capsules and 30 μL of 15% HCl added to each sample to remove carbonates. Samples were left to react before being oven dried for 2 h at 80°C and analysed for SOC content with the combustion analyser. For L-horizon samples the above process was repeated, with the exception that plant material dried at 65°C was shredded to <500 µm using a Retsch SM100 cutting mill (RETSCH GmbH, Germany) before being introduced into capsules for the combustion analyser. C stock was calculated as the product of measured SOC content and dry soil/forest floor material mass per unit area.

2.2.4. Mineral soil mass corrections

Comparing soil properties to fixed depths following land-use change, such as afforestation of previously arable land, can lead to over- or underestimations of ratio-based soil properties (von Haden et al., 2020). All ratio-based soil measurements were therefore normalised according to a reference quantity of mineral soil in order to correct for land-use change effects on bulk density and SOM. For this purpose, the aggregated control area samples were used as they are assumed to represent $t = 0$ under space-for-time substitution. Mineral soil mass was calculated as the mass of dry soil per unit area in the aggregated control area samples to five reference depths, corresponding to sample depths used in the study (0-10 cm, 0-20 cm, 0-30 cm, 0-40 cm and 0-50 cm). For a given soil property in woodland treatments (e.g., moisture, SOC), cumulative mass of the property was calculated to the same reference depths and plotted against cumulative mineral soil mass. A monotonically-increasing cubic spline function (von Haden et al., 2020) was used to fit these data, from which corrected (or equivalent soil mass - ESM) values were interpolated using cumulative mineral soil values from the reference (Control) areas. Using this procedure, ESM corrected values were calculated for SOC (concentration and stocks), total N, SOC:N, PO_4-P, NO_3-N and total plant-available N (NO_3-N + NH_4-N).

2.2.5. Statistical analysis

Data used for calculation of bulk density, SOC stocks, SOC/N ratio and moisture were tested for normality (Shapiro-Wilk) and homoscedasticity (Bartlett) in order to meet assumptions for ANOVA and pairwise Tukey tests. Where assumptions were not met, a non-parametric Kruskal-Wallis test was used in place of ANOVA, followed by pairwise Dunn's tests with a Bonferroni correction. All tests were undertaken using *SciPy* (Virtanen et al., 2020) and *statsmodels* (Seabold and Perktold, 2010) within the Python environment (v. 3.10). For land cover comparisons between woodland and arable treatments, Tukey-Kramer mean comparisons were used (owing to unequal sample sizes), with statistical significance evaluated at $p < 0.05$. Comparison between tree species treatments was undertaken using ANOVA mean comparison tests, with significant differences between individual species treatments evaluated using post-hoc pairwise Tukey mean comparisons.

Following land-cover and species treatment comparison, the significance of land cover, species and random plot variability as driving effects was determined for soil functioning indicators. A linear mixed model approach was considered for evaluation of species vs plot variability effect strengths, with plot variability treated as a random effect. However this model was not feasible as the Control treatment is no longer replicated with the same plot areas as Cherry, Hazel and Sycamore treatments. Thus, species and plot variability effects were compared using two-way factorial ANOVA which considers interactions between factors as well as their individual contributions. This was initially undertaken with individual species and plot effects and an interaction term combining them, after which, if the interaction term was not significant at $p < 0.05$, the test was repeated without the interaction term. The land cover effect could not be included in factorial ANOVA due to the absence of control (treeless) data within species and plot categories – thus pre-determined Tukey-Kramer p-values were used to determine significance of the land cover effect on soil function indicators. Although this method does not permit *relative* comparison of fixed and random effect strengths on soil properties, the constraint is imposed by changes to the study site design since planting, and we can nonetheless determine which effect is making significant contribution to variance in soil properties at any given depth.

2.3. Results

2.3.1. Bulk density, SOM and hydraulic conductivity

Mean bulk density of arable soil (1.46 ± 0.04 g cm^{-3}, $n = 8$) was nearly 20% higher than for woodland soil (1.28 ± 0.02 g cm^{-3}, $n = 24$) at 0-10 cm depth (Fig. 2.2a, Supp. Table A.2, $p < 0.001$). This was accompanied by a significant difference in SOM between woodland (8.12 ± 0.23%) and arable (6.82 ± 0.62 %) soils at 0-10 cm ($p = 0.022$). At all other depths there were no significant differences in bulk density between arable and woodland soils. Surface (0-10 cm) soil bulk density was not significantly different between each of the tree species treatments (Fig. 2.2b, Cherry 1.29 ± 0.02 g cm^{-3}, Hazel 1.29 ± 0.03 g cm^{-3}, Sycamore 1.26 ± 0.03 g cm^{-3}, n = 8, p = 0.672).

Surface soil K_s varied significantly between woodland and arable treatments (Supp. Table A.3). Mean K_s was ~2.5 x faster (3.24 mm hr^{-1}, 2.64 – 3.99) in arable soil than woodland soil (1.31 mm hr^{-1}, 1.15 – 1.49) (p = 0.003). No significant differences in surface K_s were observed between tree species (p = 0.498).

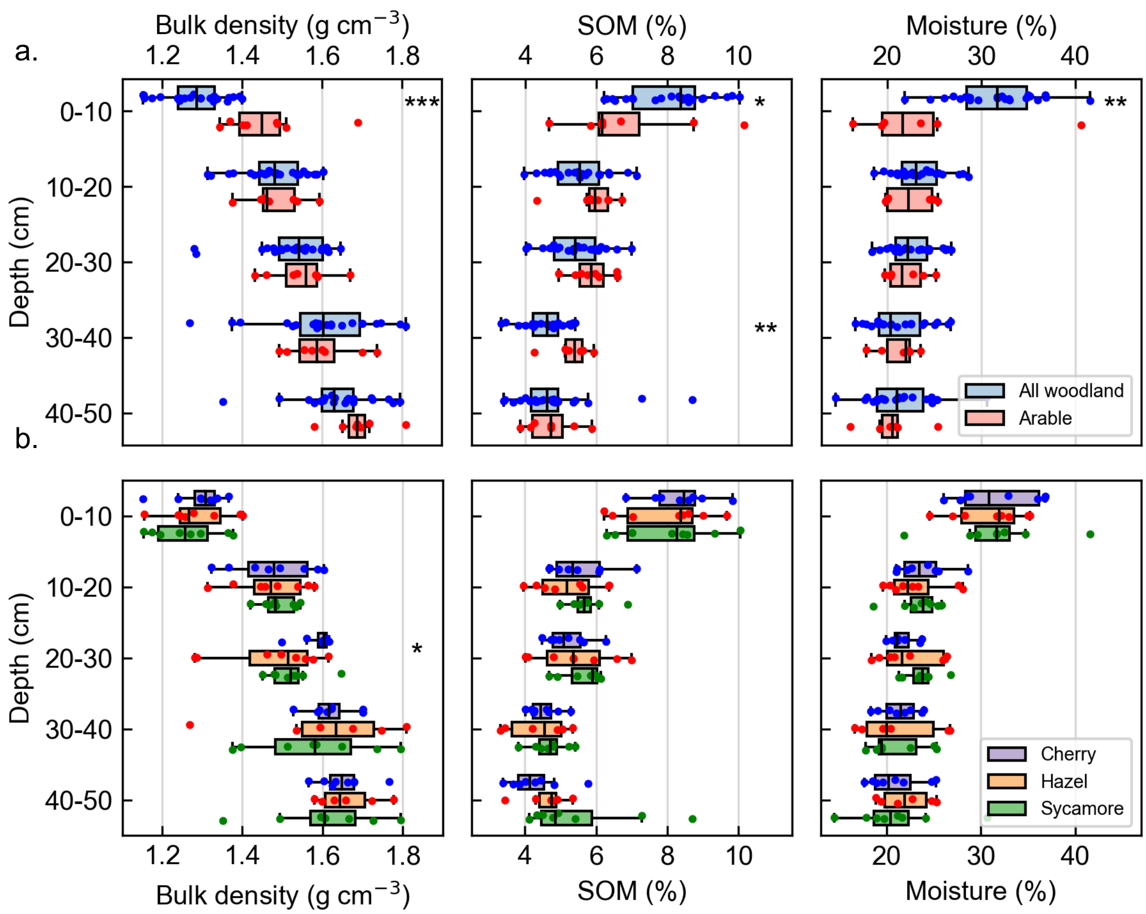

Figure 2.2 – Variation in bulk density, soil organic matter (SOM) and moisture with increasing depth for a. afforested plots (n = 24) and arable control plots (n = 8) and b. Cherry (n = 8), Hazel (n = 8) and Sycamore (n = 8). Coloured dots show actual sample values. Asterisks denote significant differences between values at the same depth (* – p < 0.05, ** – p < 0.01, *** – p < 0.001).

2.3.2. Soil organic carbon

In the top 10 cm, SOC content was significantly higher in woodland plots (3.05 ± 0.11%) than in arable areas (1.91 ± 0.06%, $p < 0.001$) (Fig. 2.3a, Supp. Table A.2). In deeper layers, however, the difference was reversed. Between 20 and 30 cm depth, soil in arable fields had higher SOC content (1.73 ± 0.06%) compared with woodland (1.48 ± 0.05%, $p = 0.010$), and at 30-40 cm depth the difference was even larger (arable 1.33 ± 0.08%, woodland 0.90 ± 0.05%, $p < 0.001$). No significant differences in SOC content were observed between any of the individual tree species treatments at any depth interval (Fig. 2.3b), nor were there any significant differences in the SOC content of either the litter or FH horizon of the forest floor beneath each of the tree species treatments.

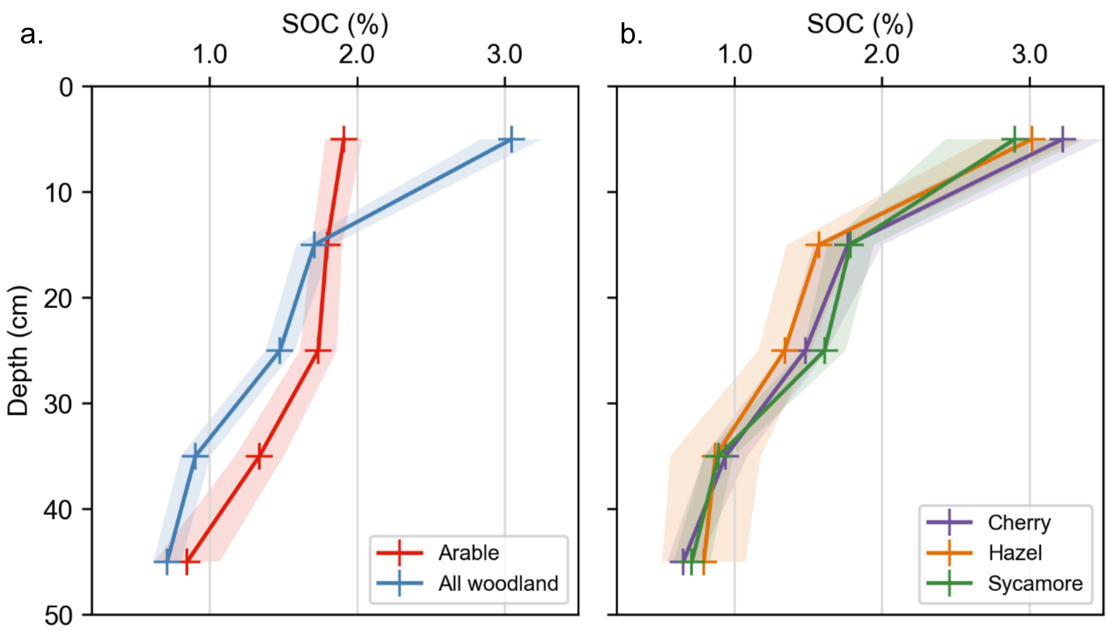

Figure 2.3 – SOC concentration with depth for a. all woodland and arable (land cover, left) and b. individual tree species (right). Shaded areas show 95% confidence intervals.

Differences in SOC stock between arable and woodland treatments were significant in topsoil (0-30 cm). This difference was most pronounced at 0-10 cm depth ($p < 0.001$), with a significant difference in SOC stock also observed at 0-30 cm ($p = 0.027$), but not at 0-50 cm ($p = 0.944$, Fig. 2.4a, Supp. Table A.2). Between 0 and 10 cm depth, 62% more SOC stock was found in soil beneath trees (45.3 ± 1.7 t ha^{-1}) compared with soil beneath arable control plots (27.9 ± 1.0 t ha^{-1}) ($p < 0.001$). Including SOC stock from LFH horizons in the woodland total increased the woodland mean SOC stock to 54.3 ± 1.9 t ha^{-1}, a difference of 26.4 t C ha^{-1} (+95%) compared with arable plots.

Differences in SOC stock between individual species treatments were not significant at 0-10 cm. However, C stock within the combined LFH horizons varied significantly between species treatments. Although the total LFH biomass beneath the three species was not significantly different between species, there was significantly more C stored in the LFH horizons beneath Hazel (11.66 ± 1.26 t ha^{-1}) compared with either Cherry (7.86 ± 0.59 t ha^{-1}) or Sycamore (7.61 ± 0.57 t ha^{-1}) (p = 0.016) (Fig. 2.4b).

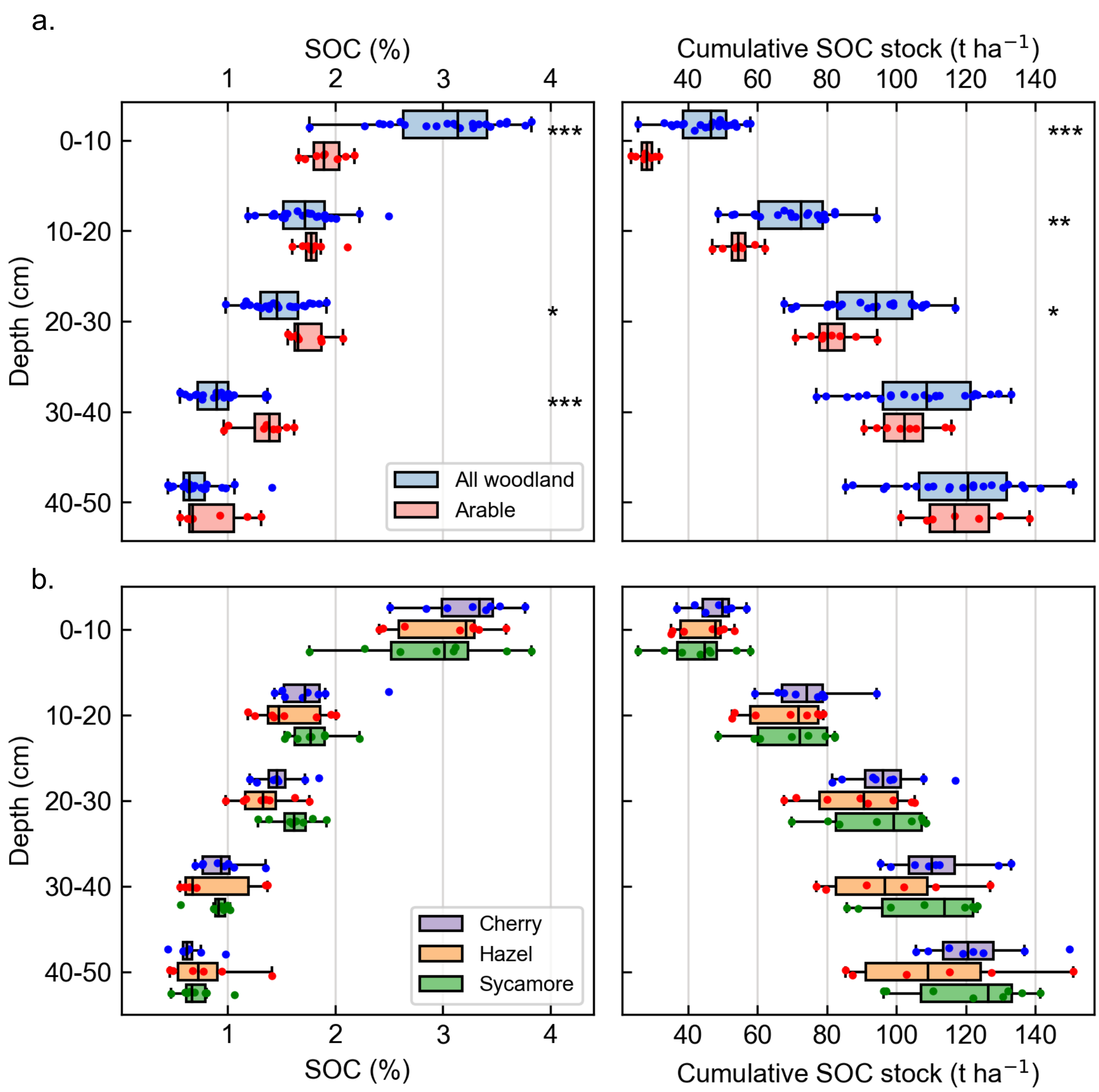

Figure 2.4 – Variation in soil organic carbon (SOC) as concentration and cumulative stock with increasing depth for a. afforested plots (n = 24) and arable control plots (n = 8) and b. Cherry (n = 8), Hazel (n = 8) and Sycamore (n = 8). Coloured dots show actual sample values. Asterisks denote significant differences between values at each depth (* : $p < 0.05$, ** : $p < 0.01$, *** : $p < 0.001$).

The whole topsoil (0-30 cm) of the afforested plots contained +15%, or 11.8 t C ha^{-1}, more SOC stock (93.3 ± 2.8 t ha^{-1}) compared with arable controls (81.5 ± 2.6 t ha^{-1}) (p =

0.027). Including the LFH horizons in the SOC total increased the topsoil woodland mean SOC stock to 102.4 ± 2.9 t ha^{-1}), a 20.9 t C ha^{-1} (+26%) difference compared with arable (Fig. 2.5) ($p < 0.001$). As observed at 0-10 cm depth, differences in 0-30 cm SOC stock between individual tree species were not significant for the topsoil.

When considering the total soil profile depth (0-50 cm), there was no significant difference in SOC stock between woodland (119 ± 4 t ha^{-1}) and arable (118 ± 5 t ha^{-1}) treatments ($p = 0.944$). This remained the case even when the SOC stock in LFH horizons were included in the woodland SOC total ($p = 0.251$), even though LFH C increased woodland C stock by 7.3% to 128 ± 4 t ha^{-1}.

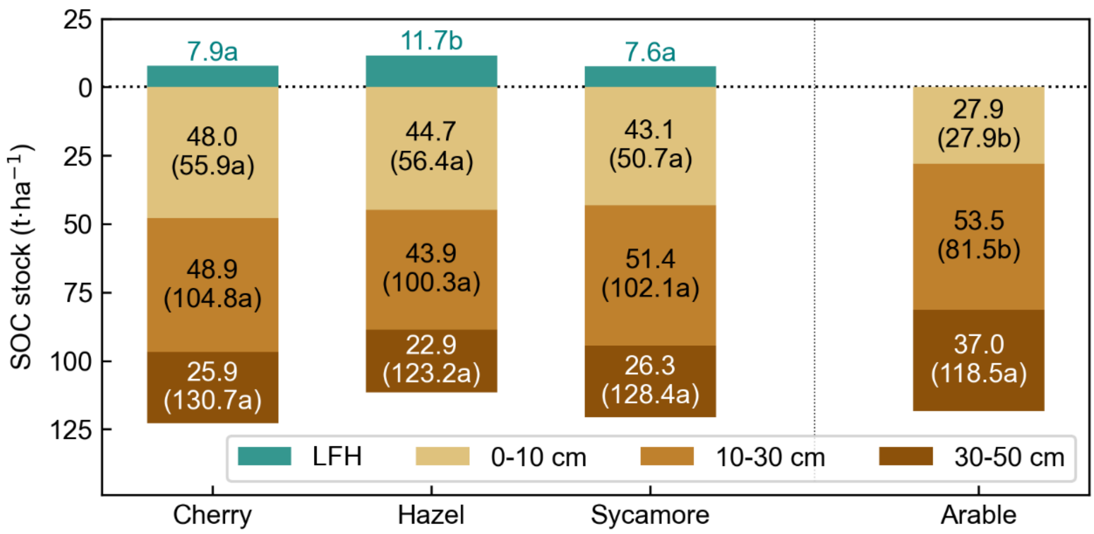

Figure 2.5. SOC stock by depth interval (LFH (tree litter and humus), 0-10 cm, 10-30 cm, 30-50 cm) for Cherry, Hazel, Sycamore and Arable treatments. SOC stock in t·ha-1 shown for each depth interval, with cumulative total (including LFH) shown in brackets. Letters denote within-depth significant differences between cumulative totals.

2.3.3. Soil nutrients and pH

The top 10 cm of arable soil contained more nitrate-N (NO_3-N) (15.7 ± 4.2 mg kg^{--1}, p = 0.011) and phosphate-P (PO_4-P) (32.3 ± 3.0 mg kg^{-1}, $p = 0.002$) compared with woodland soil (NO_3-N: 4.1 ± 1.0 mg kg^{-1}, PO_4-P: 15.7 ± 2.4 mg kg^{-1}). Differences in nutrient content were also observed between species treatments in both LFH horizons and mineral soil (Fig. 2.6, Supp. Table A.1). Total P in the L horizon was significantly different ($p = 0.011$) between species, with Hazel containing less total P (0.95 ± 0.06 g kg^{-1}) than litter beneath either Cherry (1.33 ± 0.06 g kg^{-1}) or Sycamore (1.26 ± 0.09 g kg^{-1}) (Fig. 2.6a). Total P in the combined FH horizons was not significantly different between species, although rank order matched L horizon interspecies differences (Fig. 2.6b). These differences had the same rank order (Fig. 2.6c) and were weakly correlated (Supp. Fig. A.1, $p = 0.101$) with interspecies variation in surface soil PO_4-P. At 0-10 cm soil depth, soil beneath Hazel contained less than half of the PO_4-P (8.2 ± 0.5 mg kg^{-1}, $n = 8$) than soil beneath Cherry (21.0 ± 3.0 mg kg^{-1}, $n = 8$) ($p = 0.002$). Sycamore surface soil PO_4-P (17.9 ± 5.7 mg kg^{-1}, $n = 8$) was more variable meaning differences with other species were not observed.

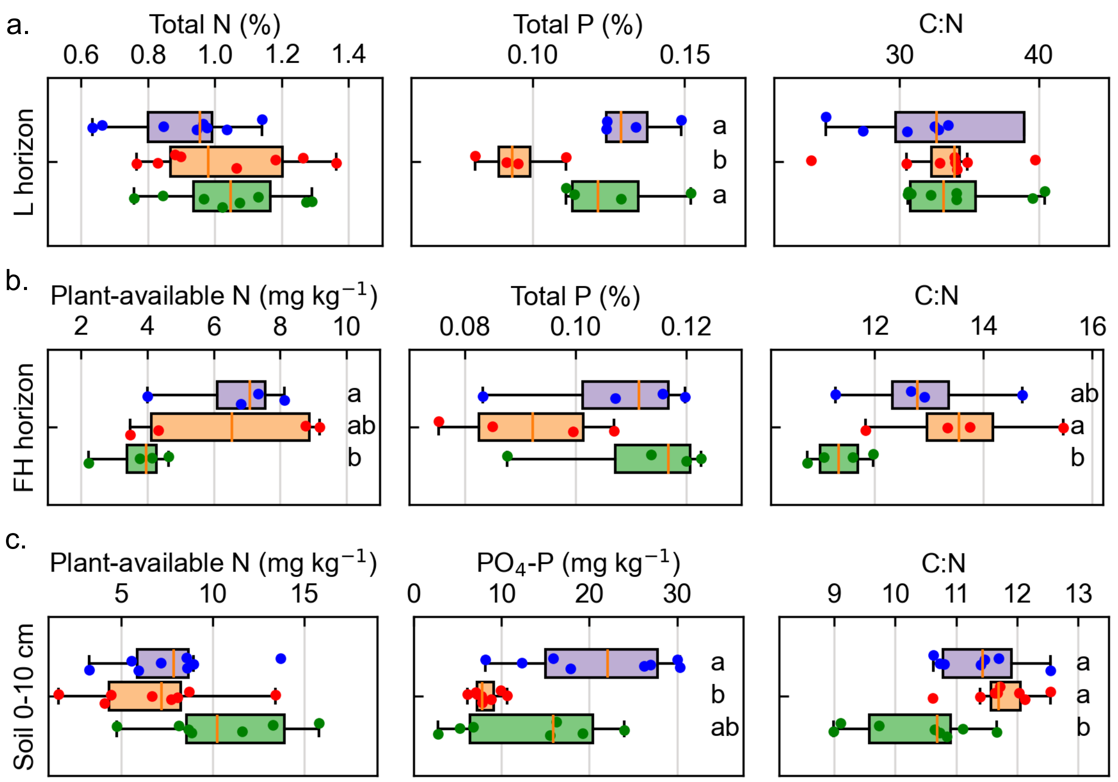

Figure 2.6. Nutrient concentrations and C:N for a. L-horizon, b. FH combined horizon and c. 0-10 cm soil between individual species treatments. Letters denote significant differences within horizons. Boxes coloured by tree species (orange – Hazel, purple – Cherry, green – Sycamore).

Total N was less variable than total P or PO_4-P between species treatments in both LFH and surface soil (0-10 cm) horizons. Interspecies total N differences in L (Fig. 2.6a) or FH (Fig. 2.6b) horizons were not observed at $p < 0.05$. However in the FH combined horizon, Cherry (6.57 ± 0.90 mg kg^{1}) and Hazel (6.44 ± 1.48 mg kg^{-1}) contained 1.75 times more plant-available N than Sycamore (3.70 ± 0.52 mg kg^{-1}) (Supp. Table A.1). Surface soil interspecies differences in NO_3-N were not significant ($p = 0.319$), while surface soil plant-available N ($NO_3 + NH_4$) showed greater but still non-significant ($p = 0.084$) variability (Fig. 2.6c).

The C:N ratio at 0-10 cm depth was significantly higher in woodland soil (11.25 ± 0.23) than arable soil (9.95 ± 0.36, $p = 0.006$). Between species, 0-10 cm Sycamore soil had a significantly lower C:N ratio (10.4 ± 0.3) than either Cherry (11.7 ± 0.4) or Hazel (11.7 ± 0.2) (Fig. 2.6c, Supp. Table A.2, $p = 0.013$). This corresponded with near-significant ($p = 0.075$) C:N differences in the FH horizons, with Sycamore FH having lower C:N (11.4 ± 0.3) than Cherry (12.9 ± 0.7) or Hazel (13.6 ± 0.7). Soil pH was not significantly variable in surface soil between arable and woodland treatments (in H_2O: $p = 0.094$; in $CaCl_2$: $p = 0.173$), nor between individual species treatments (in H_2O: $p = 0.689$; in $CaCl_2$: 0.823, Supp. Table A.3).

2.3.4. Relative impact of land cover, tree species and plot variability

The significance of each driving effect (land cover, tree species, and plot natural variability) on soil properties varied with depth (Table 2.1). Tree species had greatest impact on C and nutrient content of LFH horizons, whereas land cover accounted for variability in SOC stock, bulk density and hydrological functioning in the topsoil. In the subsoil, bulk density and cumulative SOC stock were most strongly controlled by spatial differences between plots.

Table 2.1 – Influence of land cover, plot variability and species effects on soil variables. Values are two-way ANOVA (plot variability, species) and Tukey-Kramer (land cover) p-values. Values shown in bold are significant at $p < 0.05$. Dashes indicate combinations which were either not sampled, or in the case of the cross terms (species*plot), not significant (after which ANOVA was rerun without cross term).

Sample type	*Effect*	**pH**	**BD**	**N**	**P**	**SOC**	**C:N**	**Ks**
Litter (LFH)	plot variability	-	-	**0.029**	†0.950	0.158	0.912	-
	species	-	-	**0.031**	†**0.044**	**0.012**	0.159	-
	species*plot	-	-	-	-	-	-	-
Shallow soil (0-10 cm)	land cover	0.094	**<0.001**	**0.023*	**0.002*	**<0.001**	**0.006**	**0.001**
	plot variability	0.069	0.341	0.128	0.181	0.095	**0.004**	**<0.001**
	species	0.499	0.666	0.075	**0.050**	0.435	**0.001**	0.415
	species*plot	-	-	-	-	-	-	**0.040**
Deep soil (40-50 cm)	land cover	-	0.197	-	-	0.944	**0.038**	-
	plot variability	-	**0.027**	-	-	**0.002**	0.537	-
	species	-	0.263	-	-	0.499	0.776	-
	species*plot	-	-	-	-	-	-	-

* Land-cover p-values for N and P greyed out due to influence of synthetic fertiliser on arable areas.

† P corresponds to L-horizon total P for LFH values, PO_4-P for soil measurements

2.3.4.1. LFH horizons

Tree species exerted a significant influence on the P concentration of the L-horizon (p = 0.044). In contrast, the L-horizon N concentration was not influenced by species (p = 0.290), although underlying FH-horizon plant-available N was controlled by tree species (p = 0.031). Plot variability had no effect on L horizon total P (p = 0.950) or N (p = 0.389) stock, although FH-horizon plant-available N was significantly different between plots (p = 0.029). Species differences were the dominant control on LFH total

SOC stock ($p = 0.012$), with no plot variability effect ($p = 0.158$). Land cover effects were not considered for LFH horizons due to the absence of LFH horizons in arable areas.

2.3.4.2. Topsoil

Land cover was a significant control on surface soil (0-10 cm) bulk density ($p < 0.001$), saturated hydraulic conductivity (K_s) ($p = 0.001$) and SOM ($p = 0.022$). Cumulative SOC stock in surface soil was also primarily controlled by land cover ($p < 0.001$). However, plot variability also significantly controlled K_s ($p < 0.001$) and SOM ($p = 0.020$), and SOC stock was controlled by plot variability ($p = 0.040$) when organic C from LFH horizons was included in the total.

Species differences exerted some control over surface soil nutrient stocks. PO_4-P differences were significantly controlled by species ($p = 0.050$), although species control over plant-available N was only significant at 90% confidence ($p = 0.075$). It was not possible to compare species nutrient effects with the influence of land cover due to the use of synthetic fertiliser on the arable land. No direct species control was observed on soil structural variables or SOC stock/concentration in surface soil. Surface soil C:N was the only indicator observed which was controlled by land cover ($p = 0.006$), species differences ($p = 0.001$) and plot variability ($p = 0.004$).

2.3.4.3. Subsoil

No land-cover effect on cumulative SOC stock was observed below 30 cm depth. At 50 cm, plot variability was the dominant control on both bulk density ($p = 0.027$) and cumulative SOC stock ($p = 0.002$). Notably, bedrock was reached at shallower depth in plots 1 and 4, potentially affecting compaction deeper in the soil profile. SOM concentration varied significantly between plots for all depth intervals between 0 and 40 cm (0-10 cm $p = 0.020$, 30-40 cm $p = 0.001$).

2.4. Discussion

Having presented differences in soil properties 35 years following afforestation of arable soil with three broadleaf species, we discuss implications of these differences for soil functioning at three different depth horizons, and how the contextual drivers of land cover, species and small-scale random variability control ecosystem service delivery at each depth. We summarise these effects and discuss how each component of the soil profile beneath woodland has its own signature in terms of which contextual controls are in operation, and on which soil outcomes (Fig. 2.7). It must be noted that not all depths were sampled for all soil properties and drivers (e.g. bulk density or K_s of forest floor, subsoil nutrient dynamics) and differences in management (fertiliser application in arable areas) limit some comparisons. Moreover, relative magnitude of driving effects was not discernible due to pre-existing experimental design and statistical constraints. However, this does not preclude drawing out useful high-level findings. We conclude by discussing implications, including the extent to which farm woodland can support climate resilience in future temperate landscapes.

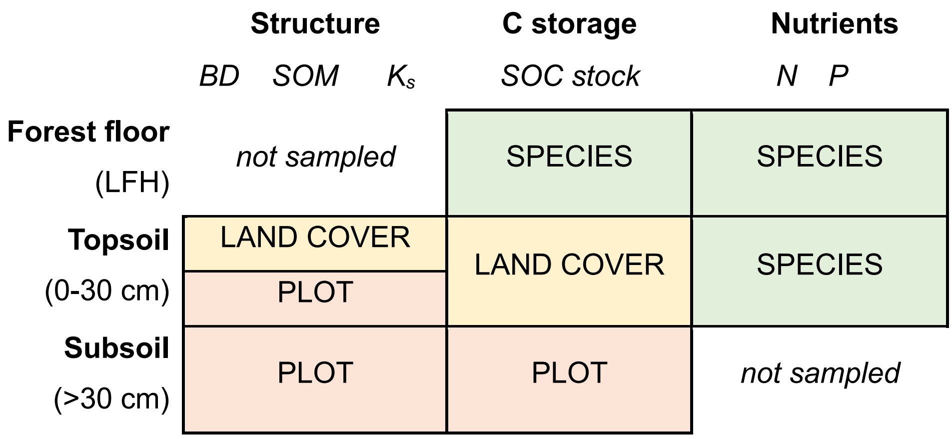

Figure 2.7 – Illustrative diagram of dominant high-level contextual controls (*land cover* – yellow, *tree species* – green, *plot variability* – red) on soil functioning by depth horizon and soil property type.

2.4.1. Forest floor functioning

Forest floor functioning was predominantly controlled by tree species (Fig. 2.7), which exerted significant control over chemical composition of litter material. This has implications for N and P dynamics – for example, significantly less P content was found beneath Hazel, and significantly less available N content found beneath Sycamore (Fig. 2.6). However, forest floor C storage was also species-controlled, with considerably more C stock beneath Hazel than either Sycamore or Cherry (Fig. 2.5).

Woodland LFH horizons in this study contributed a mean of 7% extra OC stock to SOC stored in the soil profile to 50 cm depth. A 7% mean forest floor contribution to total SOC stock is larger than contributions reported elsewhere. For example, Gao et al. (2014), in a study of two broadleaf and two conifer species of a similar age (~40 years) in a semi-arid temperate region of China, estimated a forest floor contribution of < 1% of SOC stock. Contributions of forest floor material to C stocks are important for wider-scale evaluation of C storage potential of agroforestry and farm woodland. Despite being less recalcitrant, the risk of forest floor C being removed is minimised as land beneath woodland is much less often disturbed than other agricultural areas.

The quantity of extra C accumulated in forest floor material varied by species, as was also found by Gao et al. (2014). Despite each species having accumulated a similar amount of LFH biomass over 35 years, LFH beneath Hazel contained ~ 1.5 times more C stock than either Sycamore or Cherry, although there was no difference in SOC storage between the species over the whole soil column (Fig. 2.5). Species choice must therefore be considered in estimating forest floor contributions to C storage as an ecosystem service, particularly in the case of species monocultures, and more work is needed to characterise interspecies differences in litter quality beyond the three under study.

Species differences in forest floor C and N content have implications for decomposition and transfer of litter C into surface soil. The C:N ratio of forest floor material controls its decomposition rate, however unlike studies with more sites/species for comparison (e.g. Cools et al., 2014), C:N was not significantly variable between species in our study. We did not find coupling between intraspecific litter C and surface soil OC variability, with surface soil SOC stock not controlled by tree species.

The N and P content of forest floor material was controlled by tree species (Table 2.1, Fig. 2.7) and was observed to have weak control on surface soil nutrient dynamics. Hazel litter contained less total P compared with Sycamore or Cherry, matching interspecies variability in surface soil PO_4-P, with 0-10 cm soil beneath Hazel containing less than half the PO_4-P of Sycamore or Cherry (Fig. 2.6). Litter and surface soil N concentration was more weakly species-controlled and coupled, yet plant-available N was species-controlled in the forest floor. Other studies have reported the influence of tree species on N, P and C dynamics. For example, Hoosbeek et al. (2018) found variable influence of tree species in a Nicaraguan silvopastoral system over N and P stocks in the topsoil: Jícaro (*Crescentia alata*) promoted greater N stock and Guácimo (*Guazuma ulmifolia*) promoted P stock. Similarly, Amorim et al. (2022) found surface soil (0-15 cm) beneath red oak (*Quercus rubra L.*) and pecan (*Carya illinoinensis* (Wangenh.) K. Koch.) to have significantly different C:N ratio and SOC concentration, which they attribute to distinctive leaf litter and nutrient inputs. Forest floor functioning and surface soil nutrient availability and fertility are thus closely linked with tree species choice. Afforestation is a key component of maintaining sustainable nutrient availability in agricultural soils, and species choice will depend on fertility needs of soils under management.

2.4.2. Topsoil functioning

Tree species effects were weaker in topsoil than the forest floor, and functioning was most strongly determined by land cover change. Topsoil has significant potential to promote C storage following land cover change from arable to woodland, with considerably more C stored beneath woodland at 0-10 cm (+17.4 t ha^{-1}, +0.50 t ha^{-1} $year^{-1}$), and a significant, but less pronounced, difference at 0-30 cm (+11.8 t C ha^{-}1, +0.34 t ha^{-1} $year^{-1}$, Fig. 2.4a). C is transferred to surface (and deeper) soil horizons from root and shoot litter and also root exudates (Jobbágy et al., 2001; Haichar et al., 2014), all of which are incorporated at higher rates in areas such as woodland where plant diversity is greater (Eisenhauer et al., 2017; Judson et al., 2023). Topsoil C storage was, however, insensitive to tree species (Fig. 2.4b) and there was no significant difference between plots.

Many other studies have demonstrated potential for extra C storage in afforested arable topsoils over time (Dawson and Smith, 2007; Upson and Burgess, 2013; De Stefano and Jacobson, 2018; Ashwood et al., 2019; Mayer et al., 2022), mainly reporting differences at 0-20 cm (Table 2). Ashwood et al. (2019) found that differences in topsoil SOC stock were greatest when land was converted from arable to woodland, with pasture having similar surface SOC stock to woodland. The difference of +16.0 t C ha^{-1} (+0.46 t C ha^{1} year^{1}) that we measured between arable soils and young woodland (35 year old) at 0-20 cm depth is smaller than that found by Ashwood et al. (2019) (+37.2 t C ha^{1}; +0.74 t C ha^{-1} year^{-1}) for the same depth interval, although it is larger than results found elsewhere for 0-20 cm SOC change following agriculture to agroforestry conversion (Table 2.2). Ashwood et al. (2019) included OC stock values for the partially decomposed OH litter layer above the soil surface, which contributed +9.1 t C ha^{-1} (+0.18 t C ha^{1} year^{1}), a very similar figure to the contribution from LFH in our study (+9.0 t C ha^{-1}; +0.26 t C ha^{1} year^{1}). Assuming the arable control had the same land-use history as the afforested plots prior to tree planting, woodland plots in our study contained 24.9 t C ha^{1} (+46%) extra OC in 0-20 cm topsoil and overlying LFH compared with arable fields.

Table 2.2 – Comparison of changes in SOC stock and sequestration rates between this study and studies of similar tree age, land-use change and climate zone. A 0-20 cm depth interval is used to aid comparison.

				Change in SOC stock					
				LFH/OH		0-20 cm		0-40 cm	
Study	**Location**	**Age**	**LUC**	Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})	Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})	Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})
This study	Yorks, UK	35	Arable to woodland	+9.0	+0.26	+16.0	+0.46	Not significant	
Mayer et al. (2022)	Various temperate	28	Various*	-	-	+7.0	+0.21	+10.1	+0.36
Cardinael et al. (2015)	Montpellier, France	18	Arable to silvoarable†	-	-	+17.0	+0.94	+17.7	+0.98
Ashwood et al. (2019)	Midlands, UK	50	Arable to woodland	+9.1	+0.18	+37.2	+0.74	+63.5	+1.27
Upson and Burgess (2013)	Beds, UK	20	Arable to silvoarable†	-	-	+7.2	+0.36	+22.0	+1.10

* Mayer et al. (2022) is a meta-analysis which includes the following land-use change (LUC) types: pasture to silvopasture, arable to silvoarable or hedge.

† Figures shown are for samples at similar distance-to-tree as in our study.

However, changes in SOC stock were very sensitive to depth. In contrast with other studies (Table 2.2), for the 0-40 cm depth interval (extending into the subsoil) we found no additional C stock resulting from afforestation even with LFH OC included, effectively cancelling out benefits measured in topsoil alone. This demonstrates the importance of sampling beneath topsoil to determine the holistic effect of land-use change on C storage, and the potential for soil to offset CO_2 emissions.

Topsoil nutrient dynamics were weakly controlled by species in the uppermost 10 cm with significant variability in PO_4-P content (Table 2.1), and, as mentioned above, this is likely to be the result of coupling with overlying forest floor material. However, it is difficult to contrast this with other drivers such as land cover due to the influence of synthetic fertiliser application in arable areas on nutrient dynamics. Determining afforestation control on nutrient dynamics would require further work with control and woodland areas under equivalent management, which we were not able to test.

Topsoil physical and hydrological properties were also determined by land cover change, with some small-scale plot variability. Surface soils under trees were significantly less compacted than soils under continuous arable cultivation, with bulk density of soil in the top 10 cm beneath woodland 17% lower than in arable areas. This is significant for topsoil functioning and agricultural productivity, as water and nutrient uptake by roots are inhibited in more compacted soils. The decrease in bulk density was driven by the significant increase in SOM, derived from organic matter input from leaf litter and root turnover. Litter input by woodland is a significant driver of surface bulk density changes. Topsoil bulk density (0-20 cm) beneath the 'young' category of trees examined by Ashwood et al. (2019) showed a decrease of similar magnitude to that in our study. They also found that the reduction in bulk density (at 0-20 cm depth) between arable and young secondary (50 – 60 y) woodland was also substantially greater than that observed between pasture and woodland. Similarly aged trees to those in our study, established on pasture in the UK, produced a smaller but significant reduction in bulk density (Upson et al., 2016), although hedges planted in pastures in northern England (Biffi et al., 2022) produced a similar change in bulk density (-17%) to this study, although for a greater depth (0-30 cm).

Topsoil *Ks* was controlled by land cover, and also showed considerable variability between plots (Table 2.1). However, in contrast with other studies (Marshall et al., 2014; Holden et al., 2019) it was found to be 2.5 times faster in arable areas than beneath woodland, implying better runoff mitigation in cultivated areas. It would be beneficial to test this finding using infiltration measurements at a range of hydraulic tensions to determine contributions of different pore size classes to overall saturated hydraulic conductivity, as the presence of cracks in drier arable surface soil may be contributing to the difference (Holden and Gell, 2009).

2.4.3. Subsoil functioning

In subsoil, the influence of land-cover change on soil functioning was minimal. Instead, soil functioning below 30 cm was controlled by pre-existing random variations in soil properties between plots (Table 1, Fig. 2.7).

Lack of land cover control on subsoil C storage differs substantially from other studies. Upson and Burgess (2013) found that the 20-40 cm depth interval contributed double the SOC (14.8 t ha^{-1}) compared with 0-20 cm depth (7.2 t ha^{-1}) 20 years after planting. Mayer et al. (2022) found in their meta-analysis that 80 % of studies observed an increase in SOC stock between 20 and 40 cm depth over a mean of 28 years, compared with a 70% increase in the top 20 cm of mineral soil. Only Cardinael et al. (2015) recorded a very small SOC contribution (0.7 t ha^{-1} over 18 years) between 20 and 40 cm depth. In our study, no significant difference in total SOC stock was observed under woodland compared to arable at 0-50 cm depth. In contrast, the SOC stock at 0-50 cm depth was strongly controlled by random plot differences. Sampling only at Plot 4 would imply a significant 22.2 t ha^{-1} (+18.7%) increase in woodland SOC stock compared with arable areas over 35 years, a sequestration rate of 0.63 t ha^{-1} $year^{-1}$; whereas sampling only at Plot 2 would imply a significant 18.1 t ha^{-1} (-15.3%; -0.51 t C ha^{1} $year^{-1}$) loss over the same time period.

We therefore found SOC storage over the whole (0-50 cm) soil profile to vary considerably more across the study site than as a direct result of tree planting. This implies that afforestation in some locations may not influence overall soil C stocks at all, and further highlights the importance of sampling to greater depth in evaluating C storage potential of afforestation. Differences between our result and those of similar studies mentioned (Table 2.2) may have a number of explanations. The lower SOC content beneath trees below the arable plough layer (>30 cm) found in this study potentially implies a SOC redistribution effect, in which lack of tillage and soil turnover beneath woodland accounts for significantly higher C stock in topsoil, but not subsoil. As found by Sun et al. (2011), ceasing tillage generates significant SOC increase in the topsoil but not the whole soil profile. Sampling across the whole soil profile reveals the extent to which extra C has been fixed by woodland, or whether it is simply no longer being redistributed as a result of normal arable management.

Contributions of living roots to below-ground OC stocks, which we did not estimate, would be valuable further work, may also account for discrepancies in sub-soil C stocks. Upson et al. (2016) noted the importance of herbaceous vegetation contributions to overall below-ground C beneath new farm woodland, and Drexler et al. (2024) showed in a study on hedgerows that 85% of below ground biomass C is stored in coarse roots (> 2 mm), which we did not quantify in our study. New tree roots may also lead to a priming effect, in which exudates and below ground biomass from tree roots increase decomposition of recalcitrant SOC by fungal and microbial communities in the short term (Fontaine et al., 2007), and this may explain lower SOC concentration and stocks in woodland subsoil when compared with arable soil.

Bulk density was also controlled by plot variability in the deepest (40-50 cm) soil layer. The four plots used in our study (along with arable control areas) are sufficiently close (< 200m) to minimise covariates such as soil changes or topography, and have the same age and layout of woodland. It is therefore likely that with increasing depth, soil functioning is most strongly driven by more complex pre-existing plot conditions, such that land-cover changes including afforestation have an increasingly limited effect on functioning. Proximity to limestone bedrock may also control functioning at depth. This highlights the complexity of determining the influence of a single land cover change on soil functioning throughout the whole soil profile.

2.5. Conclusions

Thirty five years after the conversion of arable land to woodland, functioning in each of three soil horizons – forest floor material, topsoil (0-30 cm) and subsoil (>30 cm) – has its own signature in terms of ecosystem service provision. Tree species controlled forest floor nutrient dynamics and C storage through mediation of chemical content of litter material. In topsoil below the forest floor, species influence was weaker, and functioning was most strongly controlled by land cover change to woodland, with +15% or 11.8 t ha^{-1} extra C stored compared with arable land. Yet, afforestation was only capable of promoting C storage in forest floor material and topsoil. Extra C storage higher up the soil profile was cancelled out by C deficit below 30 cm, such that there was no significant net difference in C storage between soil under woodland or arable crops. Instead, C storage, bulk density and hydrological properties in subsoil were most strongly determined by small scale random variability in soil properties (pre-planting) between experimental replicates.

These findings firstly demonstrate the complexity of determining ecosystem service delivery from land use change. Considering options in linear terms using single inputs (e.g. woodland, agroforestry) and outputs (e.g. C storage) is insufficient for policy evaluation of carbon inventories and future landscape sustainability. Further consideration of C dynamics between forest floor material and soil, and other interactions between woodland C pools is required to predict the potential long-term ecosystem benefits.

Secondly, variation of benefit delivery within the soil profile demonstrates the need to evaluate findings across the whole soil column. Current schemes paying land managers per hectare for land use changes, such as agroforestry, must both consider whole soil column functioning, and be more rigorously evaluated in terms of interdependencies between factors such as tree species choice, and where beneficial outcomes are delivered. Other inputs such as soil type or management, which would usefully be the subject of further work, are also likely to have a substantial influence.

Thirdly, at greater depth in the soil column, we find benefits to be highly influenced by pre-existing spatial variability in soil properties, even over short (<200 m) distances and with the same soil type and planting design. We note the limitations this implies for tradeable ecosystem 'credit' schemes such as soil C codes, which homogenise benefits over much larger distances and variation in soil or topography.

Finally, soil outcomes from afforestation may trade-off against one another. Although woodland in this study did not appear to increase net soil C storage to 50 cm over 35 years, it was capable of promoting surface soil nutrient addition in the vicinity of trees (in the absence of synthetic inputs) within a shorter, multi-decadal timeframe, and was responsible for transferring significant organic matter to surface soil, lowering bulk density with implications for water storage and flood resilience. Assessment of benefit therefore depends on stakeholder objectives for planting. Notably we did not account for above-ground biomass C stock in woodland totals, which would likely increase C storage benefit.

Future support for farm woodland creation must be context-specific in order for land to deliver both economic and ecological benefits. Moreover, rewards for good practice must respect the diverse range of goals for planting, and how these are borne out in different parts of the soil profile. Combining these objectives will increase the potential for woodland to promote sustainable change in the landscape.

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Chapter 3. Alley width and slope position influence soil carbon storage, nutrient dynamics and hydrology at a mature silvoarable site, SW England

Abstract

Optimising benefits from agroforestry requires better understanding of spatial factors such as alley width and slope position. We sampled soil (0-50 cm) from a mature organic silvoarable site in SW England with tree rows at 12 and 24 m spacing to determine the impact of these factors on soil physical properties, carbon (C) storage and fertility. We consider how functioning differs in cropped alley and tree-row components, and how alley width influences trade-offs in ecosystem benefits. Benefits from rows extended into alleys which were 8.8% less compacted and contained 70% more available P than an adjacent, treeless control. Competition for nutrients and moisture was observed at the row-alley boundary, with lower subsoil concentrations attributable to tree root uptake. Agroforestry mitigated soil erosion despite being parallel to slope: in the control area 0.8% more soil organic matter and a 3.5% higher clay fraction was observed downslope than upslope, with no equivalent effect under agroforestry. Fertility traded off with alley width, with more N and P stored in 12 m alleys. Soil and tree-biomass C differences (700 kg C ha^{-1} $year^{-1}$) compared with the control were only significant in the 12 m system (110 stems ha^{-1}) and three times lower than estimated silvoarable contributions to future UK C budgets. Moreover, planting at lower densities (~50 stems ha^{-1}) is likely due to constraints of modern farm machinery. Assessment of silvoarable contributions to temperate ecosystem service provision must therefore consider additional benefits beyond C sequestration if agroforestry is to contribute to future landscape resilience.

3.1. Introduction

Agroforestry is promoted as a solution capable of reconciling food production and ecosystem service provision from farmland (Burgess, 1999; Araujo et al., 2012; Torralba et al., 2016; Judson et al., 2023). Arable soils are among the most degraded worldwide in terms of soil organic carbon (SOC) loss (Guo and Gifford, 2002; Wei et al., 2014), biodiversity loss (Robinson and Sutherland, 2002; Banerjee et al., 2024) and structural deterioration (Greenland, 1977), and schemes such as agroforestry which aim to attenuate these losses from arable land are critical for future climate resilience (IPCC, 2013; CCC, 2020; CCC, 2025). By incorporating a perennial, woody component, silvoarable agroforestry increases the spatial heterogeneity of conventional annual cropping systems, with potential concomitant benefits for soil functioning, food production and wider ecosystem service delivery from farmland. Demonstrated soil benefits from agroforestry include C sequestration (De Stefano and Jacobson, 2018; Mayer et al., 2022) improved nutrient cycling (Oelbermann and Voroney, 2007), altered hydrological functioning (Marshall et al., 2014; Monger et al., 2022b) and biodiversity improvements (Varah et al., 2013).

However, scaling up agroforestry in temperate areas has been restricted by limited information for practitioners on benefits and trade-offs specific to context, weak policy incentives and high capital cost (Sollen-Norrlin et al., 2020). In the United Kingdom, only 3.3% of utilised agricultural land area is currently under agroforestry, less than half of the average value for Europe (den Herder et al., 2017). Recent policy has sought to address this with Sustainable Farming Incentive (SFI) payments supporting both establishment and management of low-density agroforestry (AGF1 and AGF2) (Defra, 2024), support for denser agroforestry systems (CAGF1, CAGF3) under the Countryside Stewardship scheme (Defra, 2025) and a target of 10% agroforestry cover on arable land by 2050 in order to promote an extra 2.2 t ha^{-1} year^{-1} C storage (8 t CO_2e ha^{-1} year^{-1}) from silvoarable areas over 30 years (Woodland Trust, 2022). From both a policy and practitioner perspective, better understanding is needed on how ecosystem benefits are transferred laterally from rows of trees into cropped inter-row or ‘alley’ areas. Valuable recent work has focussed on determining how soil function indicators vary with distance from unmanaged tree rows (e.g. Cardinael et al., 2015a; Mettauer et al., 2022; Vaupel et al., 2023). Yet significant knowledge gaps still exist related to system design for various practitioner contexts, interactions and trade-offs between ecosystem benefits and how these relate to specific farm objectives for planting.

The aim of this study is to consider how the spatial distribution of soil benefits in a silvoarable system is controlled by hillslope position and alley width, and the implications of these controls for ecosystem benefit delivery from agroforestry. Dealing with hillslope layout and choosing alley width are important considerations for adopters of temperate agroforestry, yet there is limited research on their effects on soil because dedicated, replicated experimental sites isolating them are very rare and little data exists to inform practitioner design choices. We use a working horticulture farm in Devon, SW England with mature agroforestry trees conducive to study these factors to begin the process of understanding the impacts of alley width and hillslope position on soil functions.

We compare two adjacent silvoarable plots with differing alley widths alongside a third, also adjacent, treeless control plot to address three research questions. The first asks how land-use change from arable to silvoarable influences soil function indicators – bulk density, saturated hydraulic conductivity, SOC and nutrient content – in tree and alley components of agroforestry systems compared with an identically-managed treeless control area. Secondly, we consider how soil functioning varies spatially across alley and row components of agroforestry, and with hillslope position. Thirdly, we consider how selection of silvoarable alley width influences trade-offs in benefit delivery, including whether agroforestry at these planting densities can contribute to nationally-determined targets for carbon (C) storage in soil and tree biomass.

3.2. Methodology

3.2.1. Study site

The study site is a certified organic arable and horticulture farm 5 km south of Exeter, southwest England. Mean annual precipitation is 829 mm and mean annual temperature is 10.9°C (1991-2020, Exeter Airport, 11 km away) (Met Office, 2020). The soils of the study site are Eutric Chromic Endoleptic Cambisols of the Crediton Series: red, well-drained, very stony, loamy brown earths found on Permo-Triassic breccias and conglomerates predominantly in Devon (Cranfield University, 2022). Soil depths can exceed 1 m, and bedrock was not reached at the soil depths surveyed in this study (0-50 cm). The land is well-draining and easily worked in most conditions, with strong horticultural tradition in addition to mixed arable and grassland use. Agroforestry was first incorporated at the study site in 2002, the same year in which the farm was certified organic. This study focusses on this earlier area with 12 m wide alleys (hereafter referred to as 'AF_{12}'), and the second area planted in 2012 with 24 m wide alleys (hereafter referred to as'AF_{24}'), in addition to an adjacent treeless control area (hereafter referred to as 'Co') which has been managed in the same way as the alleys. Prior to establishment of agroforestry, all areas were in continuous arable cultivation.

Silvoarable agroforestry was first established using 12 m-spaced alleys aligned parallel to slope over an area of approximately 0.8 ha (AF_{12}), with a whole-system (row plus alley) tree density of 110 stems ha^{-1}. This field is on a south-south-east-oriented slope with a mean slope angle of 6.6° (11.5%). A mixed assortment of 13 cultivars of apple tree (*Malus domestica*) were planted using a semi-vigorous MM111 rootstock between December 2002 and January 2003: Discovery, Egremont Russet, D'Arcy Spice, Sturmer Pippin, Herrings Pippin, Grenadier, Blenheim Orange, Sunset, Newton Wonder, English Codlin, Adams Pearmain, Golden Noble and James Grieve. Trees were planted parallel to slope in single rows, with 3 m spacing between trees (Fig. 3.1). Beneath the trees were 3 m uncultivated strips, with 9 m cultivated alleys between each tree row. Before planting, the soil was tilled with a rotovator to 10 cm depth in a strip, and mulched using a 1 m wide permeable, woven polypropylene MyPex weed membrane (Don & Low, UK) which deteriorated following planting. Soil in the tree rows has not been tilled since, with spontaneous annual vegetation growth periodically cut with a strimmer to reduce bramble pressure in the understorey.

In 2012, a second area of agroforestry was established approximately 400 m to the west of the first area, with the same tree-row orientation and on a similar mean slope of 5.8° (10.2%), but using 24 m-spaced alleys over an area of approximately 2.5 ha (AF_{24}), resulting in a whole-system tree density of 55 stems ha^{-1}. A mixed assortment of apple cultivars was planted using a moderate MM106 rootstock between December 2012 and January 2013, this time consisting of D'Arcy Spice, Pixie, Winston, Sturmer Pippin, Egremont Russet, Jupiter, Claygate Permain and Sunset varieties. Trees were planted in 3 m wide rows, with 21 m cultivated alleys between uncultivated tree rows. As at the

AF_{12} site, ground was prepared with tillage to 10 cm depth, but was mulched using a mixed wool and plastic sheet membrane. Additionally, an area of 1 m radius surrounding each tree was composted using green manure derived from on-farm rotational grass-clover leys (see below), at planting and again in 2021. Areas beneath trees were once again left uncultivated and periodically strimmed to remove brambles.

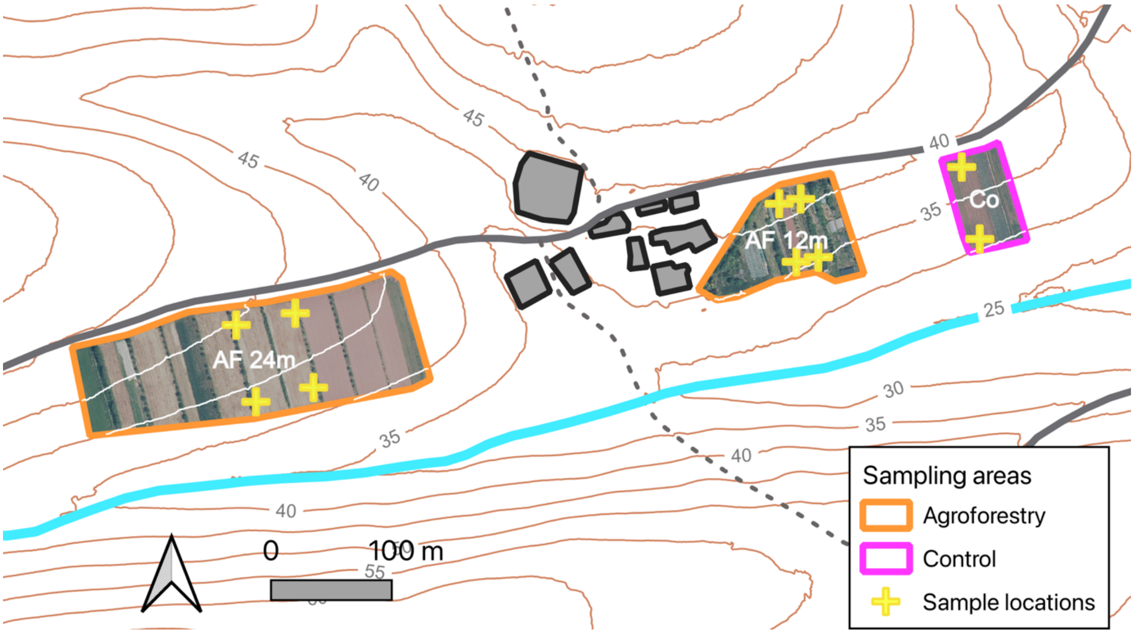

Figure 3.1 – Map of study site illustrating agroforestry areas ('AF 24m' – 24 m agroforestry alleys, 'AF 12m' – 12 m agroforestry alleys) and control area ('Co'). Built areas shown in grey. Topographic contours shown at 5 m intervals. Yellow crosses indicate the locations of the sampling layout shown in Figure 2 (four sample positions per yellow cross). Aerial photography © Getmapping Ltd., accessible at EDINA Aerial Digimap Service (2022).

A third, treeless control field (Co, Fig. 3.1) was sampled 150 m east of AF_{12}. This field shares the same soil type, aspect and similar mean slope angle (6.4°, 11.2%) to the two adjacent agroforestry areas, and has been managed in the same way as the agroforestry alleys in AF_{12} and AF_{24}. The agroforestry systems differ in tree age, some small differences in rotation are encountered and there are some minor variability in slope between the three fields. However, with sufficient replication the effect of random, uncontrolled variables is minimised (e.g. Moore et al., 1998), and even carefully-designed experimental sites can produce significant random effects on soil properties between replicates (Judson et al., 2024). The issue of tree age is only encountered when comparing the two agroforestry systems, and we control for this by comparing the magnitude of differences in soil properties with the size of the tree age difference.

Between 1983 and 2001 all three study areas were in the same continuous conventional arable rotation of wheat (*Triticum aestivum*) and barley (*Hordeum vulgare*). Since 2002, following organic conversion and initial tree planting, a six-year rotation has been in operation in the cultivated areas of all three fields as follows: brassicas (year 1), alliums (year 2), *Umbelliferae* (year 3) and legumes (year 4), and finally two years (years 5 and 6) in which cultivated areas are left in a fertility-building grass-clover ley. The only exception is for the recent period for AF_{12}, in which one of the sampled alleys had been planted with strawberries since 2021, and sample locations were sited to avoid interference with this new crop. A multispecies seed mix is used for the two-year ley, sowed at 55 kg ha^{-1} and consisting of 45% common vetch (*Vicia sativa*), 9% red clover (*Trifolium pratense*) 5% crimson clover (*Trifolium incarnatum*) and 41% Italian ryegrass (*Lolium multiforum*). Additionally, fields are sowed with 67.5 kg ha^{-1} ryegrass/vetch cover crop mix over winter, consisting of 70% common vetch (*Vicia sativa*) and 30% westerwold ryegrass (*Lolium westerwoldicum*). Cultivated areas are fertilised using green manure obtained from ley areas which is ploughed into cultivated areas before sowing and following harvesting of the main crop. This is supplemented with municipal compost from the local authority. All three treatments (AF_{12}, AF_{24}, Co) are tilled parallel to tree rows (and slope) to help weed suppression and nutrient mineralisation.

3.2.2. Experimental design

All soil samples were collected in a single week in May 2023, when study areas were predominantly in the ley phase of the rotation. Space-for-time substitution was used (e.g. Cardinael et al., 2015a; Biffi et al., 2022), in which agroforestry and control areas are assumed to have been equivalent before tree planting. Co is thus assumed to represent the baseline (t = 0) state for both of the agroforestry areas before the planting of trees. This is a reasonable assumption given the similarity in aspect and equivalent management of the three sampled areas before introduction of agroforestry.

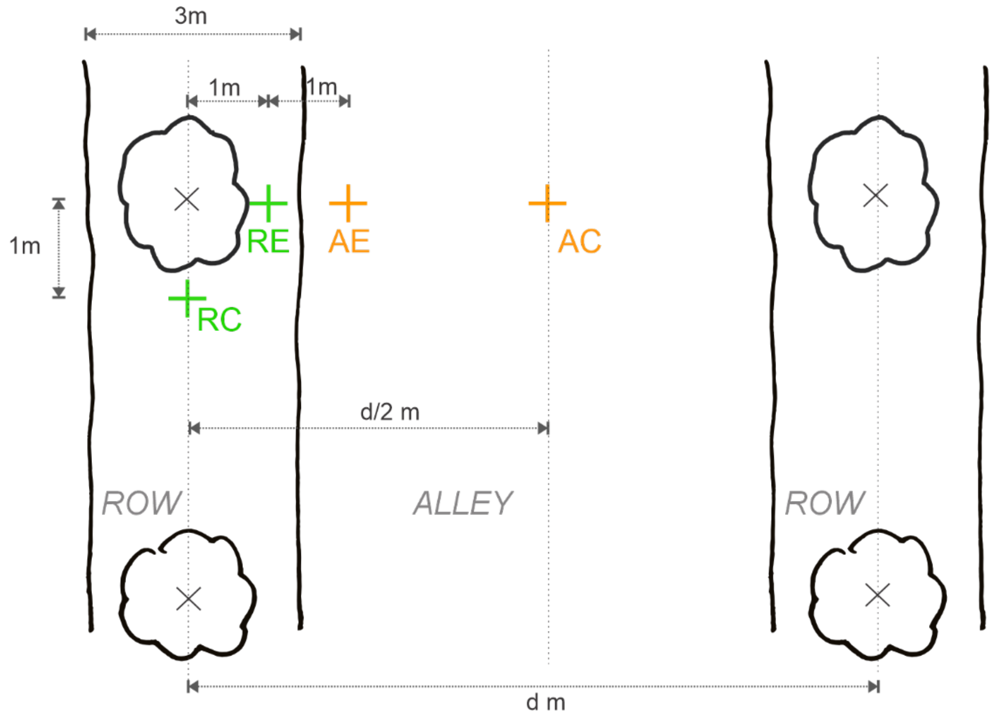

Figure 3.2 – Diagram of sampling layout within agroforestry alleys, where *d* represents alley width. Sample positions RC (row centre), RE (row edge), AE (alley edge) and AC (alley centre) are shown.

At each location (Fig. 3.1, yellow crosses), four soil samples were collected for each measurement depth at the positions shown in Fig. 3.2: row centre (RC) parallel to tree row at 1 m from tree, row edge (RE) perpendicular to tree row and at 1 m from tree (0.5 m distance to row edge), alley edge (AE) perpendicular to tree row and 2 m from tree (0.5 m to row edge) and alley centre (AC) at the midpoint of the alley. Each location (instance of Fig. 3.2) was replicated twice (on different rows and on opposite sides of the tree row) at both the top and bottom of the slope within each agroforestry treatment (Fig. 3.1), resulting in 16 sample locations each for the two agroforestry treatments. Locations were chosen within fields such that hillslope was as similar as possible between treatments while remaining within the same rotation. The chosen locations and sample positions minimise common biases associated with agroforestry transect sampling in terms of orientation, alley and tree row positions and sampling depth (Minarsch et al., 2024). An intermediate (d/4) alley data point would have been desirable but was not feasible within resource constraints of the study. These four positions were chosen to balance area coverage and replication with constraints on time and resources for sample processing. The control field was sampled using the same layout as the four agroforestry sample positions, albeit repeated just once each at the top and bottom of the slope to produce a total of eight control sample positions.

In addition to soil samples, tension infiltrometers were used to determine surface-level saturated hydraulic conductivity at each of the four sample positions (RC, RE, AE, AC). Three infiltrometers were placed on a levelled soil surface at each of the four positions at the same distance-to-tree, following removal of surface vegetation and avoiding stones. A constant tension of -0.1 cm was used, and infiltration estimates were combined with constants for loam soil (particle size distribution analysis indicated loam soil at all sampling locations, see Section 2.3) derived from the method of van Genuchten (van Genuchten and Nielsen, 1985) to determine near-surface saturated hydraulic conductivity (K_s).

Finally, tree height and diameter at breast-height (1.3 m, DBH) were measured from two tree rows in each of the two agroforestry treatments, with five trees selected at random from each row. Height was measured using a rule placed up to the full tree height, and DBH with a measuring tape placed around the circumference of the trunk and pre-marked with corresponding diameter values. These measurements were combined with allometric equations from the Forestry Commission Woodland Carbon Code: Carbon Assessment Protocol (Jenkins et al., 2018) to determine estimates of above- (AGB) and below-ground biomass (BGB) C stock for the trees.

3.2.3. Soil sampling and analysis

At each sample position soil samples were collected at three depth intervals: 0-10 cm, 10-20 cm and 20-50 cm. For the uppermost two samples, a 5 cm diameter, 100 cm^3 ring corer (Eijkelkamp, Holland) was used to extract intact soil cores at 2.5-7.5 and 12.5-17.5 cm depth, representing 0-10 and 10-20 cm intervals, respectively. For deeper soil samples, a 5 cm diameter, 600 cm^3 liner sampling soil corer (Eijkelkamp, Holland) was used to extract an intact 30 cm soil core representing 20-50 cm soil depth. On return to the laboratory, all soil samples were weighed and then oven-dried at 105°C for 24 hours (48 hours for the larger 20-50 cm samples) before being re-weighed for determination of bulk density. Moisture content was determined for each sample by comparing soil mass before and after drying at 105°C. Roots and stones were extracted with a 2 mm sieve and weighed to correct for their presence in the soil. Bulk density was calculated by subtracting the root and stone fraction from the initial mass and dividing by the volume of the corer (Poeplau et al., 2017). Sub-samples of the oven-dry soil were heated to 550°C for 12 hours in order to determine soil organic matter (SOM) content

(%) using the loss-on-ignition (LOI) method. These samples were subsequently weighed with LOI determined as the difference in mass between 105°C (oven-drying) and 550°C (ignition) samples, divided by the oven-dry mass and multiplied by 100 to give a percentage value.

Separate, loose soil samples from the same sample positions and depth intervals were collected for determination of SOC, nitrogen (N) and phosphorus (P) concentrations. On return to the laboratory, the field moist soil was passed through a 5 mm sieve as soon as possible to homogenise samples. For determination of plant available N, approximately 10 g field-moist sample was combined with 50 mL 1M KCl solution and shaken for 1 hour at 150 cycles min^{-1} using a shaker table. These samples were subsequently passed through Whatman 42 filter paper into centrifuge tubes, with nitrite (NO_2-N), nitrate (NO_3-N) and ammonium (NH_4-N) content determined using a Skalar San ++ (Skalar Analytical B.V., Netherlands) continuous flow auto-analyser. The remaining field-moist sample was dried at room temperature and passed through a 2 mm sieve, weighed and subsequently ground to <150 μm using a Retsch MM400 ball mill (RETSCH GmbH, Germany) for determination of soil C and N content. For analysis of total C and N approximately 4 mg of <150 μm sample was weighed using a six-figure balance into tin capsules, crushed into a small cube to remove air from the sample, and subsequently introduced into an Elemental Vario EL cube (Elementar Analysensysteme GmbH, Germany) combustion analyser to determine concentrations of each element. For SOC a similar procedure was adopted, with the exception that 30 μL of 15% HCl was added to each sample to remove carbonates. The samples were left to react and settle for 24 h and oven dried for 2 h at 80°C before being analysed for C content using an Elemental Vario EL cube (Elementar Analysensysteme GmbH, Germany). Olsen's P (Olsen, 1954) was determined using the <2 mm fraction of loose soil samples. Approximately 2.5 g air-dried soil was weighed into a shaker bottle, combined with 50 mL of 0.5M $NaHCO_3$ solution and mixed for 30 minutes at 150 cycles min^{-1} using a shaker table. These samples were subsequently passed through Whatman 42 filter paper into centrifuge tubes, with Olsen P (PO_4-P) content determined using a Skalar San ++ (Skalar Analytical B.V., Netherlands) continuous flow auto-analyser.

At each sample position, soil particle size distribution was determined on the surface sample only (2.5-7.5 cm, representing 0-10 cm) by gravimetry using the <2 mm fraction of loose soil. Approximately 10 g air-dry soil was weighed and combined with 100 mL deionised water followed by 20 mL dispersing agent (5% m/v sodium hexametaphosphate), with the mixture left to stand for 12 hours. The mixture was then agitated for 10 minutes before being passed through a 53 µm sieve, with the sand fraction left on the sieve removed and dried for 48 hours at 105°C before being weighed. The mixture that passed through the sieve was then left in a 1 L sedimentation cylinder for a quantity of time determined by ambient temperature (see Black *et al.*, 1983) (1 h 57 min for 21°C), after which 20 mL sample was extracted 5 cm below the liquid surface, dried at 105°C and weighed to determine clay content (< 2 µm). Silt content (2-53 µm) was determined by subtraction. Measured soil particle size distributions for each site were used to inform estimates of saturated hydraulic conductivity (see Section 2.2).

Soil pH was measured using the <2 mm fraction of air-dry soil. Approximately 20 g air-dry soil was combined with 40 mL deionised water and stirred for 15 minutes. The pH was measured in the deionised water only, before being measured again after the addition of 250 µL $CaCl_2$.

3.2.4. Mineral soil mass corrections

All ratio-based soil measurements were normalised according to a reference quantity of mineral soil in order to correct for land-use change effects on bulk density and SOM. For this purpose, the aggregated Co samples were used, as they are assumed to represent $t = 0$ under space-for-time substitution. Mineral soil was calculated as the mass of dry soil per unit area in the aggregated control area samples to three reference depths, corresponding to sample depths used in the study (0-10 cm, 0-20 cm and 0-50 cm). For a given soil property at the agroforestry locations (e.g., moisture, SOC), its cumulative mass was calculated to the same reference depths, and plotted against cumulative mineral soil mass. A cubic spline function (von Haden et al., 2020) was considered for interpolating these data, however this can produce significant artefacts (including negative values) if used to infer adjusted values outside the interpolated region. An exponential function of the form $y = A(1 - \exp(-Bx))$ was therefore used to fit data, from which corrected (or equivalent soil mass - ESM) values were interpolated using cumulative mineral soil values from the reference (control) area. Where cumulative mass data did not plateau with depth such that an exponential fit could not be used, a linear fit of the form $y = \mathrm{A}x$ was used in its place. Fit data for all variables is illustrated in the Supporting Information.

3.2.5. Calculation of whole-system C, N and P stock

Contributions of tree row and alley to whole system C, N and P stock must be area-weighted in order to avoid overestimation when upscaling results (Minarsch et al., 2024). For each of the two systems (AF_{12}, AF_{24}), soil C, N and P stocks were calculated by multiplying the mean stock for row (RC, RE) and alley (AE, AC) components according to the fractional land area each comprised in the 12 m and 24 m systems. Specifically, stock contributions of the tree row were multiplied by 25% and 12.5% for the AF_{12} and AF_{24} systems, respectively, with contributions from the alley multiplied by 75% and 87.5%, respectively, before being summed to calculate total stock in each system. The row edge (RE) and alley edge (AE) soil C stock values were taken to represent the stock values immediately on either side of the row-alley boundary, with the distribution of stock values between the edge and centre of each component assumed to be linear. In the case of C stock, estimates of above- (AGB) and below-ground tree biomass (BGB) (Section 3.2.2) were included in tree row contributions and area-weighted according to stem density.

3.2.6. Statistical analysis

Data were analysed in three stages. Firstly, all data for row and alley areas were homogenised across the two agroforestry systems to create two treatment groups (Row and Alley), each of which could be compared with control group values for a given depth interval. This facilitates comparison of soil properties between row and alley components of agroforestry and the treeless control area.

Secondly, data within Row and Alley groups were disaggregated by position (lateral – RC, RE, AE, AC; and with hillslope – UP and DOWN) and compared with control group data (control data disaggregated by hillslope position). These groups were compared in order to determine rudimentary spatial dynamics within agroforestry in sectional view, and compare these with the control area. Spatial maps were generated using linear interpolation between sample points.

Finally, Row and Alley groups were disaggregated by agroforestry system (AF_{12}, AF_{24}), to determine differences in stocks of key soil properties by alley width and compared with the control area. Stock contributions of each of the row and alley components were area-weighted as described in Section 3.2.5. Similarly, estimates of tree biomass C were weighted according to whole-system stem density.

Data used for determination of all soil indicator values were tested for normality (Shapiro-Wilk) and homogeneity of variance (Bartlett) in order to meet assumptions for ANOVA and pairwise Tukey tests. Where these assumptions were not met, a non-parametric Kruskal-Wallis test was used in place of ANOVA, followed by pairwise Dunn's tests with a Bonferroni correction. Effect sizes between groups were determined using Cohen's *d* value . Comparison of whole system C, N and P stocks was undertaken using independent two-sample *t*-tests of the combined contributions of agroforestry area components to total stock values. A mixed effect modelling approach incorporating fixed and random factors was considered, however this approach requires more sophisticated experimental design with block replication and paired data points between groups. Moreover, the purpose of this study is not to compare relative effect strengths between factors as mixed effect models can, but simply to detect the presence of significant differences between treatments. ANOVA is known to be robust when comparing unbalanced groups provided assumptions of normality and homogeneity of variance are met (Sawyer, 2013), and these were tested throughout. All tests were undertaken using *SciPy* (Virtanen et al., 2020) and *statsmodels* (Seabold and Perktold, 2010) within the Python environment (v. 3.10). Generation of spatial variability plots for Section 3.3.2 was undertaken using two-dimensional, linear interpolation between sample points using *RegularGridInterpolator*, also within Python's *SciPy* library (Virtanen et al., 2020).

3.3. Results

3.3.1. Differences in soil properties between tree row, alley and treeless control areas

3.3.1.1. Soil physical properties

Agroforestry did not have a significant effect on bulk density in topsoil (0-20 cm) compared with Co (all figures shown as mean ± standard error; tables of mean and standard error values for all indicators are included in the Supporting Information) (AF: 0.98 ± 0.02 g cm^{-3}, n = 32; Co: 1.02 ± 0.02 g cm^{-3}, n = 8; d = 0.45, p = 0.262) (Fig. 3.3, Supp. Table B.1). However, at 20-50 cm depth, bulk density of soil in agroforestry areas (1.07 ± 0.02 g cm^{-3}) was significantly lower than in Co (1.20 ± 0.03 g cm^{-3}, d = 1.12, p = 0.007). Thus, over the whole measured soil profile (0-50 cm), bulk density was significantly lower overall in agroforestry areas (1.04 ± 0.02 g cm^{-3}) compared with Co (1.13 ± 0.02 g cm^{-3}, d = 1.19, p = 0.005) (Fig. 3.2b). Bulk density of the row and alley components of agroforestry was not significantly different for either topsoil (0-20 cm, d = 0.37, p = 0.299) or subsoil (20-50 cm, d = 0.17, p = 0.633), nor over the whole soil column (0-50 cm, d = 0.02, p = 0.962).

Significant difference in topsoil SOM content was found between agroforestry (7.01 ± 0.13 %) and Co (6.43 ± 0.19 %) (d = 0.83, p = 0.044) (Fig. 3.3, Supp. Table B.1). However, SOM content did not differ between agroforestry (4.98 ± 0.09 %) or Co (5.07 ± 0.31 %) treatments over the whole measured soil profile (0-50 cm) (d = 0.17, p = 0.669).

No significant difference was observed in surface K_s between the agroforestry areas (7.98 (6.04-10.54) mm hr^{-1}) and Co (3.57 (2.51-5.05) mm hr^{-1}) (d = 0.48, p = 0.089) (Supp. Table B.1). However, the tree-row component of the agroforestry areas exhibited significantly faster surface K_s (12.37 (8.64-17.70) mm hr^{-1}) than Co (d = 0.82, p = 0.029). There was no significant difference between alley surface K_s (5.04 (3.33 – 7.61) mm hr^{-1}) and Co (d = 0.21, p = 0.569).

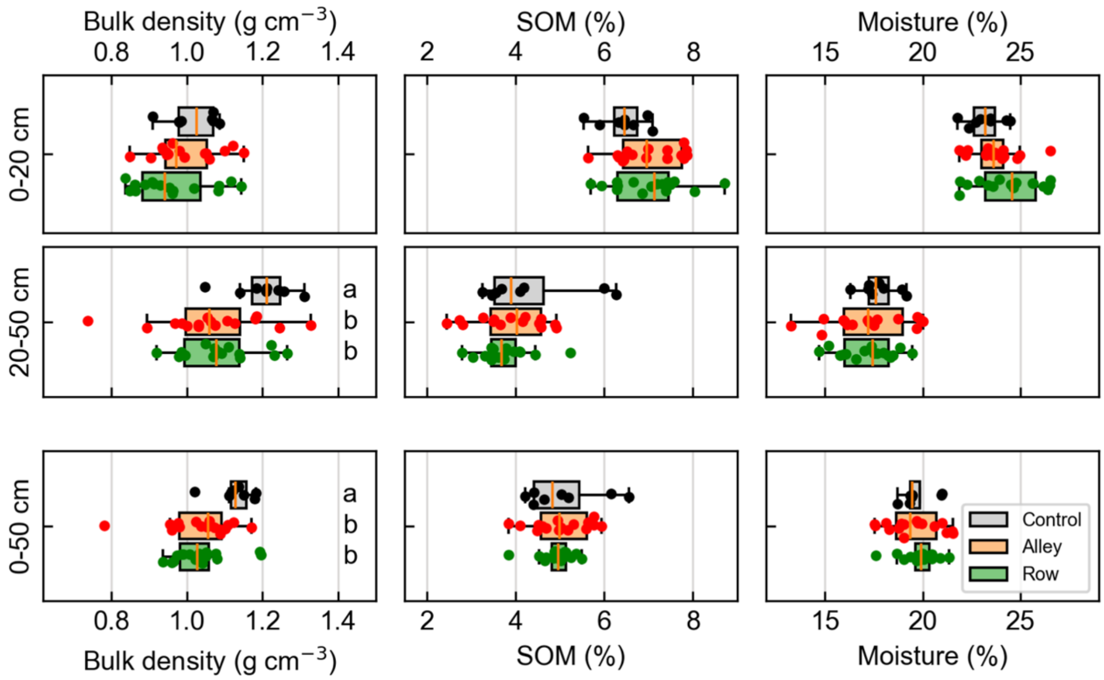

Figure 3.3 – Soil bulk density, organic matter (SOM) and moisture in Control, Alley and Row sample locations for topsoil (0-20 cm), subsoil (20-50) and whole measured soil column (0-50 cm). Mean soil property values (0-20 cm, 0-50 cm) are calculated as weighted averages, corrected for soil mass in each depth interval.

Particle size distribution varied significantly between row and alley components of agroforestry, with significantly higher clay percentage in the alley areas (18.2 ± 0.3 %) compared with the tree rows (16.5 ± 0.4 %, d = 1.10, p = 0.004) (Supp. Table 1). Tree rows (45.9 ± 0.7 %) had higher sand percentage than Co (42.6 ± 0.9 %, d = 1.20, p = 0.012), whereas rows had lower clay percentage (16.5 ± 0.4 %) than Co (18.7 ± 0.9 %, d = 1.10, p = 0.019). Particle size fractions were similar between the alley areas and Co for all three size classes.

3.3.1.2. Soil carbon

SOC concentration in topsoil (0-20 cm) was higher beneath agroforestry (2.89 ± 0.09 %) than in Co (2.33 ± 0.06 %, d = 1.21, p = 0.005, Fig. 3.4, Supp. Table B.2). However, within subsoil there was no significant difference between the agroforestry and Co areas (d = 0.23, p = 0.612). Over the whole soil profile (0-50 cm) there was significantly higher SOC content in the agroforestry (1.71 ± 0.06 %) than the Co area (1.45 ± 0.05 %, d = 0.85, p = 0.016). SOC content (0-50 cm) in the tree-rows (1.68 ± 0.05 %) was significantly higher than the control (d = 1.23, p = 0.009), but there was no difference between Co (1.45 ± 0.05 %) and alley areas (1.74 ± 0.11; d = 0.81, p = 0.111), nor between alley and tree row (d = 0.19, p = 0.651).

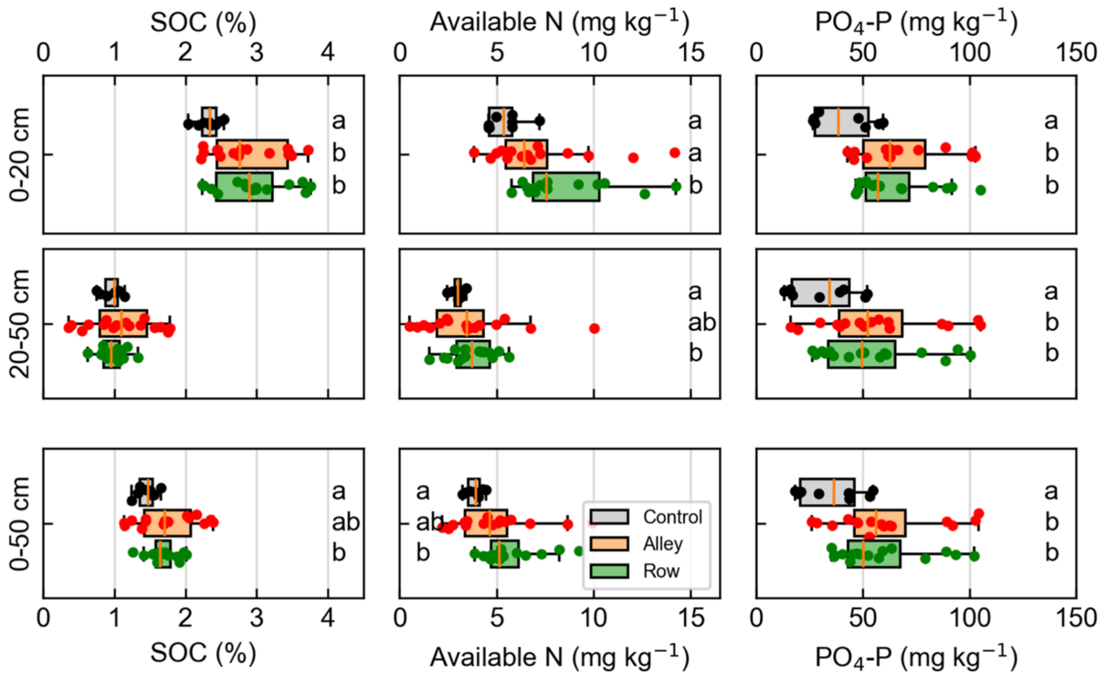

Figure 3.4 – Concentration of SOC, available N and PO4-P at Control, Alley and Row sample locations for topsoil (0-20 cm), subsoil (20-50) and whole measured soil column (0-50 cm). Mean soil property values (0-20 cm, 0-50 cm) are calculated as weighted averages, corrected for soil mass in each depth interval.

SOC stock in topsoil was greater in agroforestry (59.2 ± 1.9 t ha^{-1}) than in Co (47.3 ± 1.3 t ha^{-1}, d = 1.20, p = 0.007, Fig. 3.5), which equates to a difference of 11.9 ± 2.3 t ha^{-1} in the uppermost 20 cm between the two systems. Both tree-row (59.7 ± 2.8 t ha^{-1}, d = 1.31, p = 0.014) and alley areas (58.8 ± 2.7 t ha^{-1}, d = 1.24, p = 0.014) had significantly more topsoil C compared with Co. Over the whole soil profile, SOC stock was also significantly greater in agroforestry (96.4 ± 3.3 t ha^{-1}) than Co (82.0 ± 2.8 t ha^{-1}, d = 0.83, p = 0.018). Only tree rows contained significantly more SOC stock at 0-50 cm (94.5 ± 2.9 t ha^{-1}) compared with Co (d = 1.19, p = 0.012); in alley areas high spatial variability between locations meant there were no significant differences with Co (d = 0.79, p = 0.111).

3.3.1.3. Soil nutrients

Topsoil available N (NO_3-N + NO_2-N + NH_4-N) was higher in agroforestry areas (8.04 ± 0.55 mg kg^{-1}) than in Co (5.42 ± 0.32 mg kg^{-1}, d = 0.92, p = 0.006, Fig. 3.4, Supp. Table B.2), whereas in subsoil there was no significant difference between the treatments (AF: 3.66 ± 0.33 mg kg^{-1}, Co: 2.99 ± 0.11 mg kg^{-1}, d = 0.40, p = 0.250). Topsoil beneath tree-rows had significantly higher available N content than Co (R: 8.96 ± 0.82 mg kg^{-1}, d = 1.29, p = 0.001), whereas there was no difference in topsoil available N between the alley and Co (A: 7.13 ± 0.69 mg kg^{-1}, d = 0.73, p = 0.111). Measured as a stock to 50 cm depth, significantly more available N (+7.8 ± 2.0 kg ha^{-1}) was present beneath agroforestry areas (29.6 ± 1.9 kg ha^{-1}) compared with Co (21.8 ± 0.9 kg ha^{-1}, d = 0.82, p = 0.008, Fig. 3.5).

Available P (PO_4-P) content was significantly higher beneath agroforestry than the Co area in both topsoil (AF: 65.9 ± 3.4 mg kg^{-1}, Co: 40.8 ± 5.1 mg kg^{-1}, d = 1.36, p = 0.003, Fig. 3.4) and subsoil (AF: 55.5 ± 4.5 mg kg^{-1}, Co: 32.4 ± 5.6 mg kg^{-1}, d = 0.96, p = 0.021), and across the whole measured soil column (AF: 59.3 ± 4.0 mg kg^{-1}, Co: 35.4 ± 5.4 mg kg^{-1}, d = 1.10, p = 0.009). This equated to a PO_4-P stock difference of 134 kg ha^{-1} (+67%) between agroforestry (334 ± 23 kg ha^{-1}) and Co (200 ± 30 kg ha^{-1}) treatments to a depth of 50 cm (d = 1.10, p = 0.009, Fig. 3.5). Both alley (60.5 ± 6.2 mg kg^{-1}, d = 1.13, p = 0.016) and row (58.1 ± 5.4 mg kg^{-1}, d = 1.15, p = 0.020) components of agroforestry contained significantly higher PO_4-P content to 50 cm than Co (35.4 ± 5.4 mg kg^{-1}).

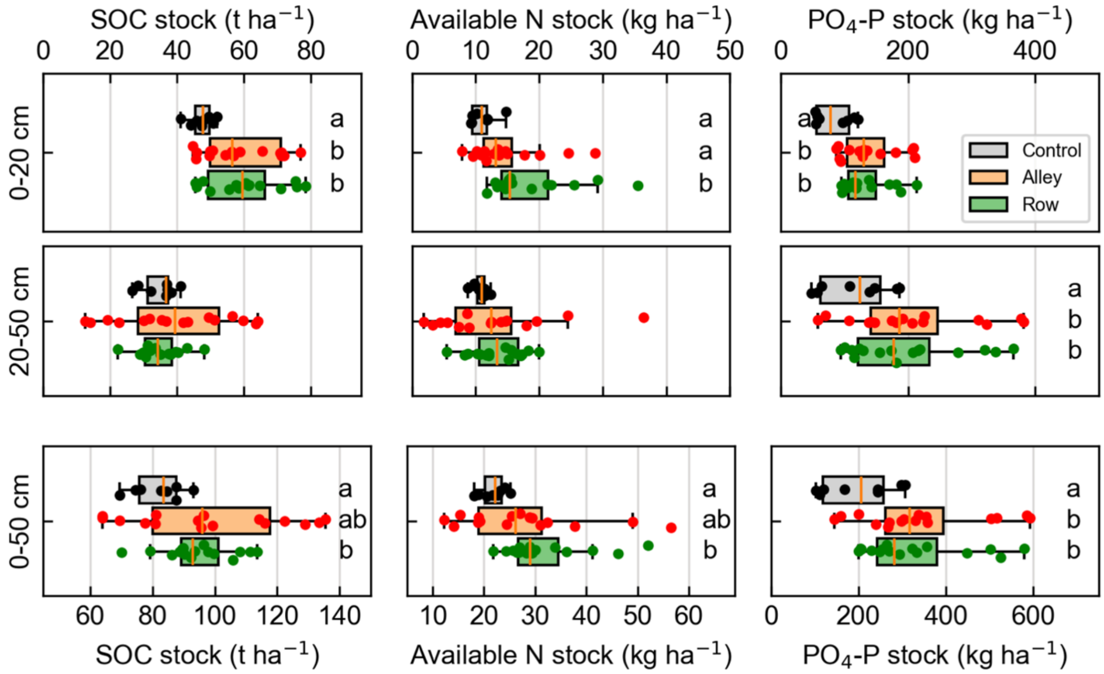

Figure 3.5 – Stock of SOC, available N and PO_4-P at Control, Alley and Row sample locations for topsoil (0-20 cm), subsoil (20-50) and whole measured soil column (0-50 cm). Mean soil property values (0-20 cm, 0-50 cm) are calculated as weighted averages, corrected for soil mass in each depth interval.

C:N (SOC:total N) was significantly higher beneath agroforestry (7.19 ± 0.24) than the Co area (5.92 ± 0.16) to 20 cm depth (d = 1.01, p = 0.008). However, there was no significant difference in C:N at 20-50 cm (d = 0.321, p = 0.437) or 0-50 cm (d = 0.60, p = 0.166) between agroforestry and Co. C:N was similar between alley (7.03 ± 0.35) and tree-row (7.35 ± 0.35) areas (d = 0.23, p = 0.521) at 0-20 cm.

3.3.2. Spatial variability in soil properties within agroforestry areas

Sample positions in this section are abbreviated as follows: row centre (RC), row edge (RE), alley edge (AE) and alley centre (AC). Figures 6 and 8 represent cross-sectional variability in indicator values with lateral position in both upslope and downslope positions, and with depth. Soil textural data (Fig. 3.7) was only measured in surface soil (0-10 cm).

3.3.2.1. Soil physical properties

Although 0-50 cm bulk density was higher in the alley centre (AC: 1.06 ± 0.01 g cm^{-3}, n = 8) than at the edge of the tree row (RE: 1.00 ± 0.01 g cm^{-3}, n = 8, d = 1.46, p = 0.011), differences in mean bulk density between lateral sample positions were not significant at $p < 0.05$ ($p = 0.169$, Fig. 3.6a). Similarly, differences in bulk density between upslope agroforestry sample positions (AF_{up}: 1.02 ± 0.02 g cm^{-3}, $n = 16$) and downslope positions (AF_{down}: 1.06 ± 0.02 g cm^{-3}, $n = 16$) were not significant ($d = 0.50$, $p = 0.166$). Bulk density was generally lower in the agroforestry plots compared with control plots ($d = 1.19$, $p = 0.005$, Fig. 3.6a), however this difference was more pronounced in downslope sample positions (AF_{down}: 1.06 ± 0.02 g cm^3, $n = 16$; Co_{down}: 1.15 ± 0.02 g cm^{-3}, $n = 4$; $d = 1.51$, $p = 0.015$) compared with upslope positions (AF_{up}: 1.02 ± 0.02 g cm^{-3}, $n = 16$; Co_{up}: 1.11 ± 0.03 g cm^{-3}, $n = 4$; $d = 1.04$, $p = 0.081$).

SOM content varied significantly in the lateral direction between alley and row sample positions ($p = 0.001$) and also between upslope and downslope positions ($p = 0.044$). SOM was highly uniform across the tree row (RC: 4.96 ± 0.12, RE: 4.88 ± 0.16, $d = 0.19$, $p = 0.916$), but varied strongly between the alley edge (AE: 4.52 ± 0.14%) and alley centre (AC: 5.51 ± 0.12%, $d = 2.65$, $p < 0.001$, Fig. 3.6b). Compared with Co, agroforestry areas had higher SOM content upslope (+0.74%, $d = 1.61$, $p = 0.010$), and lower SOM content downslope (–0.46%, $d = 1.04$, $p = 0.080$).

Moisture content was also significantly variable with lateral position ($p = 0.003$) but not with slope position ($p = 0.906$). Although not varying significantly within the tree row ($d = 0.445$, $p = 0.389$), moisture content significantly increased between alley edge and alley centre (AE: 18.6 ± 0.4%, AC: 20.5 ± 0.3%, $d = 1.90$, $p = 0.002$) (Fig. 3.6c). These differences were apparent when comparing with Co, particularly upslope. Agroforestry areas contained higher upslope moisture content across 0-50 cm depth than Co at RC (+0.74%, $d = 2.06$, $p = 0.021$) and at AC (+1.70%, $d = 2.56$, $p = 0.021$), whereas downslope moisture content was very similar between agroforestry and Co treatments. The majority of moisture content difference at the alley edge was observed in subsoil and was much less pronounced in topsoil (Fig. 3.6c).

3.3.2.2. Particle size analysis

Soil particle size distributions were only measured for surface soil samples (0-10 cm). Among particle classes, only clay content varied significantly in surface soil between lateral sample positions ($p = 0.045$), with no significant lateral variation in silt ($p = 0.865$) or sand ($p = 0.336$) content (Fig. 3.7). However, significant differences were observed within agroforestry areas between upslope and downslope positions for both silt content (AF_{up}: 36.2 ± 0.5%, AF_{down}: 39.2 ± 0.7%, $d = 1.18$, $p = 0.002$) and sand content (AF_{up}: 46.6 ± 0.6 %, AF_{down} 43.4 ± 0.7 %, d = 1.30, p = 0.001), with higher silt content downslope, and higher sand content upslope.

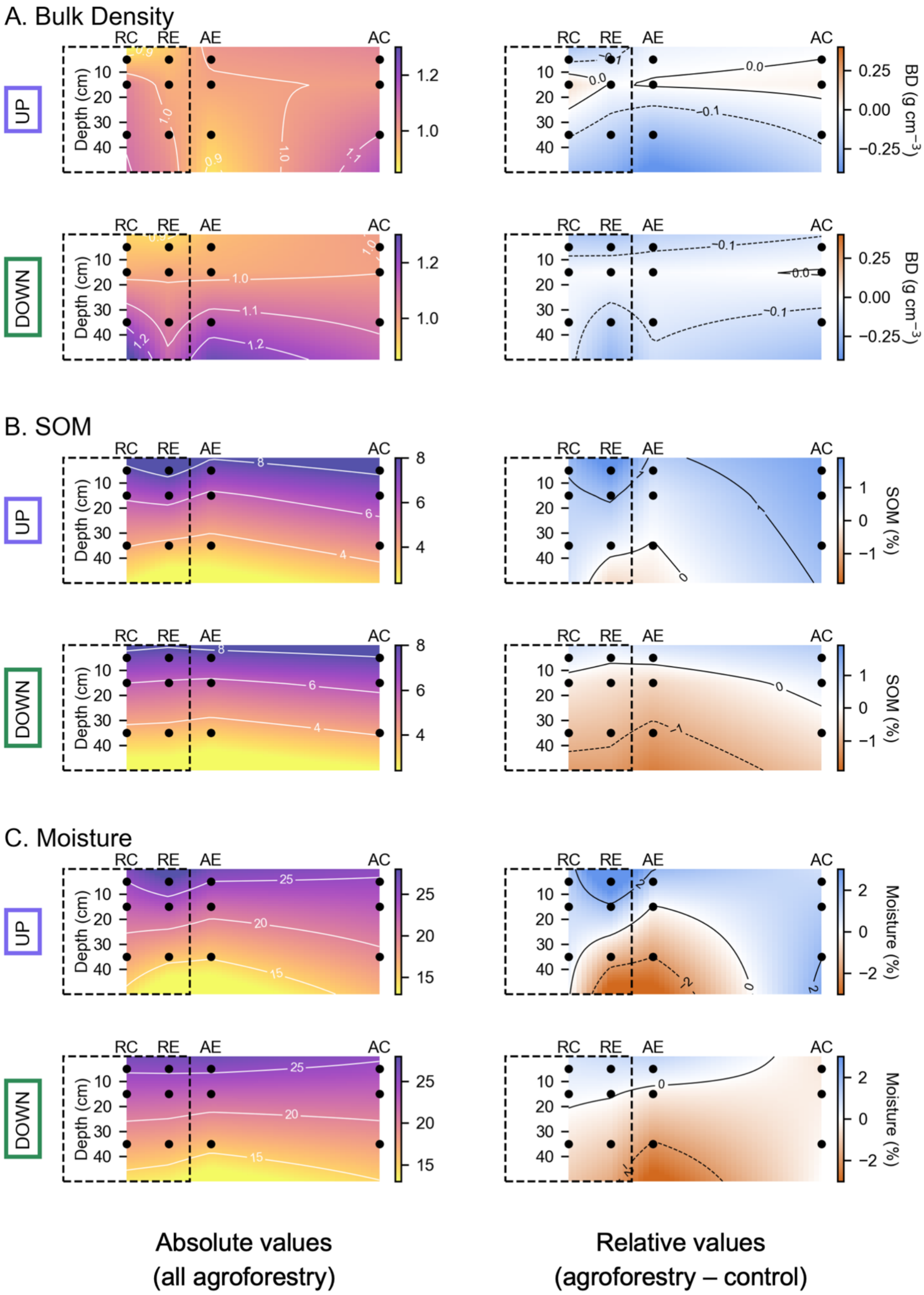

Figure 3.6 – Cross-sectional variability in a. soil bulk density, b. SOM and c. moisture in upslope (UP - purple) and downslope (DOWN - green) positions, and row centre (RC), row edge (RE), alley edge (AE) and alley centre (AC) positions. Lateral distances between points are not shown to scale. Left column of plots shows variation in absolute values within combined agroforestry areas, right column shows differences between agroforestry and control areas. Data points are shown as black dots, with $n = 4$ for each point. Contours mapped using two-dimensional linear interpolation between data points. Plots for C:N and K_s are included in the Supporting Information.

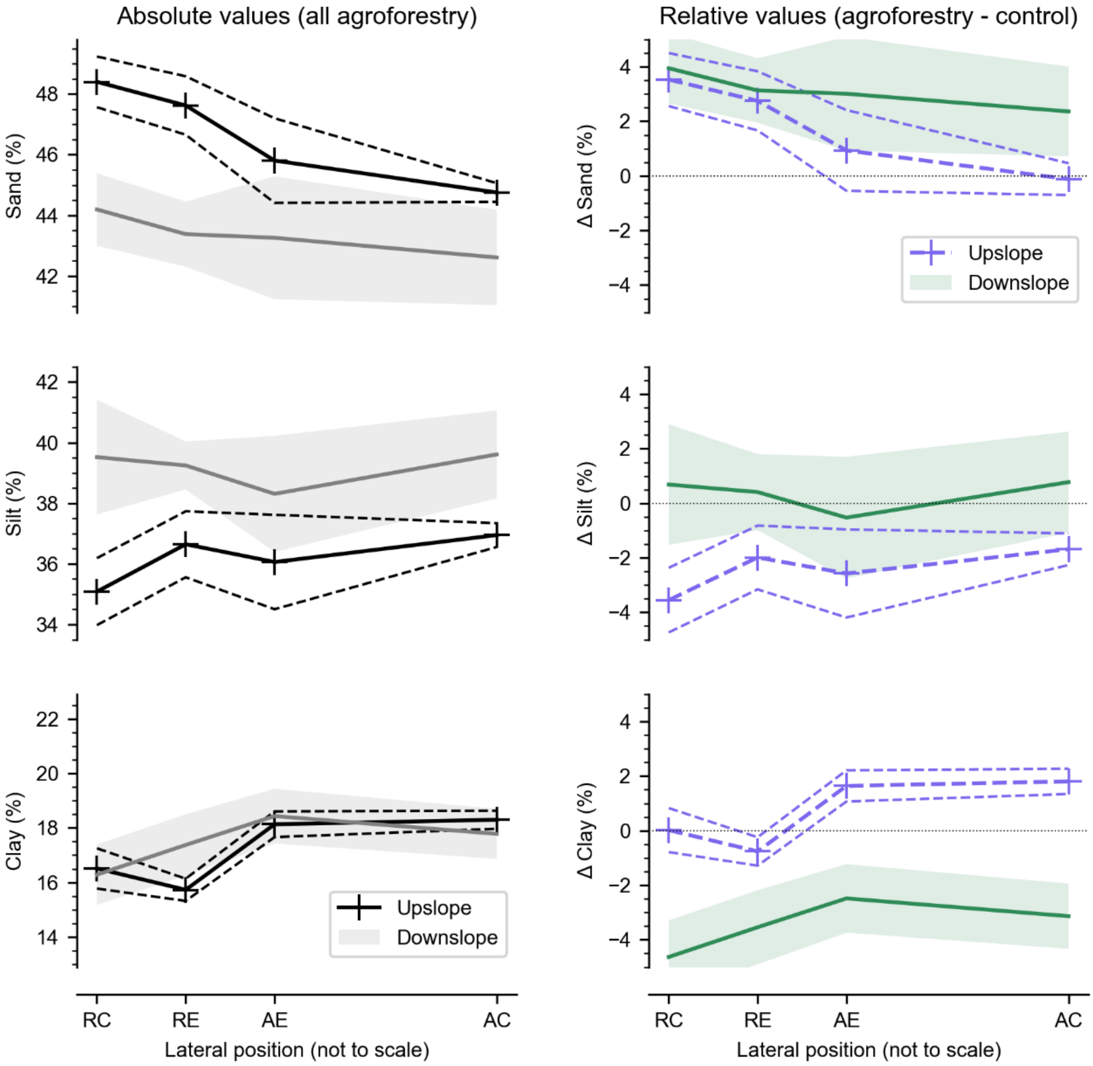

Figure 3.7 – Surface variation in soil textural classes for upslope and downslope positions, and row centre (RC), row edge (RE), alley edge (AE) and alley centre (AC) positions. Shaded/dotted areas indicate standard error bounds. Left column of plots shows variation in absolute values within combined agroforestry areas, right column shows differences between agroforestry and control areas.

In general, spatial variation in particle size fractions implied that, for agroforestry areas, silt ($p = 0.002$) and sand ($p = 0.001$) content were most sensitive to slope position, with minimal effect on clay ($p = 0.634$); whereas in the treeless control area clay ($p = 0.002$) and sand ($p = 0.001$) content were most sensitive to slope position, with minimal effect on silt ($p = 0.874$). Significantly higher clay content (+3.45 %) was observed downslope in the Co area compared with the same part of the agroforestry areas ($d = 1.77$, $p = 0.005$).

3.3.2.3. Soil carbon stock

SOC stock varied significantly with lateral position across alley and row components of agroforestry ($p < 0.001$) (Fig. 3.8a). Stocks were similar within the row (RC: 92.1 ± 3.0 t C ha, RE: 97.1 ± 5.0 t C ha^{-1}, $d = 0.43$, $p = 0.400$), but varied significantly within the alley (AE: 78.5 ± 4.4 t ha^{-1}, AC: 118.3 ± 5.3 t ha^{-1}, $d = 2.88$, $p < 0.001$), with the highest SOC stock found at the centre of the cropped alley. Variation in SOC stock between upslope and downslope agroforestry sample positions was minimal ($p = 0.243$). However, the difference in SOC stock in the sampled soil profile (0-50 cm) between agroforestry and control treatments was five times larger and significant at the top of the slope (+24.2 t C ha^{-1} in AF, $d = 1.29$, $p = 0.033$) compared with the bottom of the slope (+4.8 t C ha^{-1} in AF, $d = 0.305$, $p = 0.508$).

3.3.2.4. Available N and P stock

Available N stock over the whole measured soil column varied significantly both laterally ($p = 0.040$) and with slope position ($p = 0.019$) (Fig. 3.8b). Stock was similar in the tree row ($d = 0.79$, $p = 0.134$) but was significantly reduced at AE (21.1 ± 3.2 kg ha^{-1}) compared with RE (34.9 ± 3.4 kg ha^{-1}, $d = 1.47$, $p = 0.011$). Slope differences in available N stock between agroforestry and Co were more exaggerated in the alley than the tree row. A significant difference in N stock was only observed upslope (AF +12.2 kg ha^{-1}, $d = 1.16$, $p = 0.023$) and there was no significant difference in N stock between the treatments downslope ($d = 0.44$, $p = 0.299$). At downslope AE, N stocks were lower in agroforestry compared with Co (AF -5.1 kg ha^{-1}, $d = 2.29$, $p = 0.021$), with differences driven by lower N availability in downslope subsoil at the alley edge.

Available P (PO_4-P) stock did not vary significantly with either lateral position (*p* = 0.323) or slope (*p* = 0.407) in agroforestry areas (Fig. 3.8c). Stocks were uniformly greater in the agroforestry area compared with Co (AF +136 kg ha^{-1}, *d* = 1.10, *p* = 0.009), although the difference in available P stock between the treatments was more pronounced downslope (AF +167 kg ha^{-1}, *d* = 3.18, *p* <0.001) than upslope (AF +104 kg ha^{-1}, d = 0.68, *p* = 0.450). The difference between upslope and downslope P stocks was significant and pronounced in Co (–148 kg ha^{-1}, *d* = 4.69, *p* = 0.001) but there were no such slope position differences in the agroforestry area (*d* = 0.69, *p* = 0.407).

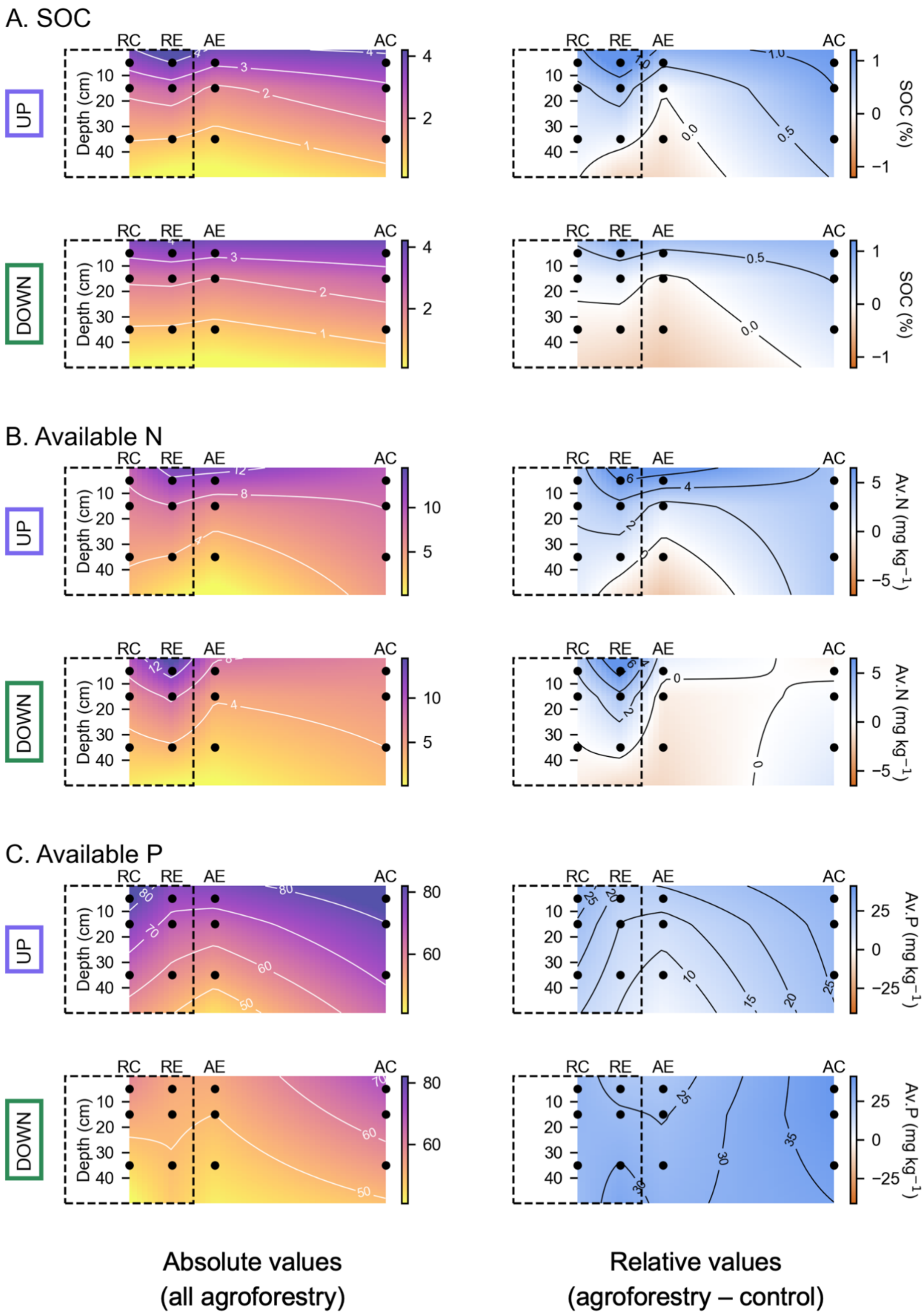

Figure 3.8 – Cross-sectional variability in a. SOC, b. available N and c. available P content in upslope (UP) and downslope (DOWN) positions, and row centre (RC), row edge (RE), alley edge (AE) and alley centre (AC) positions. See caption to Figure 6 for a full description.

3.3.3. Comparing agroforestry systems: organic C and nutrient stocks

Nutrient and SOC stocks were significantly different between the three treatments (AF_{12}, AF_{24} and Co) when adjusted for proportional area (tree row, alley). Measurement depth also controlled differences between stocks for each treatment type.

3.3.3.1. Carbon stock

When combining the AGB and BGB C stocks with the topsoil (0-20 cm) SOC, total C stocks were variable between the three treatments. The greatest total C stock was found in the AF_{24} system (Fig. 3.9a), a difference of +17.2 t C ha^{-1} compared with Co (d = 1.53, p = 0.001), with + 7.2 t C ha^{-1} in AF_{12} compared with Co (d = 1.10, p = 0.018).

For the whole measured soil column (0-50 cm), higher uncertainty meant there was no significant difference in total C stock between AF_{24} and Co (d = 0.62, p = 0.104), however AF_{12} contained a significant difference of +14.7 t C ha^{-1} compared with Co (d = 0.93, p = 0.028) (Fig. 3.9b). Estimated AGB and BGB C stocks were very small (AF_{12}: 1.02 t C ha^{-1}, AF_{24}: 0.13 t C ha^{-1}) compared with SOC stocks.

3.3.3.2. Available N stock

Available N stock in topsoil was greatest in AF_{12} with a difference of +6.4 kg ha^{-1} compared with Co (d = 1.05, p = 0.011). Available N stock in AF_{24} was intermediate and did not differ significantly from either AF_{12} (d = 0.71, p = 0.065) or Co (d = 0.70, p = 0.072). In the whole measured soil column, the ranks were the same, with +10.8 kg N ha^{-1} in AF_{12} compared with Co (d = 0.98, p = 0.015), and AF_{24} intermediate and not differing significantly from either AF_{12} (d = 0.73, p = 0.058) or Co (d = 0.21, p = 0.562).

3.3.3.3. Available P stock

The greatest topsoil available P stock was found in AF_{12}, approximately double (+78 kg ha^{-1}) the quantity found in Co (d = 1.72, p < 0.001). Stocks in AF_{24} were intermediate and also significantly higher (+29 kg ha^{-1}) than Co (d = 0.85, p = 0.030). Over the whole measured soil column, stocks beneath AF_{12} remained approximately double (+ 221 kg ha^{-1}) those in Co (d = 1.57, p = 0.001). Stocks in AF_{24} were not significantly different from Co at 0-50 cm (d = 0.53, p = 0.153).

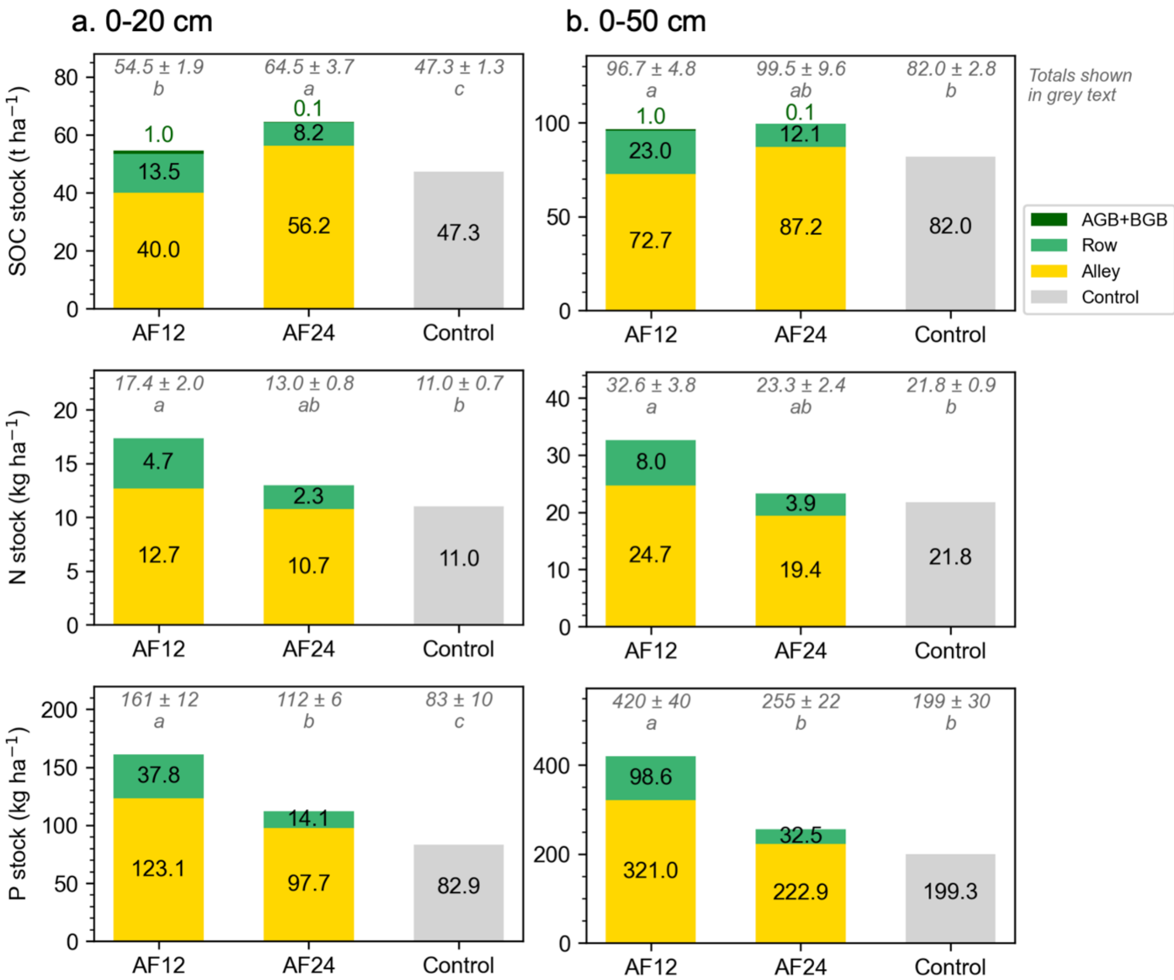

Figure 3.9 – Stock of SOC, available N and available P for a. 0-20 cm and b. 0-50 cm, by agroforestry system design (AF12 – 12 m-spaced tree rows planted 2002 with 110 stems ha^{-1}; AF24 – 24 m-spaced tree rows planted 2012 with 55 stems ha^{-1}). Per-hectare contributions of each agroforestry component to total stock are given by coloured blocks, control area contribution given by grey block. AGB+BGB represents estimated above-ground- and below-ground tree biomass C stock for the two agroforestry systems. Total stock for each system given in grey italic text, with letters denoting significant differences between totals.

3.4. Discussion

Examining soil functioning at a working farm allows us to consider the contributions of a practical agroforestry system to ecosystem benefit delivery. We discuss differences in soil functioning, firstly considering row and alley differences compared with the control area, secondly examining spatial effects, and finally separating the two agroforestry systems to consider ecosystem benefit delivery and trade-offs associated with alley width choice.

3.4.1. Functioning in tree rows and cropped alleys compared with control area

Agroforestry had a significant influence over common soil function indicators, compared with the treeless control area. We discuss which differences were significant in the tree row only, and which were also significant between control and identically-managed agroforestry alleys.

Bulk density was nearly 10 percent lower beneath agroforestry tree rows and alleys compared with the control, implying that trees can reduce soil compaction in cultivated areas as well as uncultivated areas. Decreased compaction due to afforestation is well known and observed elsewhere (Messing et al., 1997; Olszewska and Smal, 2008; Korkanç, 2014; Ashwood et al., 2019), derived from OM additions from tree root, shoot and exudate material (Jobbágy et al., 2001; Haichar et al., 2014; Judson et al., 2023). However, our findings differ from others in that reduced compaction was not confined to tree row areas but extended into adjacent cultivated alleys. Notably, the difference we observed in bulk density between control and cultivated alleys was not apparent in topsoil but was significant in subsoil. Differences in subsoil compaction may be due to the influence of tree roots extending beyond the tree rows and into the alleys at greater soil depth, an effect which has been demonstrated elsewhere (Cardinael et al., 2015b). Reduced compaction in alleys carries several benefits, such as better water and nutrient uptake by plants (Arvidsson, 1999) with positive implications for agricultural productivity. Addition of SOM from trees as root and shoot litter and C-dense exudates leads to aggregate formation and increased infiltration rates (Franzluebbers, 2002). Extra organic matter in the vicinity of trees can further reduce compaction indirectly by stimulating earthworm diversity and activity (Lavelle et al., 1998).

Similarly, available P content was elevated by 70% beneath agroforestry alleys, compared with the control. Unlike bulk density, higher available P was found in alley topsoil as well as subsoil, despite alleys having the same agricultural management as the control. Additional P is most likely derived from tree litter dispersed into the alleys decomposing into surface soil, as has been shown for other species (e.g. Guazuma ulmifolia, Hoosbeek et al., 2018). As Pardon et al. (2017) noted, soil nutrient concentration tends to decline more gradually with distance from mature trees (>15 years) compared with younger trees, and we found considerably more P in the alley of the older (AF_{12}) of the two treatments. Trees and crops in agroforestry are known to share networks of arbuscular mycorrhizal (AM) fungi (Ingleby et al., 2007) which in-turn thrive in and facilitate access to soil P (Qiao et al., 2022). Mature trees can therefore have significant positive influence on soil fertility in the cropped area of agroforestry systems.

For other soil indicators, soil function in agroforestry alleys more closely resembled the control area. For example, an additional 12.5 t ha^{-1} SOC stock to 50 cm depth was found beneath the tree row compared with the control, although no significant difference in SOC stock was found between alley areas and control. Smaller SOC stock differences in alleys compared with tree rows is a common finding – for example, Cardinael et al. (2015a) in a study of similar age trees (18 years) found a difference of +17.7 t ha^{-1} between tree row and control for soils across the 0-50 cm depth range, and a difference of only +2.6 t ha^{-1} between alley and control. Annual harvesting and regular tillage in the alley have been shown to result in higher C losses from the soil compared with undisturbed tree rows (Hooker et al., 2005; Dawson and Smith, 2007).

Similarly, K_s was three times faster in surface soil beneath tree rows, presumably due to lowered surface soil bulk density, with no significant difference in K_s between agroforestry alleys and the control where bulk density was similar. Trees and hedges are known to increase surface K_s in farm soils (Marshall et al., 2014; Holden et al., 2019). Limited evidence exists for this effect extending far from trees themselves and we did not observe the same effect in the agroforestry alleys, although hydraulic gradients may be produced up to 10 m either side of drier hedge soils (Caubel et al., 2003). On sloping ground, agroforestry can alter hydraulic functioning of soil at a distance from trees, provided rows are planted cross-slope. Siriri et al. (2006) in a study of a sloping (10%)

silvoarable terrace system found the hydrological influence of *Calliandra calothyrsus* trees to extend well into the cropped area due to their extensive rooting system. K_s in their alley crops adjacent to trees were nearly three times higher (22 mm hr^{-1}) than for a sole crop control (8 mm hr^{-1}). Although cross-slope planting produces complications for modern agricultural machinery, this nonetheless demonstrates the extra benefit of cross-slope planting for soil infiltration and potential flood mitigation compared with downslope alley planting.

3.4.2. Spatial and hillslope interactions between silvoarable soil properties

Lower available N, available P and moisture content at the alley edge sample positions imply competition for these resources between tree and crop at the row-alley boundary at the time of measurement, particularly given that lower N and P at the alley edge was found more in subsoil than topsoil. Studies utilising artificial soil barriers at the tree-crop boundary (Jose et al., 2000a; Jose et al., 2000b; Zamora et al., 2009) found that trees successfully compete for both water and available N at the alley edge. In these studies, crop water uptake was higher with a barrier in place, whereas tree water uptake was higher (at the expense of the crop) with no barrier as tree roots took up water from beneath the crop. Moreover, crop plants acquired a higher proportion of mineralised N from soil in the presence of a barrier. The row-alley competition zone we observe may coincide with tree dripline effects, however soil moisture is often higher at the dripline and particularly in subsoil (Alva et al., 1999), which is the opposite of what we observe. It must be noted, however, that although our study found lower N and P content at the alley edge, overall abundance of N and P in the agroforestry alleys was no lower (and in some cases, significantly higher) than in the control area. Moisture content was lower at the alley edge when compared with the control area, although this is less likely to inhibit crop productivity in areas of low water stress. Pardon et al. (2018) found that tree age and crop type were the major determinants of yield changes at alley edge locations. Only near mature trees were yield losses significant, and there was minimal effect on winter cereals for any tree category. Similarly, Cardinael et al. (2015b) found that trees and crops may form complementary rooting systems, facilitating better resource capture and circularity much deeper in the soil profile, in addition to the positive effect of complementary AM fungal networks (Ingleby et al., 2007). Spatial knowledge of yield data, which was not available at our study site, would be useful for determining the influence of competition effects on crop productivity at the alley edge.

Differences in particle sizes between treatments and slope positions implied that agroforestry influenced erosion susceptibility across the study area. In the absence of trees, a 3.5% higher clay fraction, 0.83% higher SOM content and 15% higher SOC stock were observed downslope, with no equivalent effect in the agroforestry areas. This implies downslope fine soil movement in the absence of trees, explaining why SOC stock differences between agroforestry and the control were only significant in upslope positions. Water erosion selectively transports finer particles from soil, leaving coarser material behind along with stones (Durán Zuazo and Rodríguez Pleguezuelo, 2008). Given that most SOM is stored in shallow topsoil, this 'fining' process leads to downslope transport and loss of organic matter from soil to watercourses, in addition to possible respiration loss of SOM as CO_2 through disturbance (e.g. Six et al., 2001). Agroforestry at this study site is therefore likely contributing to ecosystem service provision by limiting downslope erosive transport and loss of organic matter, even though trees are planted in rows parallel to slope. In addition to trees, the understorey within the tree rows is likely to be contributing to this particle size effect, as others have found (e.g. Dabney et al., 2001; Anderson et al., 2009; Monger et al., 2022a), and further study focussed on the understorey would be welcome to confirm this. Inhibiting OM loss protects soil aggregates, which in turn improve soil porosity, infiltration and ultimately, productivity (Boyle et al., 1989; Franzluebbers, 2002; Durán Zuazo and Rodríguez Pleguezuelo, 2008); meanwhile C stocks in soil undisturbed by erosion are not lost to respiration or downstream transport (Harden et al., 1999; Kirkels et al., 2014).

3.4.3. Choice of alley width and whole-system benefits of agroforestry

Narrower, 12 m tree row spacing exhibited higher soil fertility compared with the control area, equivalent to alley areas contributing 122 kg ha^{-1} more available P stock to 50 cm depth over 21 years. In contrast, the difference was only 24 kg ha^{-1} in 24 m alleys compared with the control. Although trees in the 12 m system are twice the age of trees in the 24 m system and age is likely to be contributing to some of the difference between systems, a five-fold difference in effect size between them implies that age cannot be the only contributing factor to differences in P stock. Two other effects may be contributing. Firstly, Steinfeld et al. (2024) found that more densely-planted agroforestry systems produce litter with higher concentrations of both N and P, which then decomposes in both the row and alley in the vicinity of the trees, contributing to soil nutrient stocks. Elevated N in soil has also been shown to produce higher N concentration in apple tree litter (Kowalczyk et al., 2017). The AF_{12} system has higher planting density (110 stems ha^{-1}) than AF_{24} (55 stems ha^{-1}) and is therefore likely to generate a higher density of root and shoot litter. Secondly, closer tree rows are likely to form a more 'closed loop' system, with nutrients leached into subsoil better intercepted by denser subsurface tree root and AM fungal networks beneath the main crop (Rowe et al., 1999; Ingleby et al., 2007; Tully et al., 2012; Cardinael et al., 2015b). Available soil fertility for crops therefore trades off against alley width and cultivatable arable area. Although greater fertility is available in an agroforestry system with narrower alleys, this will trade-off against factors such as shading at alley edges (Karim et al., 1993; Swieter et al., 2021) and evaluation of best width will therefore depend on the relative weighting of factors. Including tree row contributions to fertility in the AF_{12} system, extra available P stock nearly doubles to 220 kg ha^{-1} over 21 years, although extra fertility beneath trees is considerably less available to crops.

Recent policy recommendations in the UK advise that 10% of cropland (440,000 ha) be converted to silvoarable agroforestry, in order to store an extra 2.2 t C ha^{-1} $year^{-1}$ (3.5 Mt CO_2e $year^{-1}$) contributing to net-zero emissions by 2050 (Woodland Trust, 2022; Defra, 2023; CCC, 2025). However, of the two agroforestry systems surveyed here, only the 12 m system stored significant extra total C to a depth of 50 cm, compared with the control system (Table 3.1). The control-agroforestry C stock difference in the 24 m system, although slightly larger, was not significant at 95% confidence (Table 3.1). Nonetheless we can use the AF_{12} total C (SOC + tree biomass C) difference of +14.7 t ha^{-1} to consider the feasibility of stated policy recommendations. The SOC component of this difference (+13.7 t C ha^{-1}) corresponds to an annual soil C storage of 650 kg C ha^{-1} $year^{-1}$ over the 21-year lifespan of trees in the AF_{12} system, which represents a contribution of 29% to the 2.2 t C ha^{-1} $year^{-1}$ figure for potential annual C storage in silvoarable systems (Woodland Trust, 2022). However, we estimate only 1.0 t C ha^{-1} (50 kg C ha^{-1} $year^{-1}$) extra C contributed by tree biomass in the AF_{12} system, for a total of 700 kg C ha^{-1} $year^{-1}$ which is more than three times less than the target storage figure.

Table 3.1. Comparison of control-agroforestry differences in SOC stock and sequestration rates between this study and studies with similar tree age and climate zone. Differences marked (ns) are not significant at 95% confidence. 'LFH/OH' refers to LFH/OH soil horizons.

				Difference in SOC stock					
				LFH/OH		**0-20 cm**		**0-50 cm**	
Study	**Location**	**Age**	**Land Use Change**	Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})	Amount (t ha^{-1})	Rate (t ha^{-1} yr^{-1})	Amount (t ha-1)	Rate (t ha^{-1} yr^{-1})
This study (12 m system)	Devon, UK	21	Arable to silvoarable	-	-	+7.2	+0.34	+14.7	+0.70
This study (24 m system)	Devon, UK	11	Arable to silvoarable	-	-	+17.2	+1.56	*+17.5(ns)*	*+1.59(ns)*
Judson et al. (2024)	Yorks, UK	35	Arable to woodland	+8.9	+0.26	+15.8	+0.45	*+1.0(ns)*	*+0.03(ns)*
Mayer et al. (2022)	Various temperate	28	Various†	-	-	+7.0	+0.21	+10.1	+0.36
Cardinael et al. (2015a)	Hérault, France	18	Arable to silvoarable	-	-	+4.5	+0.25	+5.0	+0.28
Ashwood et al. (2019)	Midlands, UK	50	Arable to woodland	+9.1	+0.18	+37.2	+0.74	+63.5	+1.27
Upson and Burgess (2013)	Beds, UK	20	Arable to silvoarable	-	-	+7.2	+0.36	+20.8	+1.04

† Mayer et al. (2022) is a meta-analysis which includes the following land-use change (LUC) types: pasture to silvopasture, arable to silvoarable or hedge.

For SOC storage rates similar to this study, more than 70% of total C storage must therefore come from tree biomass for the 2.2 t C ha^{-1} $year^{-1}$ target to be reached. Yet literature values for proportional tree biomass contributions to silvoarable C storage are frequently much lower than 70%. Shi et al. (2018), in a meta-analysis of C sequestration potential of 217 silvoarable (alley cropping) systems globally, found C stock increases in silvoarable systems to be similar in soil and above ground biomass. Delivering 2.2 t C ha^{-1} $year^{-1}$ from silvoarable systems is therefore likely to require a large and significant SOC storage contribution of > 1 t C ha^{-1} $year^{-1}$, and where tree biomass is regularly harvested or pruned or where planting densities are lower than 150 stems ha^{-1}, SOC storage will have to be considerably greater still. In contrast, our study found that tree biomass C represented just 7% of annual C additions, with 93% derived from SOC.

Trees in this study were regularly pruned for apple production, limiting the contribution of tree biomass C to overall C storage to an extent, but this highlights practical considerations about the magnitude and longevity of above ground silvoarable C stocks.

Shi et al. (2018) found the sum of AGB C and SOC stock differences in silvoarable systems to be lower than for other systems such as shelterbelts and silvopasture, in part due to planting densities. Yet even shelterbelt and silvopastoral systems rarely generate forecast C storage at planting densities equivalent to this study. In a silvopastoral system of 14-year-old trees with a planting density of 110 stems ha^{-1} (equivalent to AF_{12} in this study), Upson et al. (2016) found area-adjusted tree biomass contributions to be just 4.0 t C ha^{-1} (290 kg ha^{-1} $year^{-1}$). A 2.2 t C ha^{-1} $year^{-1}$ figure was only reached in the case of continuous woodland (1,600 stems ha^{-1}), for which contributions from tree biomass C rose to 2.6 t ha^{-1} $year^{-1}$.

Finally, planting density for the 2.2 t C ha^{-1} target is assumed to be 150 stems ha^{-1}. This is high for conventionally cropped silvoarable land and is 1.5 and 3 times denser than that used in the AF_{12} and AF_{24} systems, respectively. Although new Countryside Stewardship funding (Defra, 2025) in England supports planting at 150 stems ha^{-1}, the 110 stems ha^{-1} figure from this study corresponds to 12 m alleys and it is considerably unlikely that new adopters of silvoarable agroforestry will plant in excess of this density due to the constraints it places on the size of modern farm machinery. Planting densities of ~50 stems ha^{-1}, appropriate for a 24 m system, are more likely in newly planted silvoarable agroforestry.

A realistic annualised SOC storage figure for silvoarable agroforestry is therefore likely to be considerably lower for the majority of systems. A number of explanations for low observed soil C sequestration are possible. For example, C stocks in the soils prior to planting may have been near saturation, such that organic matter derived from trees would not increase storage further (Stewart et al., 2007; Breure et al., 2025). However, this is unlikely as storage rates in this study are comparable to those found elsewhere for a range of soil types (Table 3.1). It is possible that the small difference in C storage between agroforestry and control areas in this study is due to similar incorporation of green manure in both treatments as they are under organic management, promoting build-up of C stocks in both and thus minimising the difference between them.

However, comparable studies of non-organic agroforestry systems (Table 3.1) did not observe C storage differences close to the modelled figures.

3.5. Conclusions

Using soil samples and measurements from a working silvoarable site first planted with trees in 2002 and subsequently in 2012, in Devon, SW England, we demonstrate how key soil functions – C storage, nutrient availability and hydrological functioning – are influenced spatially by hillslope position and with alley width. Benefits commonly associated with tree rows such as reduced compaction and nutrient addition to soil were readily transferred from tree rows to adjacent cultivated alleys, most likely via litterfall and lateral tree root influence. Reduced compaction was delivered in subsoil beneath crops, implying that SOM addition from tree roots extended into the cropped area. Alleys were 70% enriched in available P in both topsoil and subsoil compared with treeless areas. The strongest soil fertility improvements were observed in narrower 12 m cropped alleys, implying greater circularity in system nutrient dynamics and denser root and AM fungal networks than wider, 24 m alleys, albeit producing a trade-off between soil fertility benefits and cultivatable arable area. Future work incorporating yield data to estimate land-equivalent ratio would be welcome for determining the extent of this trade-off. Demonstrable soil structure and fertility improvements in agroforestry alleys imply that loss of cropable area from afforestation can be offset by improved functioning, in addition to known benefits from tree rows.

Several important spatial effects were found. Competition at the tree row/alley boundary was observed for N, P and soil moisture, which were depleted at the alley edge compared with the tree row and alley centre. Differences were predominantly found in subsoil, implying that tree roots extending into alleys were successfully competing with crops for resources. Nonetheless, overall alley N and P concentrations remained greater in the alley than the control area, implying that the competition effect had not diminished overall fertility. Agroforestry successfully mitigated erosive loss of SOM, despite tree rows being orientated parallel to the slope. Reduced erosion improves downstream water quality and limits C loss from topsoil. Although it presents issues for use of modern farm machinery, we hypothesise that this effect would be considerably stronger with cross-slope alleys, which would provide greater flood mitigation potential due to elevated saturated hydraulic conductivity in the vicinity of tree rows. Erosion and flood mitigation studies on cross-slope silvoarable systems would greatly enhance knowledge of these effects.

Considerable attention has been given to the contribution of agroforestry to C emissions mitigation from arable land. We found a significant difference of +14.7 t C ha^{-1} stored in the 12 m agroforestry system, compared with the treeless control. At 700 kg C ha^{-1} $year^{-1}$, this is more than three times smaller than the UK target for extra C storage in silvoarable systems of 2.2 t C ha^{-1} $year^{-1}$ (8 t CO_2e ha^{-1} $year^{-1}$). C storage contributions modelled in policy scenarios assume planting densities considerably higher (150 stems ha^{-1}) than is practicable for most silvoarable practitioners. Potential for silvoarable systems to contribute to national C budgets may therefore have been overestimated, although organic management in all treatments may explain small differences in soil C stock between control and agroforestry in this study. Further work could usefully compare C sequestration in organic systems with conventionally-managed agroforestry, and we note the need for dedicated experimental sites at which factors such as hillslope and silvoarable alley width can be more carefully constrained. We conclude that the contribution of silvoarable systems to temperate ecosystem service provision must be considered in terms of demonstrated multiple benefits that go beyond C sequestration, such as soil fertility benefit, natural flood management and biodiversity improvements. A holistic view of the ecosystem benefits of temperate agroforestry will ensure it contributes to future landscape resilience.

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Chapter 4. Spatial layout of trees on farms influences magnitude and timing of river flow peak

Abstract

Agroforestry, defined as farming with trees, is growing in popularity for its potential to deliver multiple landscape benefits, including natural flood management (NFM). Most NFM research has been concentrated in uplands, whereas optimising tree placement on farms for NFM in lowland catchments has not been studied. Using spatially-distributed TOPMODEL we consider rainfall-runoff changes under various agroforestry planting scenarios in a 1.75 km^2 lowland catchment in Yorkshire, UK, asking whether tree planting at densities practical for farming can significantly alter flood peaks, and which placement strategies generate the most significant effect on flood risk reduction for a fixed area of afforestation. Riparian woodland covering 2.5 % of farmed area (0.6 % of the catchment) attenuated flood peak by 2.6 % and delayed its arrival by 6.6 % (19 minutes). Shelterbelt or in-field copses had a more modest influence on flood peak, and linear features such as in-field tree rows or hedges had a negligible or opposing effect. Increasing tree coverage generated a diminishing return and woodland over the whole farmed area (25 % of the catchment) produced only 1.6 x the peak size reduction as 0.6 % riparian cover in low flow scenarios, rising to 13 times the reduction for higher flows, despite requiring 40 times more land taken out of agricultural production. Agroforestry sited on shallow gradients or in the riparian zone can therefore contribute to NFM without substantially impacting land available for productive agriculture. These findings demonstrate the capacity of agroforestry to deliver multiple landscape benefits.

4.1. Introduction

Flooding is among the costliest of global natural hazards, with losses totalling \$320bn in 2024 (Hartmann et al., 2019; Munich Re, 2024). Interrelated pressures such as climate forcing, intensifying resource demand and land fragmentation from infrastructure development mean that damages from flooding are growing with ‘high confidence’ in Europe and the United Kingdom (Hirabayashi et al., 2013; IPCC, 2014b). Millions of properties and significant quantities of critical infrastructure in the UK are threatened annually by flood risk, and the frequency of autumn rainfall events exceeding 50 mm has increased by 60% since 1900 (Thorne, 2014; Cotterill et al., 2021). The UK Met Office estimate extreme rainfall with a 100-year return period to be 2.5 times more likely by 2100 (Christidis et al., 2021), and according to the most recent UK Climate Change Risk Assessment (CCRA3, Sayers et al., 2021), damages from UK flooding in a 2°C future are expected to increase from £2bn in 2021 to £2.7-3.0bn by the 2080s, if flooding continues to be managed as it is now.

The changing nature of flood risk has prompted a move towards adaptive management and nature-based solutions. Decentralised and multifunctional green options for water management capable of delivering multiple benefits are increasingly favoured in place of traditional and capital-intensive ‘grey’ infrastructure solutions (Butler et al., 2017). In rural catchments, this includes natural flood management (NFM) – distributed measures which restore or enhance catchment processes to reduce flood risk, while fostering environmental benefits such as biodiversity, soil and water improvement, carbon sequestration, better agricultural productivity and greater public enjoyment of the countryside (Dadson et al., 2017). The goal of NFM is to enhance catchment hydrological functions such that the rate at which water is delivered to areas of downstream flood risk is reduced following rainfall (Lane, 2017), either by delaying or desynchronising the flood peak (Kingsbury-Smith et al., 2023). Changes in land use or management are generally deployed to achieve this, with the goal of reducing overland flow velocities, promoting inline or offline water storage and improving soil conditions in favour of infiltration (Zhu et al., 2024). Afforestation might be an effective means of achieving these goals in the form of riparian, cross-slope or other strategic planting strategies (Ferguson and Fenner, 2020; Marapara et al., 2020). In addition, trees in combination with understorey vegetation are known to increase surface roughness

(Monger et al., 2022a) and water holding capacity (Ashwood et al., 2019) of soils, decreasing runoff when compared with unforested land uses (Farley et al., 2005).

Land-use interventions for NFM include those which ‘spare’ land for flood risk reduction, or ‘share’ land for both flood mitigation and economically productive use such as agriculture. Agroforestry, defined simply as farming with trees (Woodland Trust, 2022), is a land-sharing strategy known to be capable of reconciling food production and ecosystem service provision from farmland (Burgess, 1999; Araujo et al., 2012; Torralba et al., 2016; Judson et al., 2023), with benefits including improved hydrological functioning of agricultural soils leading to reduced local flood risk factors (Marshall et al., 2014; Monger et al., 2022b). Trees may be variably sited within agroforestry systems, such as in rows with arable inter-crops, as isolated trees or copses within pasture, as shelterbelts, or in riparian zones (Nair, 2005; Judson et al., 2023). As for upper-catchment NFM strategies, the effectiveness of agroforestry on flood risk reduction depends on placement strategies for trees in the farmed landscape (Marapara et al., 2020; Monger et al., 2024), and strategies will trade off with economic priorities for land depending on how placement interferes with traditional agricultural activity. Although questions surrounding flood peak sensitivity to NFM vegetation management have been addressed in upland areas (Gao et al., 2016; Bond et al., 2022; Kingsbury-Smith et al., 2023; Monger et al., 2024), spatial evaluation of agroforestry systems for NFM benefit in lowland catchments has not been undertaken. Stakeholders either planting or funding lowland agroforestry for flood benefit have the same questions as those supporting NFM in uplands, namely, optimal coverage rates to benefit effect on flood peak size (Monger et al., 2024), whether systematic strategies such as focussing on steeper or shallower areas of a catchment variably ‘slow the flow’ (Maske and Jain, 2013; Gao et al., 2016; Bond et al., 2022), and how these factors trade off with other constraints on land use. As temperate agroforestry grows in popularity as a means of diversifying ecosystem benefit delivery from farmland, particularly in support of net-zero (Woodland Trust, 2022) but also slowing or storing water in the landscape, these questions will only grow in importance.

This study asks two research questions: firstly, whether agroforestry planted at densities practical for temperate zone farming can significantly alter downstream flood risk, and secondly, which systems generate the most significant effect on flood risk reduction for a fixed area of afforestation. Understanding the effects of localised NFM interventions at catchment scale depends on thorough understanding of spatial connectivity of flow paths and timings (Rogger et al., 2017), so we use a spatially-distributed version of TOPMODEL (SD-TOPMODEL) (Gao et al., 2015) to consider a range of agroforestry planting strategies in a small (175 ha) lowland catchment in Yorkshire, UK. In this way, we address important considerations for practitioners looking to design agroforestry systems to maximise flood management benefits within financial or land-area constraints, and policy efforts to deliver multiple goods from farmed landscapes.

4.2. Methodology

4.2.1. Study site

A small, 175 ha lowland catchment in the Vale of York, 11 km NE of Leeds, UK, was chosen for the study of which 25% (44 ha) comprised a livestock farm (Fig. 4.1) in which agroforestry scenarios were modelled. By 'lowland' we refer to a standard UK conception of lowland grasslands (appropriate in the context of a pasture farm such as this): 'grasslands that occur either in enclosed or unenclosed situations but typically below the upper level of agricultural enclosure in any area and usually below 300 m' (Jefferson et al., 2014). Mean annual precipitation (1991 to 2020) is 620 mm (Dec-Feb: 149 mm, Jun-Aug: 170 mm) and mean annual temperature is 10.0°C (Dec-Feb mean: 4.5°C, Jun-Aug mean: 15.9) (Church Fenton Met Station, 11 km away). Snow is rare and short-lived in the catchment. Elevation within the catchment ranges from 60 m in the southeast extremity to 115 m in the north and west (Fig. 4.1) and is hence a lowland catchment, with a mean slope of 4.9 %. The entire catchment drains through the livestock farm (Fig. 4.1), such that discharge at the outlet is maximally sensitive to land-use changes within the farm boundary. The catchment comprises arable land (39%), deciduous woodland (34%) and improved grassland (26%), with a very small area of hardstanding (1%) (Morton et al., 2023) (Fig. 4.1c).

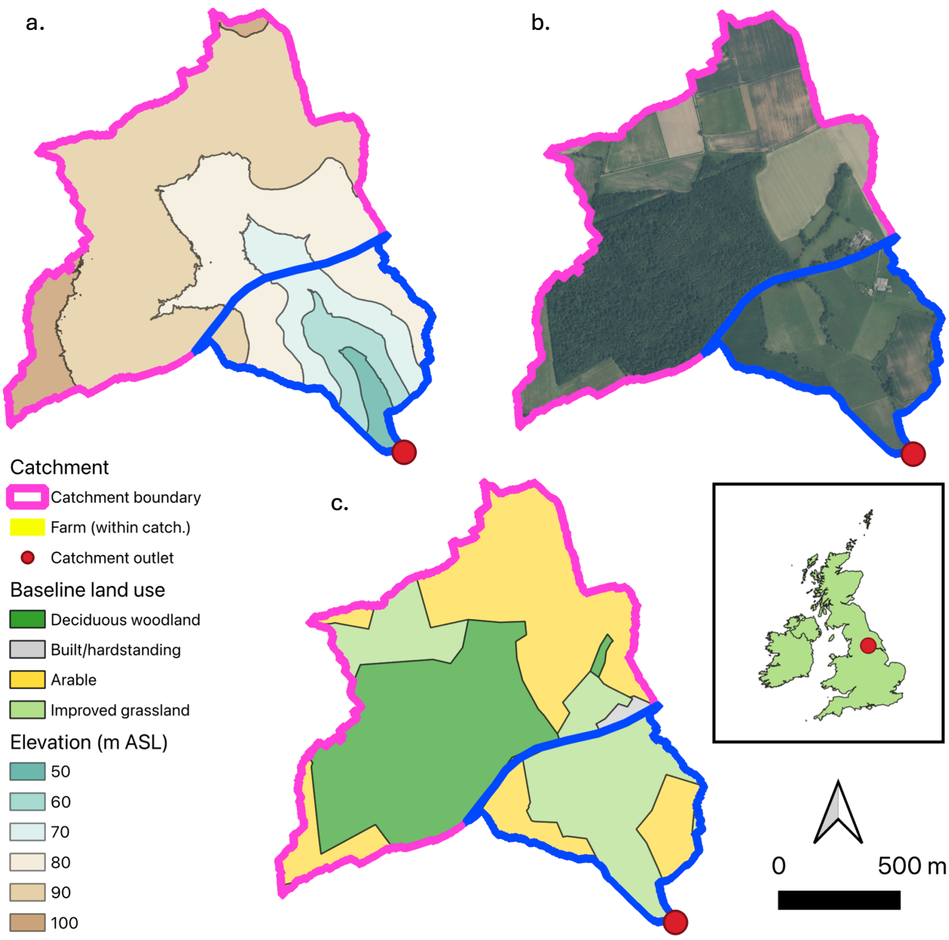

Figure 4.1 – Map of study catchment showing a. elevation, b. aerial photography and c. UKCEH baseline land use. Catchment area within farm boundaries delineated in blue. Based upon Land Cover Map 2023 © UKCEH 2024. Contains Ordnance Survey data © Crown Copyright 2007, Licence number 100017572. Aerial photography © Getmapping Ltd., accessible at EDINA Aerial Digimap Service (2022).

The catchment predominantly comprises fine loamy over clayey stagnogley soils up to 1 m deep from the Dunkeswick series (Cranfield University, 2022). The clayey subsoil is slowly permeable making it susceptible to seasonal waterlogging and rapid runoff in winter, and land is predominantly in seasonal pasture. In the lowest-lying 10 % (17 ha) of the catchment, adjacent to the outlet, are calcareous brown earths extending to ~50 cm depth from the Aberford series, well-draining soils typically found on low-dipping Permian and Jurassic limestones in mid- and northern England (Cranfield University, 2022). In this instance limestone from the Caedby formation underlies 43 % of the catchment to the west, with the remaining 57 % underlain by mud-, silt- and sandstones from the Rough Rock and Pennine Lower Coal formations (BGS, 2013).

4.2.2. SD-TOPMODEL

We use SD-TOPMODEL (Gao et al., 2015) to examine changes in catchment hydrological functioning derived from land-use change scenarios within the farm boundary. SD-TOPMODEL is a spatially-distributed version of TOPMODEL (Beven and Kirkby, 1979), a computationally-inexpensive and parameter-sparse flood-modelling algorithm using a soil water storage approach to predict saturation-excess overland flow within a catchment. This approach is particularly well-suited to UK basins in which infiltration rates are high compared with rainfall intensity, such that overland flow generation is controlled more strongly by saturation-excess rather than infiltration-excess mechanisms. Calibration of the model is achieved with three parameters for a given catchment: m – active depth for subsurface flow, K – a notional hydraulic conductivity of soil across the catchment, and k_v – an overland flow velocity constant proportional to $1/n$ where n is Manning's coefficient of roughness. Overland flow is treated as a function of rainfall intensity and saturated area within a basin, both of which vary by timestep introducing significant nonlinearity to a catchment's hydrographic response.

SD-TOPMODEL solves the original TOPMODEL runoff equations per-grid-cell across a two-dimensional user-defined grid extending uniformly across the catchment. Water is permitted to reach the saturated zone at different times according to local wetness, and is routed between cells according to local, temporal conditions (Bond et al., 2022). Importantly for this study, values for K and k_v are permitted to vary spatially by land cover type within the catchment, and for each land-use change scenario a spatial distribution of K and k_v is incorporated to evaluate catchment response to a synthetic storm event. The model is therefore ideal for determining the extent to which land cover changes within the livestock farm boundary influence the hydrograph response of the study catchment through spatial alteration of these two parameters. Additionally, SD-TOPMODEL is well-suited to areas of shallower soil and moderate topography (Gao et al., 2015; Beven et al., 2021; Bond et al., 2022), as are found at the study catchment.

4.2.3. Catchment and rainfall data

LiDAR elevation data were obtained at 2 m resolution (Ordnance Survey, June 2023) to generate a digital elevation model (DEM) of the study catchment (Fig. 4.1 a). This was combined with land-use data obtained from UKCEH (Morton et al., 2023), from which a baseline land cover map at the same resolution as the DEM was generated reflecting the current state of the catchment without the most recent NFM interventions included at the study site (Fig. 4.1 c).

Rainfall and runoff data for four synthetic storm events were generated for the catchment using the Revitalised Flood Hydrograph (ReFH, Kjeldsen, 2007). The ReFH model is widely used to generate synthetic storms based on catchment descriptors for ungauged catchments in the United Kingdom (Faulkner and Barber, 2009), and has been used to validate several applications of SD-TOPMODEL elsewhere (Gao et al., 2016; Bond et al., 2022; Kingsbury-Smith et al., 2023). We use combinations of two storm durations (4-hour and 10-hour) and two return periods (10-year and 50-year) (Table 4.1). A four hour length was chosen as it is the recommended storm duration for the study catchment (Kjeldsen, 2007), and an alternative, 10-hour duration and two respective intensities chosen to maximise consistency of model performance (Monger et al., 2024). Observations for these storms were at 15-minute intervals (timesteps). The purpose of using synthetic storms was to calibrate initial parameter values of K, k_v and m for the catchment, after which performance of chosen parameter values was validated using field measurements at the study catchment. Additionally, a baseflow value for the catchment was estimated using ReFH and applied at calibration, validation and scenario testing stages. As in previous work with SD-TOPMODEL (Bond et al., 2022; Kingsbury-Smith et al., 2023), storm events were chosen such that specific return period events could be represented, and rainfall for each storm was distributed with a Gaussian profile typical of winter frontal storm events in which rain falls continuously with peak intensity midway through the storm. Typical evapotranspiration levels of 1-2 mm day^{-1} found in the UK (Blyth et al., 2019) were treated as negligible compared with storm intensity for these events (min. 80 mm day^{-1}) and therefore excluded from the model.

Table 4.1 – Synthetic storm events (derived from the Revitalised Flood Handbook) used for model calibration.

Storm duration (h)	Storm recurrence interval (years)	Total rainfall (mm)	Mean rainfall intensity (mm hr^{-1})	Maximum rainfall intensity (mm hr^{-1})
4	10	24.89	5.86	15.10
4	50	34.57	8.64	20.96
10	10	33.27	3.33	8.55
10	50	45.00	4.50	11.57

4.2.4. Parameter sources

Land use for different agroforestry scenarios was represented by spatially-variable values of K and k_v specific to each land cover. As elsewhere (Gao et al., 2015; Bond et al., 2022; Kingsbury-Smith et al., 2023; Zhu et al., 2025), these were estimated using literature sources and expressed as relative values (Table 4.2) to be multiplied by calibrated factors determined in Section 4.2.5. We also express values of K and k_v relative to a single land cover (Deciduous Woodland) for ease of comparison (Table 4.2, values in parentheses).

In order to understand trade-offs between land-use scenarios and differing placement of trees, only one set of parameters was used to model all types of woodland used for agroforestry (silvopasture, shelterbelt and riparian woodland, Table 4.2). Previous studies using SD-TOPMODEL (Bond et al., 2022; Kingsbury-Smith et al., 2023) differentiate between deciduous woodland of different ages or types when assigning parameter values, reflecting differences in establishment or understorey cover, and we do so only in differentiating between well-established woodland included in UKCEH baseline land cover, and more recent broadleaf woodland established for NFM. The purpose of the study is not to describe current hydrological characteristics of the catchment in detail, but to hypothesise more generally about differences between planting scenarios which could be replicated elsewhere, and to compare the effectiveness of planting an equivalent area of trees in different parts of the catchment. We therefore choose equivalent parameters for all agroforestry systems modelled, assuming that the trees are deciduous, well-established and fenced from livestock to allow spontaneous development of understorey vegetation. This allows us to preserve model simplicity and prevent overfitting. We nonetheless note that parameter choice represents the most significant source of uncertainty in the model (Kingsbury-Smith et al., 2023), and more work is needed to harmonise parameter estimates across land covers.

Significant variability exists between studies in parameter estimates, with estimates for K and k_v in some cases differing by an order of magnitude or more for similar land cover descriptions. Given that relative differences between empirical values are necessary for the model we prioritised consistency of literature sources over breadth, such that estimates were derived from the same methodology where possible. Lowland estimates for k_v were not available for land cover types in this study. In the manner of Kingsbury-Smith et al. (2023) we therefore base estimates on literature values for Manning's n from Chow et al. (1988), with k_v proportional to $1/n$, in order to preserve consistency of methodology between estimates aiding relative comparisons. As with Kingsbury-Smith et al. (2023) we distinguish between prior, established woodland and agroforestry planted for NFM.

Table 4.2 – Land-use parameters for model calibration and testing hydrograph response of spatially-explicit land-use scenarios. Relative differences in parameter values between land cover types are estimated from field measurements and literature sources (for ease of comparison, these are given relative to Deciduous Woodland in parentheses for K and k_v).

Land cover	Baseline cover (%)	Current cover (ha)	k_v	K	m	Interception (%)
Deciduous woodland	34	59.70	0.8 (1.0)	0.90 (1.00)	1.0	20
Improved grassland	26	43.95	2.4 (3.0)	0.55 (0.61)	1.0	0
Arable	39	65.69	2.4 (3.0)	0.55 (0.61)	1.0	0
Suburban/hardstanding	1	1.41	10.0 (12.5)	0.10 (0.11)	1.0	0
Silvopasture/Copse	-	0.04	1.0 (1.25)	0.80 (0.89)	1.0	20
Shelterbelt	-	0.93	1.0 (1.25)	0.80 (0.89)	1.0	20
Riparian woodland	-	1.28	1.0 (1.25)	0.80 (0.89)	1.0	20
Hedge	-	0.60	1.0 (1.25)	2.00 (2.22)	1.0	20

We obtained relative values of K using field measurements by Kingsbury-Smith et al. (2023) in N Yorkshire, UK. These included relative values for established deciduous woodland, woodland planted for NFM and improved grassland. Arable land use for the catchment was set to the same K value as improved grassland. We use measurements from Holden et al. (2019) at a site less than 5 km from this study site on a farm with similar topography and soil type to obtain a relative value of *K* for hedge cover. For hardstanding we set *K* to the model's minimum value (Bond, 2022).

Interception values for the land cover types studied (Table 2) also vary considerably between estimates from temperate, broadleaf studies such as 13% (Dolman, 1987) or 25% (Herbst et al., 2008). A conservative, intermediate value of 20% was therefore set for hedgerows and all woodland types. As in Bond et al. (2022), values for winter were used as a conservative estimate of interception, given that the values were not derived at the same site. Interception for land cover without woody vegetation was set to zero.

As elsewhere (Beven and Kirkby, 1979; Bond, 2022), *m* was treated as a scaling parameter only and held constant across the catchment. Although *m* is related to soil depth in that it is a measure of the rate at which flow reaches zero with increasing depth, many other soil physical properties influence its spatial variability (Beven, 2012). Soil type is known to exert stronger influence on soil water storage than vegetation type (Geris et al., 2014); while soil from the Dunkeswick series is typically deeper (1 m), it is slowly permeable in subsoil and therefore well-draining to an approximately equivalent depth (50 cm) as soils in the Aberford series (Cranfield University, 2022).

4.2.5. Calibration

A baseline land cover map of the catchment (Morton et al., 2023) with 10 m grid cells was used to calibrate the model. Calibration refers to establishing three multipliers corresponding to *K*, k_v and *m* specific to the study catchment ('calibrated values', Table 4.3), which are used to transform relative values specific to each land cover given in Table 2. The UKCEH baseline land cover was used as it most closely represents land cover assumed by ReFH design storms.

Model performance was evaluated for the calibration stage by comparing the flood hydrograph predicted by SD-TOPMODEL with the four chosen ReFH synthetic storms (Fig. 4.2). ReFH design storms are widely used in this way to estimate discharge for ungauged UK catchments based on known catchment characteristics (Monger et al., 2024). Goodness of fit was determined by Nash-Sutcliffe Efficiency (NSE), a commonly-used measure of model performance (Ritter and Muñoz-Carpena, 2013; Gao et al., 2015; Bond et al., 2022).

Table 4.3 – Parameter bounds and step size used for model calibration, including calibrated values

Parameter	Minimum value	Maximum value	Step	Calibrated value
K	4.0	8.0	1.0	**5.0**
k_v	2.5	15	2.5	**5.0**
m	0.010	0.020	0.002	**0.014**

Parameter sets for calibration were chosen by scanning the parameter space uniformly between minimum and maximum values (Table 4.3) (Gao et al., 2015). Choice of reasonable figures for minimum and maximum values was guided by prior studies using SD-TOPMODEL, and 720 simulations (180 per storm) were used in which each parameter set was applied to the four synthetic storms (Table 4.3). The best parameter set was chosen to be that which resulted in the highest average NSE value between the four storms (Fig. 4.2).

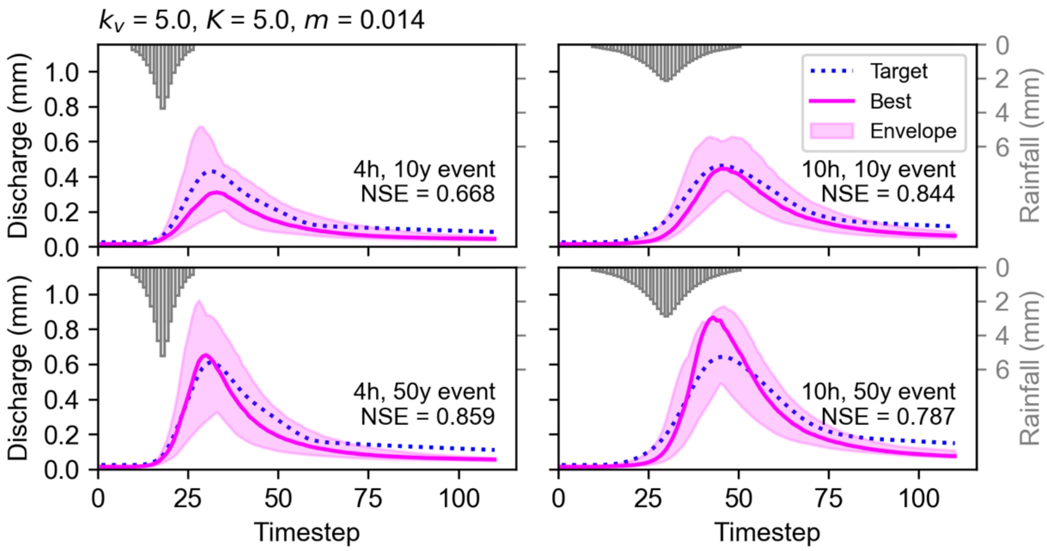

Figure 4.2 – Calibration results for the study catchment. Grey bars represent study catchment synthetic rainfall events estimated from the Revitalised Flood Handbook (ReFH) for four storm events. Blue dotted line represents estimated ReFH hydrograph. Pink line shows SD-TOPMODEL estimated hydrograph using selected parameter values for K, k_v and m. Pink shaded area indicates hydrograph bounds for parameter sets with NSE in highest 20% of tested values.

4.2.6. Validation

The fully calibrated model was validated for its performance against empirical rainfall and river level data measured between April and December 2024 at the study catchment outlet, using a land cover map with 2 m resolution based on current land use. The catchment is ungauged, and discharge data were not available, so goodness of fit was determined by visual comparison of time-to-peak between river level and calibrated-model responses to measured rainfall (Fig. 4.3).

Table 4.4 – Measured storm events used for validating calibrated model.

Validation storm no.	Storm duration (h)	Total rainfall (mm)	Mean rainfall intensity (mm hr^{-1})	Maximum rainfall intensity (mm hr^{-1})
1	10	17.80	1.78	6.26
2	18	35.82	1.99	6.22
3	23	30.23	1.31	4.74

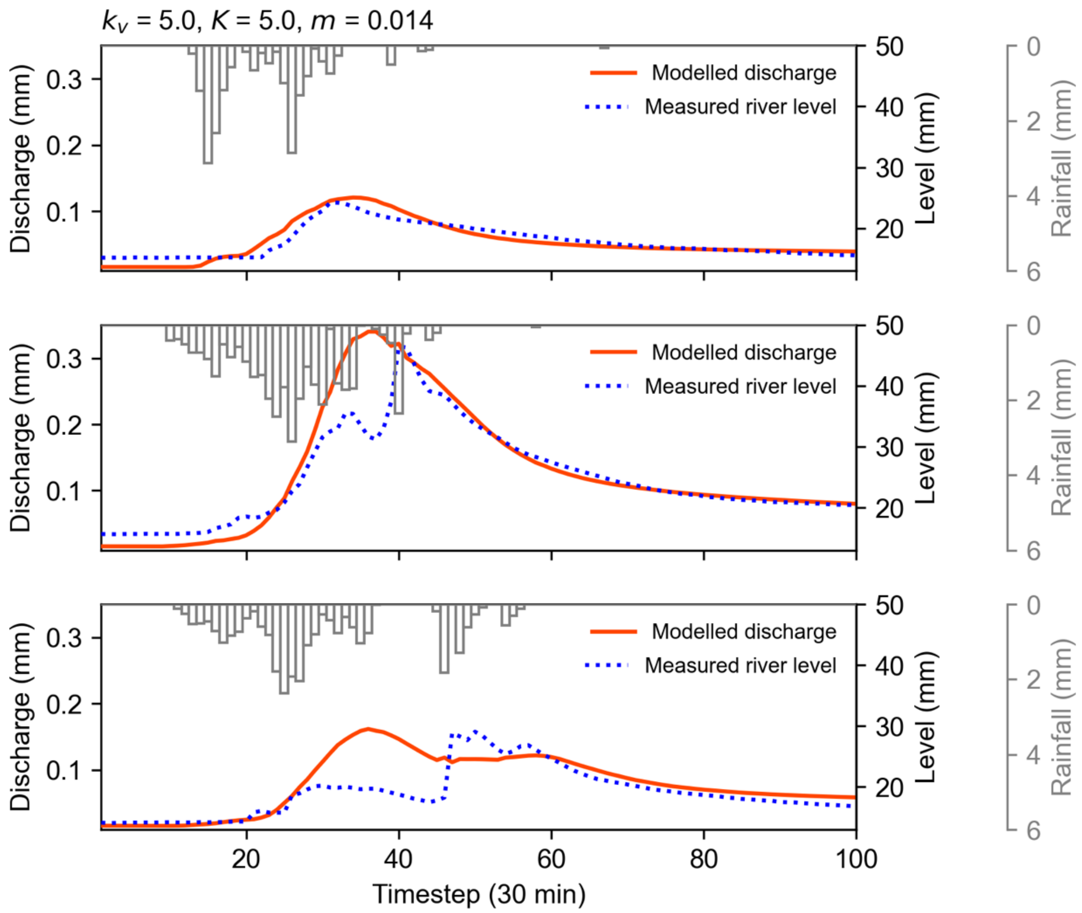

Figure 4.3 – Validation results for the study catchment, for three storm events of a. 10 , b. 18 and c. 23 hours in length. Grey bars represent rainfall recorded Spen Farm (4 km away). Blue dotted line (right black axis) indicates river level recorded at the catchment outlet. Orange line represents SD-TOPMODEL estimated hydrograph for recorded rainfall, using parameter values for *K*, k_v and *m* selected at the calibration stage. With only river level data, curves are not intended to match, rather the purpose of validation is matching peak timing between measured river level and modelled runoff.

4.2.7. Land cover scenarios

A number of agroforestry and general woodland scenarios represented at 2 m resolution were tested using the model, in order to consider how a range of placement strategies for trees and hedgerows influence the hydrological response of the catchment, compared with the land-use baseline. All scenarios tested correspond to planting within the livestock farm boundary only.

4.2.7.1. Current land use

The first group of scenarios examines whether introduction of hedgerows and trees at a scale similar to that currently planted at the farm and typical of lowland agroforestry schemes has a meaningful effect on downstream flood management, and how an equivalent area (1 ha) of several realistic interventions – hedgerow, copse, cross-slope tree rows, down-slope tree rows, shelterbelt and riparian woodland – compare in terms of potential to attenuate flood peak following a significant storm event. Planting 1 ha of each agroforestry type corresponds to 2.5% of the livestock farm area and 0.6 % of the whole catchment, and is also currently the minimum woodland planting area to qualify for financial support under the England Woodland Creation Offer (Forestry Commission, 2025). A hypothetical 'endmember' scenario, in which the whole farm area is planted with continuous woodland (25 % of catchment) is included to determine a theoretical limit to NFM benefit of tree planting at the study site against which other agroforestry options can be compared.

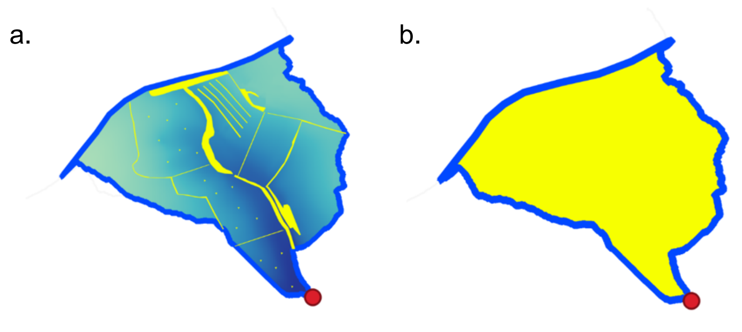

Figure 4.4 – Within-farm scenarios examining a. combined agroforestry as planted (current land use) and b. a theoretical endmember scenario with the whole farmed area under woodland. Combined agroforestry is also subdivided into separate scenarios for each vegetation type (shelterbelt, riparian woodland, cross-slope tree rows, hedgerow, copse) with each normalised to a 1 ha area, illustrated in the Supporting Information. Outlet point shown in red.

The purpose of testing agroforestry scenarios mirroring current land use is to consider whether minimal intervention commensurate with normal farm operations can contribute to NFM, which planting type has the greatest impact, and how the influence of current planting strategies compare with an endmember scenario.

4.2.7.2. Equal area scenarios

The second group of scenarios, mirroring the approach of Gao et al. (2016), examines systematic approaches to tree planting based on catchment topographic characteristics. Equal areas of woodland are placed within the farm boundary in the following areas: areas of steepest gradient, areas of shallowest gradient, the riparian zone and in woodland copses dispersed at random.

The areas compared correspond to 10% tree cover of the livestock-farmed area (Fig. 4.5a – 4.5d) – equivalent to the amended Universal Action for Tree Planting in Wales as part of the Sustainable Farming Scheme (Welsh Government, 2024) – and 20% tree cover of the livestock-farmed area (Fig. 4.5e – 4.5h). The purpose is to consider which coverage of trees deliver greatest NFM benefit, and how benefit increases with cover.

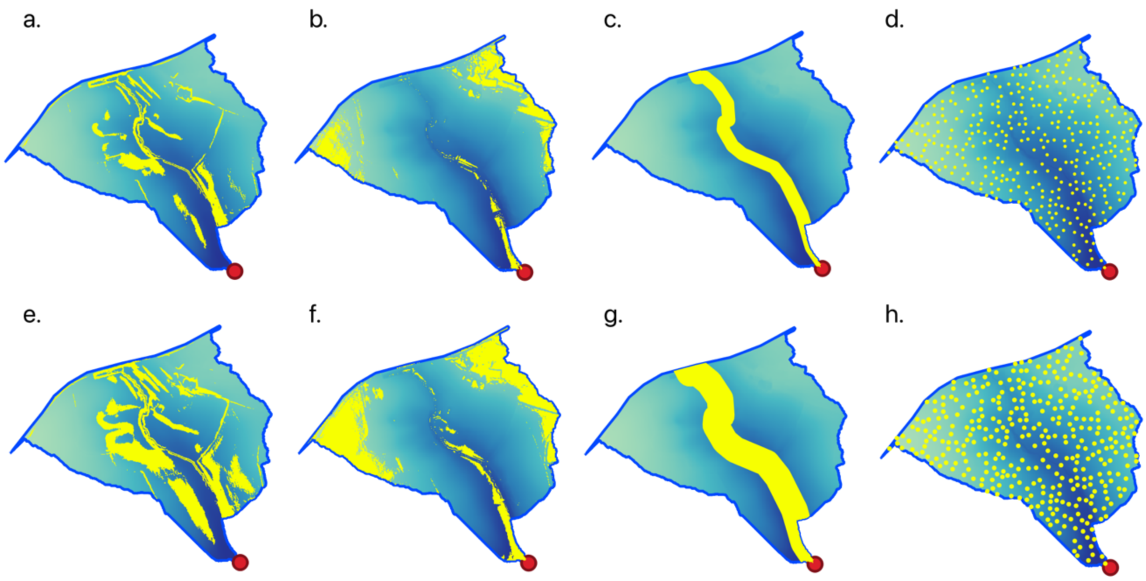

Figure 4.5 – Equal area agroforestry scenarios within the farm boundary. Scenarios a – d illustrate 10% woodland cover of a. steepest gradient, b. shallowest gradient, c. riparian zone and d. randomly distributed copses. Scenarios e – h illustrate 20% cover of the same planting strategies. Catchment outlet where stream level was recorded shown in red.

4.2.8. Analysis methods

For each planting scenario, as in other studies, two metrics – size of peak and time to peak – were used to evaluate differences in runoff characteristics compared with UKCEH baseline land use for the catchment (Gao et al., 2016; Bond et al., 2022; Kingsbury-Smith et al., 2023). Response hydrographs were smoothed using an interpolating B-spline function, removing artefacts whereby peaks may be located between two points of similar discharge value, and allowing us to compare small differences between peak size and time to peak without compromising the shape or meaning of observed or modelled time series (Unser et al., 1993; Oh et al., 2020). For this we used the *make_interp_*spline algorithm from the *SciPy* package within *Python* (Virtanen et al., 2020). Hydrograph differences between scenarios are illustrated in the Supporting Information (Supplementary Fig. C.3). We compare scenario responses for storms of the same two return periods and durations used in model calibration to understand response dynamics of tree planting scenarios under different flow intensities.

4.2.9. Limitations and uncertainty

As others have noted (Gao et al., 2016; Kingsbury-Smith et al., 2023), parameter selection represents the largest source of uncertainty in the model, particularly as estimates of parameter values corresponding to land cover types can vary so considerably between studies and contexts. However, the possibility of multiple behavioural parameter sets is a well-known limitation of hydrological modelling more generally (Cameron et al., 1999; Gao et al., 2015), and is overcome here and elsewhere by systematic scanning of the parameter space informed by prior studies using TOPMODEL (Beven, 1997; Gao et al., 2015; Kingsbury-Smith et al., 2023) and across storm types to determine best-fit values. We also note considerations with using ReFH data for calibration: the ReFH model is based on catchment descriptors alone and does not account for complex and asynchronous flood generation patterns (Faulkner and Barber, 2009; Rogger et al., 2017). This introduces uncertainty to flow estimates, and is only intended to provide an estimated range of peak flow values in the absence of recorded discharge data. For this reason, we include empirical river level data as a check on calibrated values, and good peak timing fit between the calibrated model and measured river level data indicate that the model is capable of generating sufficiently realistic response hydrographs for the purpose of comparing outcomes from land-use scenarios. The purpose of the study is not to determine absolute values for flood peak sizes and arrival times for the catchment, but to consider relative differences between scenarios and baseline land use. Methodological consistency between scenarios, in combination with robust calibration and validation, ensure that parameter and modelling uncertainty are minimised.

4.3. Results

4.3.1. Equal area: 1 ha agroforestry scenarios

For shorter, 4-hour storm durations, the majority of 1 ha agroforestry scenarios produced small differences in flood peak timing, with the exception of riparian woodland which reduced peak size by 2.6% and delayed flood peak arrival by 6.6 % (19 minutes) for a 10-year return period storm event (Table 4.5, Fig. 4.6). These figures were –2.4 % and +4.2 %, respectively, for a 50-year return period event (Table 4.5). Shelterbelt planting produced a small effect on peak size (10-year: –0.6 %, 50-year: –1.2 %) as did copse planting (10-year: –0.7 %, 50-year: –0.6 %), whereas hedgerows, cross-slope and downslope tree rows all produced very small or opposing differences; the 1 ha hedgerow scenario produced a very small increase in peak runoff (+0.3 %). For all 1 ha agroforestry systems (or tree planting schemes), a longer return period of 50 years led to a larger absolute difference in peak runoff compared with baseline land use, with riparian planting still producing the largest difference; and smaller absolute differences in time-to-peak for most 1 ha scenarios.

For longer, 10-hour storm durations (Table 4.6), change in peak runoff compared with baseline land use was smaller and more similar between scenarios, particularly for higher flow (Table 4.6). None of the 1 ha scenarios produced a peak size difference exceeding 1 % in a 10-hour event (Table 4.6, Fig. 4.6), contrasting with the larger peak size differences observed for 4 hour events. Differences in time-to-peak flow compared with the baseline tended to be smaller in percentage terms for longer storms, but similar in absolute terms. Riparian woodland delayed peak arrival by 18 minutes (+3.6 %) for a 10-year, 10-hour event, whereas all other 1 ha scenarios caused only very small differences in peak timing following longer storms.

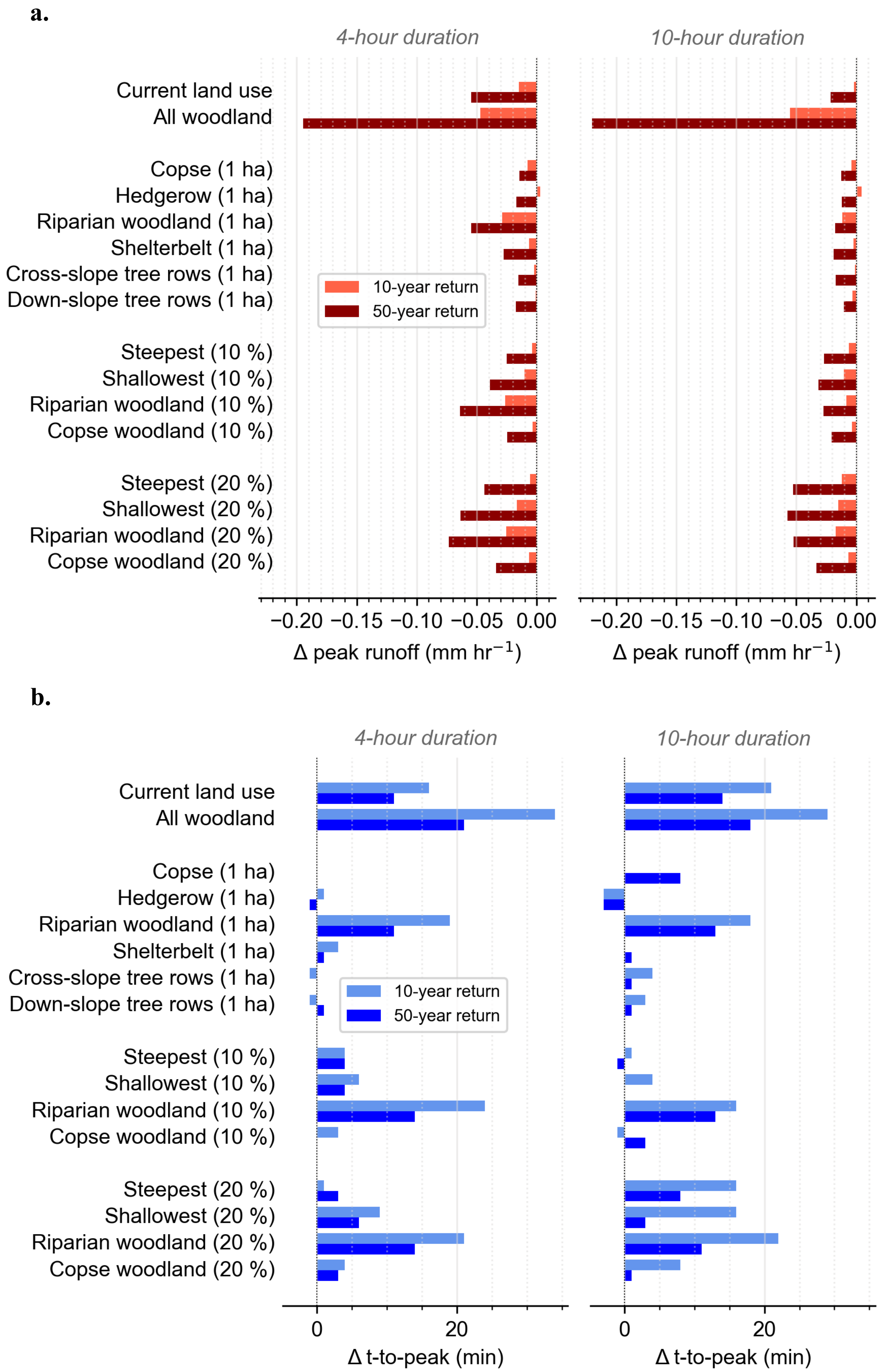
a.
4-hour duration
10-hour duration
Current land use
All woodland
Copse (1 ha)
Hedgerow (1 ha)
Riparian woodland (1 ha)
Shelterbelt (1 ha)
Cross-slope tree rows (1 ha)
Down-slope tree rows (1 ha)
Steepest (10 %)
Shallowest (10 %)
Riparian woodland (10 %)
Copse woodland (10 %)
Steepest (20 %)
Shallowest (20 %)
Riparian woodland (20 %)
Copse woodland (20 %)
10-year return
50-year return
−0.20 −0.15 −0.10 −0.05 0.00
Δ peak runoff (mm hr^{-1})
b.
4-hour duration
10-hour duration
0
20
Δ t-to-peak (min)

Figure 4.6 (preceding page) – Absolute changes in a. peak runoff and b. time-to-peak values compared with UKCEH baseline land use for all scenarios, for both 4-hour and 10-hour duration storms and 10- and 50-year return periods. 'All woodland' is a scenario in which the whole farmed area is converted to woodland. 'Steepest' and 'Shallowest' refer to the steepest and shallowest gradients within the farmed area, respectively. Percentage values in brackets indicate proportion of the farmed area converted to each land cover type. Proportional differences in time-to-peak and peak runoff are illustrated in the Supporting Information (Appendix C).

Across both storm durations and return periods, riparian woodland consistently produced the largest changes to flood peak size and time-to-peak when compared with a 1 ha area of other tree planting types, with the effect particularly pronounced for shorter, 4-hour events. Only in high-flow (50-year) and longer (10-hour) storms were differences between 1 ha scenarios less evident. Moreover, the flood hydrograph corresponding to 1 ha of riparian woodland most closely resembled the hydrograph for all combined agroforestry types (riparian woodland, cross-slope alleys, hedges, shelterbelt and copse) implying that catchment response characteristics between the two scenarios were similar. Finally, it should be noted that the difference in peak size between the whole-farm-under-woodland scenario and 1 ha riparian woodland was only a factor of 1.6 for shorter storms/lower flow, rising to 13-fold for longer/higher flow while the all-woodland scenario required a 40-fold larger area of woodland (Fig. 4.6a). Across all storm types, the all-woodland scenario produced at most a 1.9-fold longer peak time delay than the 1 ha riparian woodland scenario (Fig. 4.6b).

4.3.2. Equal-area: 10% and 20% tree cover scenarios

In general, flood peak size was more sensitive to planting scenario for shorter duration storms, but more sensitive to area coverage for longer storms (Fig. 6a). Among scenarios with 10% tree cover, riparian woodland produced the largest change in peak size and time-to-peak compared with baseline land use for 4-hour storm events (Fig. 6, Table 5). Peak runoff was reduced by 2.4 % (0.026 mm hr^{-1}) and 2.8 % (0.064 mm hr^{-1}) for 10- and 50-year events, respectively (Fig. 6a). Tree planting on the shallowest gradients on the farm produced a smaller change (10-year: –0.9 %, 50-year: –1.7 %), followed by steepest gradient planting (10-year: –0.3 %, 50-year: –1.1 %) and copse planting (10-year: –0.3 %, 50-year: +1.1 %).

The effect of each tree planting scenario on time-to-peak broadly followed the same pattern, with the largest difference observed for riparian woodland (10-year: +7.4 %, 50-year: +5.7 %), followed by trees on shallowest gradients (10-year: +1.6 %, 50-year: +2.3 %), copse (10-year: +1.1 %, 50-year: +0.6 %) and trees on steepest gradients (10-year: +0.5 %, 50-year: +0.6 %). Increasing tree area from 10% to 20% had only a small effect on time-to-peak for most of the tree planting scenarios.

At 20 % tree cover, differences in peak runoff for most scenarios following 4-hour storms only marginally increased and not commensurately with tree area. For example, peak runoff reduced by 3.2 % (0.073 mm hr^{-1}) under 20 % riparian woodland cover for a 50-year event, compared with 2.8 % (0.064 mm hr^{-1}) under 10% cover. However, for longer 10-hour storms, doubling tree area led to a doubling of peak size reduction for almost all scenarios, including riparian woodland (10 % cover: -0.027 mm hr^{-1}, 20 % cover: -0.052 mm hr^{-1}), shallowest-gradient (10%: -0.031 mm hr^{-1}, 20%: -0.057 mm hr^{-1}), and steepest-gradient planting (10%: -0.027 mm hr^{-1}, 20%: -0.053 mm hr^{-1}, all examples for 50-year return).

4.3.3. Endmember scenario

Absolute differences in flood peak size and timing between the baseline and whole-farm-under-woodland scenario were sensitive to storm intensity but not storm duration. For a 4-hour storm duration, the all-woodland scenario altered flood peak size by -0.042 mm hr^{-1} (-4.2 %) for a 10-year event, and -0.195 mm hr^{-1} (-8.4 %) for a 50-year event. These figures were –0.055 mm hr^{-1} (-3.9%) and –0.220 mm hr^{-1} (-8.0%), respectively, for a 10 hour storm. Flood peak in a 4-hour storm was delayed by 36 minutes (11.6 %) compared with baseline land use for a 10-year event, and 21 minutes (7.8%) for a 50-year event. These differences were similar for a longer, 10-hour storm. Differences in peak size between the all-woodland scenario and agroforestry scenarios were an order of magnitude smaller (1.6 fold) for shorter, lower intensity storms, rising to 13 fold for longer, more intense events. For flood peak timing, all-woodland produced at most a 1.9-fold longer delay than the best agroforestry scenarios across all storm types.

Table 4.5 – Peak runoff and time to peak values for all scenarios tested for a 4 hour storm duration with 10- and 50-year return periods. Differences are given relative to UKCEH baseline values. Dashes denote no change.

4 hour duration	10-year return period				50-year return period			
Scenario	Peak runoff (mm hr^{-1})	Diff. cf. baseline (%)	Time to peak (h)	Diff. cf. baseline (%)	Peak runoff (mm hr^{-1})	Diff. cf. baseline (%)	Time to peak (h)	Diff. cf. baseline (%)
UKCEH baseline	*1.118*	-	*4.93*	-	*2.313*	-	*4.57*	-
All current planting	1.104	-1.3	5.20	+5.5	2.258	-2.4	4.76	+4.2
All woodland	1.071	-4.2	5.50	+11.6	2.118	-8.4	4.93	+7.8
Copse (1 ha)	1.111	-0.7	4.93	-	2.298	-0.6	4.57	-
Riparian (1 ha)	1.090	-2.6	5.26	+6.6	2.258	-2.4	4.76	+4.2
Shelterbelt (1 ha)	1.112	-0.6	4.98	+1.1	2.285	-1.2	4.60	+0.6
Hedge (1 ha)	1.121	+0.3	4.96	+0.6	2.296	-0.7	4.55	-0.6
Cross-slope (1 ha)	1.116	-0.2	4.90	-0.6	2.297	-0.7	4.57	-
Down-slope (1 ha)	1.117	-0.1	4.90	-0.6	2.295	-0.8	4.60	+0.6
Steepest (10 %)	1.115	-0.3	5.01	+1.7	2.287	-1.1	4.66	+1.8
Shallowest (10 %)	1.108	-0.9	5.04	+2.2	2.273	-1.7	4.66	+1.8
Riparian (10 %)	1.092	-2.4	5.34	+8.3	2.249	-2.8	4.82	+5.4
Copse (10 %)	1.115	-0.3	4.98	+1.1	2.288	-1.1	4.57	-
Steepest (20 %)	1.113	-0.5	4.96	+0.6	2.269	-1.9	4.63	+1.2
Shallowest (20 %)	1.102	-1.5	5.09	+3.3	2.249	-2.8	4.68	+2.4
Riparian (20 %)	1.093	-2.3	5.28	+7.2	2.239	-3.2	4.82	+5.4
Copse (20 %)	1.112	-0.6	5.01	+1.7	2.279	-1.5	4.63	+1.2

Table 4.6 – Peak runoff and time to peak values for all scenarios tested for a 10 hour storm duration with 10- and 50-year return periods. Differences are given relative to UKCEH baseline values. Dashes denote no change.

10 hour duration	10-year return period				50-year return period			
Scenario	Peak runoff (mm hr^{-1})	Diff. cf. baseline (%)	Time to peak (h)	Diff. cf. baseline (%)	Peak runoff (mm hr^{-1})	Diff. cf. baseline (%)	Time to peak (h)	Diff. cf. baseline (%)
UKCEH baseline	*1.434*	-	*8.34*	-	*2.763*	-	*8.04*	-
All current planting	1.432	-0.1	8.69	+4.3	2.741	-0.8	8.28	+3.1
All woodland	1.379	-3.9	8.83	+5.9	2.542	-8	8.34	+3.7
Copse (1 ha)	1.430	-0.3	8.34	-	2.750	-0.4	8.17	+1.7
Riparian (1 ha)	1.423	-0.8	8.64	+3.6	2.745	-0.6	8.26	+2.7
Shelterbelt (1 ha)	1.432	-0.2	8.34	-	2.744	-0.7	8.07	+0.3
Hedge (1 ha)	1.439	0.3	8.28	-0.7	2.750	-0.4	7.98	-0.7
Cross-slope (1 ha)	1.433	-0.1	8.42	+1.0	2.745	-0.6	8.07	+0.3
Down-slope (1 ha)	1.431	-0.2	8.39	+0.7	2.752	-0.4	8.07	+0.3
Steepest (10 %)	1.428	-0.4	8.37	+0.3	2.736	-1	8.01	-0.3
Shallowest (10 %)	1.424	-0.7	8.42	+1.0	2.731	-1.1	8.04	-
Riparian (10 %)	1.426	-0.6	8.61	+3.3	2.735	-1	8.26	+2.7
Copse (10 %)	1.431	-0.2	8.31	-0.3	2.742	-0.7	8.09	+0.7
Steepest (20 %)	1.422	-0.8	8.61	+3.3	2.710	-1.9	8.17	+1.7
Shallowest (20 %)	1.419	-1.1	8.61	+3.3	2.705	-2.1	8.09	+0.7
Riparian (20 %)	1.417	-1.2	8.72	+4.6	2.710	-1.9	8.23	+2.4
Copse (20 %)	1.428	-0.4	8.47	+1.6	2.729	-1.2	8.07	+0.3

4.4. Discussion

Tree cover at densities sufficiently low for normal farm activity was shown to have a considerable and beneficial influence on flood peak size and timing. The capacity for trees on farms to attenuate flood peak aligns with the majority of work comparing woodland with other land-use types, in which increased interception, infiltration and water holding capacity in afforested soils contribute to smaller and delayed flood peaks (McCulloch and Robinson, 1993; Archer et al., 2013; Dadson et al., 2017; Monger et al., 2024). However, the extent of this influence was strongly determined by both storm intensity and spatial layout of tree planting, with important considerations for farm management and policy support for agroforestry and other types of farmland afforestation. Results demonstrate that even a very small area of trees on farms, correctly sited, can positively influence flood discharge in a small catchment.

4.4.1. Tree placement

For nearly all planting densities, storm return periods and durations, riparian woodland delivered the largest downstream flood risk reduction for a fixed area of planting. With just 1 ha (2.5 % of farmed area, 0.6 % of study catchment) converted to riparian woodland, flood peak size reduced by 2.6 % and arrived 19 minutes (6.6%) later for a typical storm event with a 10-year return period. With a 50-year return period these figures were 2.4 % and 4.2 %, respectively. Studies elsewhere have found this to be the case for very different catchment scales and vegetation types. For example, Gao et al. (2016) determined that flood peak attenuation in blanket peatland basins was most sensitive to vegetation cover on the peat surface in the vicinity of streams. Although the reverse effect has been found – that rougher surfaces at higher elevation produce a later flood peak than rougher surfaces nearer the stream (Huang and Lee, 2009; Maske and Jain, 2013) – Gao et al. (2016) suggest this discrepancy may be caused by differences in catchment gradients. Nonetheless, they demonstrate the same riparian area sensitivity even with a flattened DEM. Furthermore, they found, as we did, that trees on shallower slopes were more effective in attenuating flood peak than trees on steeper slopes, for an equivalent area of woodland. Mean slope in our study catchment is 4.9 %, intermediate between Gao et al. (2016) (> 9 %) and Maske and Jain (2013) (1-3 %). Monger et al. (2024), also found riparian woodland to be effective in attenuating flood peak in a steeper, upland catchment of similar scale (~2 km^2), albeit not as effective as cross-slope placement. However, our study is the first demonstration of flood peak sensitivity to riparian planting in the context of lowland agroforestry.

Riparian zone planting carries high potential for multiple ecosystem benefits in addition to flood risk reduction. Restoration of riparian areas create marginal habitat and can host watercourse restoration where streams have been modified for historic farmland reclamation. In-stream ecological heterogeneity is increased leading to increased habitat potential and riparian plant species diversity (Rohde et al., 2005; Randhir and Ekness, 2013). Riparian woodland can foster climate resilience by lowering spring and summer in-stream temperatures, with biological significance for aquatic organisms (Bowler et al., 2012; Justice et al., 2017). Where woodland is fenced, watercourses are better protected from livestock interference reducing sediment, pathogen and nutrient loading to streams, along with denitrification benefits of trees (Campbell and Allen-Diaz, 1997; Hutchins, 2012; Scott et al., 2023). Finally, whereas in-field trees such as copses, cross- or downslope tree rows can generate a trade-off through interference with use of farm equipment, riparian buffers are typically set aside from cultivated areas, potentially limiting impact on agricultural management and productivity.

The influence of linear row of tree or hedgerow placement strategies on flood peak was limited, and in some cases, isolated measures caused flood peaks to arrive earlier or be larger than for baseline land use, with either negligible or inconclusive findings. Moreover, comparison between hydrograph responses for combined current NFM planting and riparian planting alone suggest that the latter had the most significant influence on peak flow reduction. The complexity of distributed, linear features for catchment hydrology has been noted elsewhere. For example, Fiener et al. (2011) note that hydraulic functioning of linear woody vegetation is rarely modelled, and that depending on placement features can either increase or decrease hydraulic connectivity within catchments. Given the growing promotion of agroforestry, such as tree and hedgerow planting (Biffi et al., 2023; Rolo et al., 2023), the need to understand these effects in temperate catchments is urgent if agricultural landscapes are to contribute to flood risk reduction while mitigating and adapting to climate change.

4.4.2. Woodland area

Although the ‘all woodland’ scenario had the largest positive effect on discharge across storm events, our results illustrate that increasing the area of on-farm woodland to cover the whole farm does not produce a commensurate increase in flood peak attenuation compared with other agroforestry scenarios (Fig. 4.6). The importance of tree area coverage over placement for NFM increases for longer, more intense storms, and at 10 – 20 % coverage, tree area aligns more closely with peak size reduction in very intense events. However, this is not the case for a much higher rate of tree coverage (all-woodland), and a diminishing return is generally observed between increasing woodland cover and discharge response for both peak size and timing. This diminishing return is considerably stronger in the case of flood peak timing, with only a marginally increasing delay with increasing tree area.

The above findings imply that the trade-off between land lost to NFM and flood risk reduction is manageable, and especially so for shorter, more frequently-encountered extreme rainfall events. Monger et al. (2024) estimated a 3.3 % increase in flood peak attenuation for each 10 % area increase in riparian woodland up to 30% of their catchment, although the difference between each 10% step was highly nonlinear. Increasing from 0-10% riparian cover in their scenario decreased flood peak size by 2.6 %, whereas moving from 10-20% in their scenario led to a further 5.7 % drop, followed by an additional 1.6 % drop from 20-30 %. Depending on storm duration and return frequency we observe a 0.6 – 2.8 % drop for 10 % riparian cover, increasing to 1.2 – 3.2 % for 20 % cover within the farm boundary alone. Buechel et al. (2022) estimated a 1.4 ± 0.6 % reduction in the top 1% of flows for each 10% increase in woodland cover for large UK catchments (500-10,000 km^2), although they found weaker correlation between increasing tree area and decreasing flow in the case of high flows. We found woodland planting over the whole farm area (25 % of the catchment) decreased flow peak by just 1.6 times more (4.2 %) than 1 ha of riparian woodland (2.6 %) at lower flow, rising to 13 times more at the highest flows, while requiring 40 times more catchment area taken out of production. As Monger et al. (2024) note, the effect of riparian woodland on flood peak magnitude and timing may also be underestimated because of in-stream debris further attenuating flow (Thomas and Nisbet, 2012; Cooper et al., 2021), which present modelling does not account for. These findings are significant in demonstrating the importance of tree siting considerations for optimal NFM benefit at lower area coverage and for shorter storm durations, as opposed to a focus on maximising woodland area alone. Furthermore, spatially distributed hydrological modelling should be used to support the design of future agroforestry schemes in order to maximise potential for flood risk reduction.

4.4.3. Scaling findings

As demonstrated here and elsewhere, planting for NFM in a small catchment can have positive local effects on flood peak magnitude and timing (Marshall *et al.*, 2014; Dadson *et al.*, 2017). Yet although our findings show that small amounts of farm woodland can substantially influence catchment hydrology, these results are for a small catchment (175 ha) in which tree planting at a single farm representing 25% of catchment area has significant impact on whole-catchment hydrological functioning. Over a wider area, catchment hydrology will depend on a larger and more complex network of sub-catchments and land holdings, and understanding of spatial connectivity of flow processes across networks such as these remains poor (Rogger et al., 2017).

Studies have attempted to address this empirically, such as at Pontbren in Wales where extensive sampling and instrumentation over a ~10 km^2 catchment were used to examine changes in overland flow generation and infiltration under woodland and intensively grazed pasture (Wheater et al., 2012; Marshall et al., 2014). Overland flow generation in the vicinity of woodland was reduced by as much as two thirds at Pontbren, mirroring the local effect of trees in our own modelling. However, verifying effects such as these for different scales and landscape features is complicated by the logistical challenge of undertaking large empirical studies, along with complications in transferring results between differing catchment geographies. As we show, this is likely to be the case for linear features such as hedgerows or rows of trees; two common types of agroforestry system.

Modelling at larger scales is therefore needed to verify whether a flood peak reduction within a smaller catchment delivers downstream flood peak reduction at scale, or conversely whether the delayed peak from the small catchment results in flood peak synchronicity (and peak increase) in downstream, larger river channels. Additionally, modelling studies considering wider catchment processes and longer-term responses beyond single storm events would be welcome. Anecdotal evidence exists for positive wider-scale benefits of NFM interventions (e.g. Nisbet, 2017), yet little is known about underlying mechanisms or complex interactions of summing local effects over whole landscapes. Nonetheless, if our findings hold for small catchments elsewhere, we believe it is likely that flood benefits can be realised at larger scales.

4.5. Conclusions

Using SD-TOPMODEL we analysed changes to rainfall-runoff characteristics following various agroforestry planting scenarios in a 1.75 km^2 lowland catchment in Yorkshire, UK. We have shown that using very small areas of agricultural land for tree planting can have substantial benefit for flood peaks, providing woodland is appropriately sited. Riparian woodland covering just 2.5 % of farmed area, or 0.6 % of the catchment, was capable of attenuating flood peak by 2.6 % and delaying its arrival by 6.6 %, or 19 minutes. The effect of an equivalent area of shelterbelt or in-field copse planting was much more modest, and linear features such as in-field tree rows or hedges had a negligible or opposing effect on flood peak size and timing, reflecting the complexity of their hydrological characteristics. More work is needed to understand these effects considering the growing promotion of in-field agroforestry schemes and trees on farms (IPCC, 2014a; CCC, 2020; Woodland Trust, 2022; Defra, 2025). Increasing the area of trees produced a diminishing return in terms of changes to the flood peak, and hypothetical woodland cover over the whole farmed area (25 % of the catchment) produced only 1.6 times the peak size reduction as 0.6 % riparian cover in low flow scenarios, rising to 13 times at the highest flows, despite requiring 40 times more land taken out of production.

We therefore conclude that if sited on shallow gradients or in the riparian zone, agroforestry can significantly contribute to NFM without substantially impacting land available for productive agriculture. Integrated policy support that connects agroforestry to hydrological benefits would be welcome in the future and broader benefits could also be studied (such as reduced soil erosion and transport) so that policy incentives consider the full range of ‘stacked’ environmental benefits. These findings are significant in demonstrating the greater importance of tree siting considerations over coverage rates in lowland NFM, and the capacity of agroforestry to deliver multiple landscape benefits.

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Chapter 5. Synthesis

5.1. Chapter outline

This final chapter synthesises the research findings of chapters 2 – 4. In section 5.2, I briefly summarise key findings from each chapter, before integrating findings in more detail into themes outlined in the Introduction in section 5.3. In section 5.4, I discuss applicability of findings to wider stakeholder issues, including policy- and practitioner-focussed viewpoints. Section 5.5 outlines limitations of the study and priorities for further research, before I draw some overall conclusions in section 5.6.

5.2. Summary of key findings

The aim of this thesis was to determine spatial effects of four agroforestry design and context factors – generic tree planting ('trees'), tree species choice, tree placement and landscape – on three key soil outcome categories in temperate settings. Responding to research needs identified from the literature and practice, and to capture different stakeholder perspectives I differentiated between spatial scales across which benefits and trade-offs are found, and how outcomes are controlled by design choices and contextual constraints faced by practitioners. Three lateral scales of observation were considered – plot, field and catchment. In the process, I sought to show how benefits and trade-offs can be superimposed in the landscape. Three research questions were considered, focussed on the role of agroforestry in temperate ecosystems, originally outlined in section 1.5:

RQ 1. What is the relative influence of trees, tree species choice and random pre-planting soil variability on common indicators of soil function at plot scale?

RQ 2. What is the relative influence of trees, hillslope and alley width on common indicators of soil function at field scale?

RQ 3. What is the relative influence of trees and on-farm tree placement on soil hydrology and natural flood management potential at catchment scale?

Chapters 2 – 4 dealt with each of these questions in turn. Addressing the RQ1, Chapter 2 used a field sampling and measurement approach at a site in Yorkshire to consider the sensitivity of three arable soil depth horizons – forest floor, topsoil (0-30 cm) and subsoil (>30 cm) – to three contextual factors driving soil function, each of which are pertinent to planters of agroforestry. I found that tree species choice most strongly influenced soil functioning in the forest floor litter and in topsoil, and that the effect was strongest for nutrient dynamics; that general agroforestry planting (land-use change) most strongly influenced topsoil and that the effect was strongest for soil structure and C storage; and that in subsoil (below 30 cm) random pre-planting variability dominated the effects of trees or tree species on soil functioning. Each soil depth therefore had its own functioning 'signature' in response to agroforestry.

For RQ2, Chapter 3 used a field sampling approach at a site in Devon to consider how distribution of soil and ecosystem benefits in a silvoarable system are controlled by hillslope and alley width between rows of trees. A number of key findings emerged, which I will summarise here. Benefits from tree rows readily extend into cropped alley areas, which were 8.8% less compacted and contained 40% more plant-available phosphate than a treeless control. Tree rows mitigated soil loss despite being planted parallel to slope. Trees and crops competed for N, P and moisture in subsoil at the boundary between tree rows and cropped alleys. Considering alley width, fertility benefits traded-off with distance between tree rows, with more N and P stored in a 12 m alley width system compared with a 24 m wide alley system. Although I found that additional C was stored in soil beneath agroforestry (700 kg C ha^{-1} $year^{-1}$) at the study site, this was only a fraction of the target C storage for silvoarable agroforestry in the UK (8 t C ha^{-1} $year^{-1}$), meaning that potential for silvoarable systems to contribute to national C budgets may have been overestimated. This also confirms that agroforestry must be considered in terms of multiple benefits that go beyond C sequestration, such as soil fertility benefit, natural flood management and biodiversity improvements.

Addressing RQ3 in Chapter 4, I used a flood-modelling approach to assess rainfall-runoff characteristics following a range of agroforestry planting scenarios in a 1.75 km^2 lowland catchment in Yorkshire, UK. It demonstrated that even very small areas of on-farm woodland can have a meaningful effect on flood peak size and timing, provided the trees are appropriately situated within the landscape. Riparian woodland covering just 2.5 % of farmed area, or 0.6 % of the whole catchment, was capable of attenuating flood peak by 2.6 % and delaying its arrival by 6.6 %, or 19 minutes. The effect of an equivalent area of shelterbelt or in-field copse planting on flood peak size and timing was considerably more modest, and linear features such as in-field tree rows or hedges had a negligible or opposing effect on flood peak size and timing, reflecting the complexity of their hydrological characteristics. Increasing the area of trees produced a diminishing return in terms of changes to the flood peak, particularly for shorter and less intense storms. Hypothetical woodland cover over the whole farmed area (25 % of the catchment) produced 1.6 times the peak size reduction in lower flow scenarios, rising to a 13-fold greater reduction at higher flows compared with 0.6 % catchment riparian cover, despite the former requiring 40 times more land taken out of production.

5.3. Integrating key findings

There has previously been a lack of research on temperate agroforestry which integrates results across soil C, soil nutrient and soil water domains (Cardinael et al., 2020). It is therefore fitting to conclude by considering these domains in combination as *multiple benefits*, viewing benefits through the lens of the four design and context factors outlined in the Introduction and summarised in Table 1.1. Effectively these can be thought of as four simple questions, each of which are answered across the research chapters of the thesis –

- How does simply planting trees for agroforestry (in any configuration) influence soil outcomes?
- How does tree species choice influence soil outcomes?
- How does tree placement influence soil outcomes?
- How does configuration of tree planting within the landscape influence soil outcomes?

The following discussion deals with each of these in turn, summarising relevant findings across the three research studies.

5.3.1. Planting trees

Considered as a whole, planting trees for agroforestry produced mixed soil outcomes across the three studies. Benefits were realised in some cases, but their horizontal and vertical spatial extent was highly heterogeneous, demonstrating the importance of design, context and scale in evaluating outcomes.

Overall, SOC stock changes observed in this study were small compared with results from meta-analyses extending to a similar soil depth (De Stefano and Jacobson, 2018; Shi et al., 2018), including those focussed on temperate zones (Mayer et al., 2022). There was no significant difference between woodland and treeless control areas at 0-50 cm depth at Leeds University Farm in Chapter 2, nor in the younger silvoarable system with tree rows spaced at 24 m in Chapter 3 at Shillingford in Devon. Only in the older, 12 m Devon tree rows did I find an 18% increase in SOC stock over 21 years between the agroforestry system and treeless control, corresponding to soil C additions of 700 kg C ha^{-1} $year^{-1}$. Variability in the potential for agroforestry to result in SOC stock changes is well documented (e.g. Mayer et al., 2022), yet this study reveals how variable these changes can be in response to planting, even with near-identical planting and management conditions. As ever, soil depth was a critical factor in determining whether SOC stock differences were positive, negligible or in some horizons, negative between agroforestry and treeless areas. Although topsoil (0-30 cm) in Chapter 2 responded to tree planting with a significant, positive change in SOC stock (assuming the SOC was similar to that of adjacent arable fields before trees were planted), this was not reflected at 0-50 cm depth. These findings are in contrast with those presented in meta-analyses such as that conducted by De Stefano and Jacobson (2018), in which trees planted on arable land were found to produce a positive change in SOC stock at both 0-30 and 0-100 cm depths, although soils at the study site in Chapter 2 were shallower, at 50 cm deep. Similarly in Chapter 3, at 0-20 cm depth, an increase in SOC stock was found in both agroforestry systems relative to treeless arable controls, whereas at 0-50 cm depth the only significant difference was found in the 12 m spaced tree system. Based on these findings, simply planting trees on arable land cannot be assumed to lead to extra C storage in farm soils to depth.

Planting trees was beneficial for overall fertility (available N and P), despite finer scale spatial and depth heterogeneities. Fertility could not be effectively compared between systems in Chapter 2 as the control areas received synthetic fertiliser additions, whereas the tree areas did not. However, all systems were organic in Chapter 3, meaning stocks could reliably be compared between each treatment. Not only were higher stocks of N and P at 0-50 cm depth found in both the 12 and 24 m agroforestry systems, fertility per unit area in the cropped alleys within the agroforestry systems were higher than the equivalent area in the treeless control, despite all systems being identically managed. The importance of this is that the area of cultivatable land lost in the tree rows has potential to be offset by extra fertility in the alleys between the tree rows.

Hydrological outcomes were similar for the field-scale agroforestry systems and the treeless control area in Chapter 3. Point-in-time moisture did not significantly differ overall between the agroforestry and control areas, and there was no mean difference in saturated hydraulic conductivity between the two systems. In Chapter 4, agroforestry systems were shown, through modelling, to influence NFM potential, generating appreciable effects on flood peak size and timing, and aligning with the findings of other studies (McCulloch and Robinson, 1993; Archer et al., 2013; Dadson et al., 2017; Monger et al., 2024). Yet for shorter, less intense, storms the size of this effect was strongly determined by the spatial layout of trees, with riparian planting by far the most effective strategy. ‘Planting trees’ alone as a factor (with no spatial information) was only sufficient in determining potential benefits for NFM from agroforestry in more extreme, rarely-encountered storm events. I therefore discuss these outcomes in more detail in sections 5.3.3 and 5.3.4, below.

5.3.2. Tree species selection

Including the second consideration – which trees to plant – added nuance to the question of outcomes. Tree species choice had an important influence on SOC and nutrient dynamics under agroforestry. The three species examined – Hazel, Cherry and Sycamore – produced similar quantities of litter at the forest floor. However, as Gao et al. (2014) found for a temperate system in China, litter C stock varied by species: Hazel litter introduced 1.5 times more C stock to the soil surface compared with the other two species. Tree species choice is therefore an important control on the nature of litter C additions to forest floor and surface soil in agroforestry systems. Similarly, nutrient content of litter varied by species, with less P content in Hazel litter, and less N content in Sycamore litter. Unlike differences in litter C between species, which were not reflected in surface soil SOC differences between species, litter nutrient content differences between species corresponded to those in surface soil. This mirrored findings from tropical studies such as the Nicaraguan system of Hoosbeek et al. (2018), in which topsoil beneath Jícaro (*Crescentia alata*) trees had higher N content, and topsoil beneath Guácimo (*Guazuma ulmifolia*) trees developed higher P content. Nutrient dynamics derived from interspecific tree litter additions are critical not just for surface soil fertility but also for controlling topsoil decomposition via control over the C:N ratio of surface soil (Amorim et al., 2022). Although these differences were not resolvable at the study site in Chapter 2, they are likely to be significant across other species and have been demonstrated elsewhere (Cools et al., 2014). Soil moisture and infiltration differences related to tree species were also not observed at the site in Chapter 2, but require further study as they will have implications for functioning if found in other tree species combinations for agroforestry.

This study therefore demonstrates tree species control over soil outcomes in temperate agroforestry, particularly for C and nutrients, albeit a control which was found only for topsoil and the forest floor. However, I only examined three hardwood timber species, and others such as leguminous trees and shrubs are known to influence N dynamics in deeper soil horizons (Zhang et al., 2008; Cardinael et al., 2020). In addition to leguminous species, much more evidence linking other tree species with specific soil outcomes is needed in order to support practitioner design choices. Although increasingly comprehensive guides have been published relating tree species choices with above-ground outcomes, management or economic considerations (e.g. Soil Association, 2019; Staton et al., 2024), these lack evidence for soil outcomes which are themselves critical in determining how ecosystem services are transferred from agroforestry to other parts of the landscape.

Tree species choice is also very likely to control NFM potential for agroforestry due to varying root and litter characteristics and associated effects on surface rugosity. Although outside the scope of this project, varying effects of different tree and shrub species associations on hydrological parameters have been measured in upland contexts (Monger et al., 2022), and a similar differentiation between possible lowland agroforestry land cover types would be a useful subject for further work. I mention the need for more work assessing control of different tree species associations on soil outcomes in Section 5.5, below.

5.3.3. Tree placement

The choice of where to plant trees relative to one another was of substantial importance for soil outcomes. In Chapter 3, as might be expected, for both soil C and nutrients, a closer spacing of tree rows (12 m, 110 stems ha^{-1}) favoured higher soil stocks, and therefore storage, within the agroforestry system as a whole, compared with wider rows (24 m, 55 stems ha^{-1}). Higher tree density leads to higher C and nutrient inputs of root and shoot litter and exudates from the perennial components of the system, per unit area (Jobbágy et al., 2001; Haichar et al., 2014; Steinfeld et al., 2024); denser trees produce greater understorey coverage which is important for OC inputs (Upson et al., 2016; Cardinael et al., 2018); all confirming the assertion of De Stefano and Jacobson (2018) that greater ecological complexity produces higher SOC stocks in agroecosystems. In Chapter 3, nutrient stocks (normalised for tree age) were significantly enhanced in narrower tree rows, with denser root and associated arbuscular mycorrhizal (AM) fungal networks allowing better circularity and resource capture within the system and reducing nutrient loss to leaching (Ingleby et al., 2007; Cardinael et al., 2015). More extensive tree root networks draw nutrients from depths previously inaccessible to shallower-rooting crops (Zhang et al., 2008). As Cardinael et al. (2020) noted, many of these results are well-known in tropical zones, but it is timely to have examples from temperate zones, particularly for practitioners to consider the influence of alley width choice on a range of ecosystem services. Although significant competition for N was observed in subsoil at the alley edge in Chapter 3 in line with findings elsewhere (Jose et al., 2000b; Zamora et al., 2009), the 12 m agroforestry system overall contained higher N and P stocks than the adjacent, treeless control. Similar edge effects in more nutrient-limited areas can also be offset using more sequential tree crop interactions (Cardinael et al., 2020).

Tree placement was also significant for soil water. Competition for moisture at the row-alley boundary was observed as for nutrients, with significant depletion in alley-edge subsoil where tree and crop roots are most likely to compete for resources (Jose et al., 2000a). Additionally, infiltration beneath the tree rows was considerably faster, affecting water storage capacity across the system. These effects are likely to be more pronounced where trees are closely spaced, with implications for practitioners in areas of greater water scarcity.

The greatest influence of tree placement on water outcomes was found in Chapter 4 at the catchment scale. The potential for an equivalent area of trees to offer NFM as a benefit was highly dependent on configuration of tree placement and location within the landscape, with agroforestry in riparian zones delivering a much larger downstream flood risk reduction than traditional, linear features such as tree rows within fields. Others have found riparian planting to produce greater flood benefit for an equivalent area (Gao et al., 2016), however my study is the first to confirm the effect for agroforestry, even at very low coverage rates (< 1% of catchment). Crucially, a diminishing return was found for extra tree coverage. Substantially increasing coverage from < 1 % to 25% of the catchment produced only 1.6-fold flood peak size reduction compared with < 1 % riparian woodland at low flows. Although for higher flow scenarios (more critical in the context of NFM) this rose by an order of magnitude to 13-fold, the former necessitates 40 times more land area excluded from production. Placement and configuration of trees is therefore a stronger determinant than coverage in delivering potential for NFM.

5.3.4. The role of landscape

Landscape and topography factors add extra context and detail to the question of where trees should be situated relative to one another. In Chapter 3, hillslope position was shown to have a significant effect on several soil outcomes. In the treeless control area, a higher clay fraction and SOM content were found downslope, whereas in the presence of trees this was not found to be the case, implying some mitigation of downslope soil transport by the presence of agroforestry. Water erosion selectively transports finer particles of topsoil downslope, and given the majority of SOM is stored in topsoil, it is also transported downslope as a result of erosion and commonly lost to watercourses (Durán Zuazo and Rodríguez Pleguezuelo, 2008). The extent to which agroforestry can deliver benefit in the form of erosion mitigation is therefore mediated to an extent by hillslope.

Others have contended that planting cross-slope rows is most beneficial for mitigating soil loss to erosion, yet this study implies that even tree rows parallel to the slope can deliver a meaningful erosion reduction benefit. Intuitively this is not surprising – tree roots and litterfall increase surface roughness, promote aggregate stability and soil porosity (Liu et al., 2016), such that water both within but also adjacent to tree rows can be slowed and stored more effectively, reducing erosion susceptibility (Battany and Grismer, 2000). To corroborate this, I found higher water storage upslope within tree rows than in treeless control areas, implying better storage of water within the landscape. However, no demonstrations of slope-parallel agroforestry tree rows which mitigate erosion have been directly made, and my findings therefore merit further exploration, potentially yielding a useful result for practitioners for whom cross-slope rows create significant challenges with operating machinery. The effect can be further augmented with careful management of the understorey beneath trees, along with incorporation of cover crops (Dabney et al., 2001), although these were not explicitly considered in this study.

Finally, for landscape consideration, based on findings from Chapter 2, is the presence of random, pre-existing variability in soil properties which can obscure the effects of agroforestry, even decades after planting. For example, of the four notionally identical plots sampled in Chapter 2, one recorded 18.7 % higher SOC stock over 35 years compared with a treeless control, whereas a second recoded 15.3 % lower SOC stock, despite both being on the same soil type and topography and having identical tree species, placement and management. These discrepancies originated in the subsoil (>30 cm) and raise an important consideration, which is that pre-existing conditions can influence outcomes even when there is a very high degree of effect from other drivers on the system.

5.4. Implications for policy and practice

As encountered in many agroecological contexts (Reith et al., 2022), the question of tree placement presents trade-offs for practitioners and landscape planners, for which optimal solutions will depend on objectives for planting. For example, on steeper slopes, planting agroforestry tree rows parallel to slope appeared to store moisture higher up the slope more effectively and limit downstream transport and potential loss of fine soil, available N and organic matter. However, planting on steeper slopes has been shown here and by others (Gao et al., 2016; Monger et al., 2022) to offer limited catchment-scale NFM benefit. If siting a limited area of agroforestry, the best solution will therefore depend on the relative value of limiting local erosion susceptibility or delivering catchment scale flood risk reduction. A third consideration in choosing where to site trees is the extra nutrient facilitation benefit for crops in the alleys between tree rows, which was found to be significant where tree rows were closer to one another (12 m spacing). At plot scale, as illustrated in Chapter 2, woodland was not capable of increasing SOC storage to 50 cm depth over 35 years, yet it was readily capable of promoting surface soil nutrient addition in the vicinity of trees, as were younger trees in the 12 m silvoarable system of Chapter 3. These trade-offs illustrate the importance of capturing stakeholder value and appreciating context when designing support for agroforestry.

The importance of a multiple benefit focus for agroforestry emerged from all three studies. This was particularly the case in Chapters 2 and 3, in which potential SOC storage in agroforestry was shown to be limited compared with stated targets for and contributions to net zero (CCC, 2020; Woodland Trust, 2022). Recognising complexity and avoiding linear solutions is also a theme of the key findings presented in this thesis: for example, in the context of NFM and Chapter 4, focussing on siting trees 'correctly' within the landscape was shown to deliver considerably greater economic and environmental co-benefit, compared with targets for tree planting which simply focus on maximising planted area (e.g. Welsh Government, 2024) and reducing land available for productive agriculture. Considering 'stacked' benefits of agroforestry also improves its potential as a climate adaptation solution, in which benefits such as water retention, shading for livestock and greater agrobiodiversity improve farm resilience against future environmental stresses.

Limited understanding of benefits is a known issue preventing scaling of agroforestry (Sollen-Norrlin et al., 2020). As an example, if temperate agroforestry cannot reliably deliver appreciable soil C benefits in line with estimates across a wide range of contexts, its value must instead be communicated for the many other benefits it offers to practitioners if they are to incorporate trees: nutrient facilitation, NFM and protection against soil loss, in addition to other known benefits not considered here such as soil microbial diversity (Judson et al., 2023) and ecological connectivity (Haggar et al., 2019; Vagge et al., 2024). Applications such as C trading or offsetting are not appropriate if a C benefit cannot be reliably delivered and outcomes differ so widely over even short length scales, as this research implies. Multiple benefits of agroforestry and the interactions between them must be more readily considered in research, and communicated and valorised for practitioners.

5.5. Limitations and priorities for further work

The lack of long-term temperate agroforestry sites available for research constrain which problems can be addressed within the scope of the thesis. Research focus is therefore necessarily guided by availability of sites. Nonetheless, site choice can be guided to consider the research questions presented, and I have sought to make use of available agroforestry systems of sufficient age to examine differences in outcomes (e.g. Smith, 2004) in a manner which benefits research and practitioner decision-making. Some compromises were necessary. For example, in Chapter 3 I make use of a site that is a working farm and not set up as an agroforestry experiment, in order to take advantage of a unique planting layout in which alley width and hillslope effects can be considered. As a working farm, not all effects can be isolated from one another as would be possible at a dedicated experimental site, and organic management across the site also provides some limitation to the applicability of results in other systems. However, the lack of research and alternative sites addressing these controls give the findings considerable relevance, and further work can usefully build on the methods introduced.

Given the low uptake of agroforestry in temperate areas, research to date has predominantly focussed on the tropics. Temperate studies are far less widespread and many interdependencies between controlling factors and outcomes remain unaddressed in both the literature and the scope of this thesis. To name just a select few: the question of soil type, benefit accumulation over time, seasonality, management and a greater number of tree species and associations. Other useful outcomes not addressed are the role agroforestry can play in improving water quality, improving soil biological functioning and effects of agroforestry design strategies on crop yield in silvoarable alleys between trees, particularly in response to described fertility benefits. The findings I generated are associated only with the study sites in question and further work is required to ensure they have broader transferability elsewhere, including in deeper and more varied soils; I did not consider soil depths below 50 cm; the first two sites are silvoarable and pasture is not sampled; only three tree species are considered; and the catchment studied at the final site only represents effects at a single scale.

Another major component receiving little or no attention within scope is the crucial role of understorey vegetation beneath trees for C storage (Upson et al., 2016; Cardinael et al., 2018), nutrient facilitation (Battie-Laclau et al., 2019) and hydrological functioning

(Anderson et al., 2009; Monger et al., 2022). Although I acknowledge the likely role of the understorey mediating a number of processes across the research chapters of this thesis, it is not directly studied, and future work elaborating on the processes outlined here should focus on the role of understorey vegetation.

Finally, broadening the scope of these findings to include other interdependencies between agroforestry design and context factors and outcomes can only be enhanced through greater availability of study data from sites elsewhere. This would ease analysis of factors not addressed here such as soil type, tree species, understorey effects and alternative planting scales and geometries, in addition to supporting wider meta-analyses of controls on agroforestry outcomes. The approach used here is intended to be a contribution to previously limited agroforestry knowledge in temperate zones, such that predictability of outcomes can be improved for practitioners and policymakers. Future work will almost certainly focus on other interdependencies, further strengthening the case for agroforestry and the multiple landscape benefits it offers.

5.6. Conclusions

This thesis investigated how key soil and ecosystem benefits from agroforestry are distributed in temperate landscapes in the vicinity of trees. I sought to strengthen understanding of linkages between four contextual/design factors: planting trees, tree species selection, tree placement and landscape and topographic context; and three soil outcome domains: soil C, soil nutrients and soil water, in order to improve practitioner understanding of agroforestry benefits. In the process, I demonstrated distribution and superposition of benefits and trade-offs in the landscape, and describe spatial effects of the design and contextual factors under study.

A focus on simply planting trees was of limited use for understanding soil benefits across the three domains. Although some findings emerged, such as greater overall nutrient stocks within agroforestry compared with treeless areas and stronger NFM response to tree coverage at higher storm intensities, a spatially-explicit view of controls was needed for benefits to be more fully characterised. Species choice was an important control at plot-scale, with influence over soil outcomes particularly in the forest floor and surface soil horizons. Tree species is therefore an important consideration for stakeholders focussed on benefits within the uppermost soil horizons, such as farmers looking to support fertility needs for cultivation. Deeper in the soil profile tree species was not observed to have an influence on outcomes, although we note the limited number and functional diversity of species examined and the need to consider others. Tree placement, meanwhile, was of greatest importance in controlling benefit delivery across the three domains, and at both field and catchment scale. Narrower alleys produced significant soil C storage and nutrient facilitation benefit, in addition to increased hydrological functioning, whereas planting in riparian areas delivered significant NFM benefits with only a very limited area of trees. A number of trade-offs between placement priorities were described, further emphasising the importance of stakeholder priorities for determining appropriate planting strategies. Finally, landscape and topographic context interacted with tree placement to moderate benefit delivery, with hillslope position controlling the extent to which agroforestry influenced erosion susceptibility.

The results contribute to an area of much-needed research in agroforestry, namely, practitioner-focussed controls on multiple soil benefits and how they are applied in temperate contexts. Establishing support which recognises spatial heterogeneity of outcomes alongside contextual and stakeholder differences is critical for agroforestry to become an attractive land use option, adapting agroecosystems to a changing climate and delivering sustainable public goods in the landscape.

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Appendix A. Supporting Information for Chapter 2

Supplementary Table A.1 – Mean results with standard error by treatment type for litter (L) and fermented and humus (F+H) combined horizons for all measured soil indicators, including values by species and by plot number. Significant differences denoted by letters (note that significant differences are calculated within dotted boxes only); ns – not significant.

			Biomass			**Total N**			**NO_3-N**			**Plant-available N***			**Total P**			**Organic C**					**C:N**	
Layer	**Treatment**	n	t ha^{-1}	Sig	n	g kg^{-1}	Sig.	n	mg kg^{-1}	Sig.	n	mg kg^{-1}	Sig.	n	g kg^{-1}	Sig.	n	%	Sig	t ha^{-1}	Sig.	n	ratio	Sig.
L	*All woodland*	*24*	*9.96 ± 0.79*		*24*	*9.92 ± 0.41*								*12*	*1.18 ± 0.06*		*24*	*33.4 ± 1.1*		*3.32 ± 0.28*		*24*	*34.7 ± 1.6*	
	Cherry	8	8.78 ± 0.80	ns	8	9.01 ± 0.63	ns							4	1.33 ± 0.06	a	8	31.7 ± 2.1	ns	2.78 ± 0.35	ns	8	37.0 ± 4.6	ns
	Hazel	8	11.60 ± 1.98	ns	8	10.31 ± 0.78	ns							4	0.95 ± 0.06	b	8	33.4 ± 2.2	ns	3.81 ± 0.60	ns	8	33.0 ± 1.6	ns
	Sycamore	8	9.51 ± 0.99	ns	8	10.45 ± 0.67	ns							4	1.26 ± 0.09	a	8	35.1 ± 1.7	ns	3.36 ± 0.44	ns	8	34.0 ± 1.4	ns
	Plot 1	6	9.67 ± 1.11	ns	6	10.14 ± 1.20	ns							3	1.16 ± 0.04	ns	6	36.1 ± 3.3	ns	3.53 ± 0.62	ns	6	37.3 ± 4.8	ab
	Plot 2	6	8.36 ± 0.58	ns	6	9.33 ± 0.42	ns							3	1.15 ± 0.17	ns	6	29.8 ± 1.4	ns	2.47 ± 0.16	ns	6	32.3 ± 2.0	ns
	Plot 3	6	10.16 ± 1.95	ns	6	10.99 ± 0.65	ns							3	1.23 ± 0.17	ns	6	33.4 ± 1.5	ns	3.35 ± 0.58	ns	6	30.7 ± 1.5	a
	Plot 4	6	11.65 ± 2.23	ns	6	9.23 ± 0.77	ns							3	1.18 ± 0.16	ns	6	34.5 ± 2.0	ns	3.91 ± 0.66	ns	6	38.5 ± 3.5	b
F+H	*All woodland*	*24*	*85.1 ± 5.0*		*12*	*5.00 ± 0.28*		*12*	*5.57 ± 0.67*		*12*	*5.57 ± 0.67*		*12*	*1.03 ± 0.05*		*12*	*6.34 ± 0.49*		*5.61 ± 0.38*		*12*	*12.6 ± 0.4*	
	Cherry	8	94.0 ± 8.6	ns	8	4.66 ± 0.49	ns	4	6.57 ± 0.90	ns	4	6.57 ± 0.90	a	4	1.07 ± 0.08	ns	8	5.98 ± 0.63	ns	5.46 ± 0.54	ns	8	12.9 ± 0.7	ab
	Hazel	8	86.4 ± 9.9	ns	8	5.28 ± 0.66	ns	4	6.44 ± 1.48	ns	4	6.44 ± 1.48	ab	4	0.92 ± 0.07	ns	8	7.30 ± 1.28	ns	6.53 ± 0.79	ns	8	11.4 ± 0.3	a
	Sycamore	8	74.8 ± 6.8	ns	8	5.06 ± 0.37	ns	4	3.70 ± 0.52	ns	4	3.70 ± 0.52	b	4	1.11 ± 0.08	ns	8	5.73 ± 0.37	ns	4.84 ± 0.46	ns	8	13.6 ± 0.7	b
	Plot 1	6	66.9 ± 8.3	a	6	6.06 ± 0.46	a	3	6.63 ± 1.49	ns	3	6.63 ± 1.49	ns	3	1.13 ± 0.07	a	6	7.77 ± 1.49	a	6.30 ± 1.26	ab	6	12.6 ± 1.4	ns
	Plot 2	6	111.3 ± 8.7	b	6	3.84 ± 0.30	b	3	3.24 ± 0.52	ns	3	3.24 ± 0.52	ns	3	0.82 ± 0.04	b	6	4.68 ± 0.24	b	4.87 ± 0.42	a	6	12.2 ± 0.3	ns
	Plot 3	6	79.6 ± 6.5	a	6	4.66 ± 0.20	b	3	5.10 ± 0.86	ns	3	5.10 ± 0.86	ns	3	1.04 ± 0.10	ab	6	6.07 ± 0.50	a	4.98 ± 0.70	ab	6	13.1 ± 1.2	ns
	Plot 4	6	82.4 ± 7.6	a	6	5.44 ± 0.16	a	3	7.32 ± 1.37	ns	3	7.32 ± 1.37	ns	3	1.13 ± 0.04	a	6	6.83 ± 0.45	a	6.28 ± 0.24	b	6	12.5 ± 0.5	ns

* NO_3-N + NH_4-N

Supplementary Table A.2 – Mean results with standard error by depth interval and treatment type for bulk density (corrected for root and stone content) and ESM-corrected moisture, SOM, SOC (concentration and stock) and C:N. Significant differences denoted by letters (note that significant differences are calculated within dotted boxes only); ns – not significant.

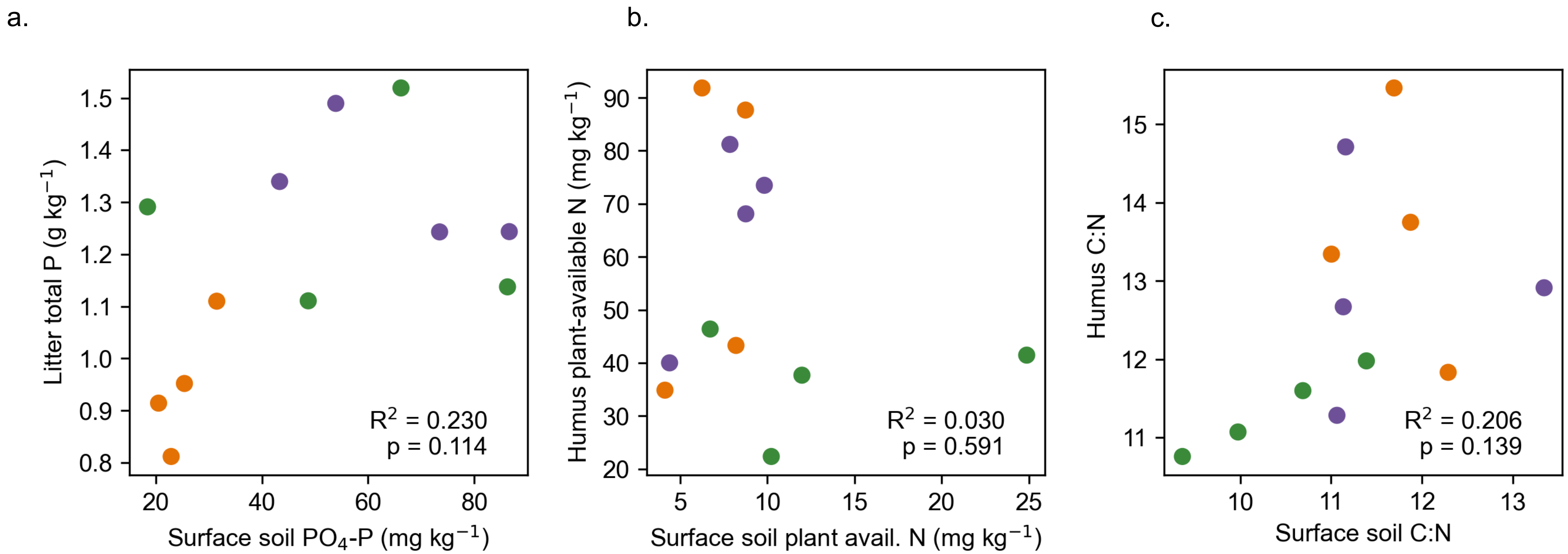
			BD			**Moisture**			**SOM**			**SOC**						**C:N**		
Depth	**Treatment**	***n****	g cm^{-3}	Sig.		%	Sig.		%	Sig.		%	Sig.		cum. t ha^{-1}†	Sig.		ratio	Sig.	
0 - 10 cm	*Control (arable)*	8	1.46 ± 0.04	b	a	24.1 ± 3.5	a	ns	6.82 ± 0.62	a	a	1.91 ± 0.06	a	a	28.0 ± 1.0	a	a	9.95 ± 0.36	a	a
	All woodland	24	1.28 ± 0.02	a		31.4 ± 0.9	b		8.12 ± 0.23	b		3.05 ± 0.11	b		45.3 ± 1.7	b		11.25 ± 0.23	b	
	Cherry	8	1.29 ± 0.02		b	31.7 ± 1.6		ns	8.35 ± 0.32		b	3.22 ± 0.14		b	48.0 ± 2.3		b	11.68 ± 0.42		b
	Hazel	8	1.29 ± 0.03		b	30.8 ± 1.4		ns	7.97 ± 0.44		ab	3.02 ± 0.16		b	44.7 ± 2.5		b	11.71 ± 0.20		b
	Sycamore	8	1.26 ± 0.03		b	31.6 ± 2.0		ns	8.04 ± 0.47		ab	2.90 ± 0.24		b	43.1 ± 3.8		b	10.35 ± 0.34		a
10 - 20 cm	*Control (arable)*	8	1.48 ± 0.02	ns	ns	22.4 ± 1.2	ns	ns	5.90 ± 0.25	ns	ns	1.79 ± 0.05	ns	ns	54.6 ± 1.7	a	a	9.92 ± 0.39	ns	ns
	All woodland	24	1.48 ± 0.02	ns		23.4 ± 0.5	ns		5.49 ± 0.16	ns		1.71 ± 0.06	ns		70.6 ± 2.3	b		10.41 ± 0.39	ns	
	Cherry	8	1.48 ± 0.04		ns	23.8 ± 0.9		ns	5.54 ± 0.30		ns	1.77 ± 0.12		ns	74.1 ± 3.8		b	10.05 ± 0.18		ns
	Hazel	8	1.47 ± 0.03		ns	23.0 ± 1.1		ns	5.19 ± 0.32		ns	1.57 ± 0.11		ns	67.9 ± 3.9		b	10.05 ± 0.31		ns
	Sycamore	8	1.49 ± 0.02		ns	23.4 ± 0.8		ns	5.74 ± 0.20		ns	1.79 ± 0.08		ns	69.6 ± 4.4		b	11.11 ± 1.13		ns
20 - 30 cm	*Control (arable)*	8	1.55 ± 0.03	ns	ab	22.0 ± 0.9	ns	ab	5.86 ± 0.20	ns	b	1.74 ± 0.06	b	b	81.5 ± 2.6	a	a	9.34 ± 0.26	ns	ab
	All woodland	24	1.53 ± 0.02	ns		22.6 ± 0.5	ns		5.41 ± 0.16	ns		1.48 ± 0.05	a		93.3 ± 2.8	b		9.57 ± 0.17	ns	
	Cherry	8	1.59 ± 0.01		a	21.6 ± 0.5		a	5.21 ± 0.21		a	1.48 ± 0.07		ac	96.9 ± 4.1		b	10.11 ± 0.27		b
	Hazel	8	1.48 ± 0.05		b	22.5 ± 1.2		ab	5.39 ± 0.38		ab	1.34 ± 0.09		a	88.6 ± 5.1		ab	9.33 ± 0.23		a
	Sycamore	8	1.52 ± 0.02		b	23.7 ± 0.6		b	5.63 ± 0.20		ab	1.61 ± 0.07		bc	94.5 ± 5.3		b	9.27 ± 0.33		ab
30-40 cm	*Control (arable)*	8	1.60 ± 0.03	ns	ns	21.2 ± 0.9	ns	ns	5.31 ± 0.18	a	b	1.34 ± 0.08	b	b	102.8 ± 3.1	ns	ns	9.33 ± 0.44	ns	ns
	All woodland	24	1.60 ± 0.03	ns		21.2 ± 0.6	ns		4.54 ± 0.12	b		0.90 ± 0.05	a		106.9 ± 3.5	ns		8.66 ± 0.35	ns	
	Cherry	8	1.62 ± 0.02		ns	21.3 ± 0.7		ns	4.52 ± 0.15		a	0.94 ± 0.07		a	111.8 ± 4.7		ns	8.50 ± 0.61		ns
	Hazel	6	1.60 ± 0.08		ns	21.2 ± 1.8		ns	4.38 ± 0.35		a	0.87 ± 0.16		a	98.1 ± 7.9		ns	8.63 ± 1.00		ns
	Sycamore	8	1.58 ± 0.05		ns	21.0 ± 1.3		ns	4.68 ± 0.18		a	0.89 ± 0.05		a	108.6 ± 5.6		ns	8.83 ± 0.31		ns
40-50 cm	*Control (arable)*	7	1.69 ± 0.03	ns	ns	20.5 ± 1.2	ns	ns	4.71 ± 0.27	ns	ab	0.85 ± 0.11	ns	ns	118.7 ± 5.2	ns	ns	10.60 ± 1.53	b	ns
	All woodland	22	1.64 ± 0.02	ns		21.3 ± 0.8	ns		4.81 ± 0.26	ns		0.71 ± 0.05	ns		119.0 ± 4.0	ns		8.41 ± 0.32	a	
	Cherry	8	1.65 ± 0.02		ns	20.9 ± 1.0		ns	4.28 ± 0.27		a	0.65 ± 0.06		ns	122.9 ± 5.2		ns	8.66 ± 0.43		ns
	Hazel	6	1.66 ± 0.03		ns	22.0 ± 1.1		ns	4.58 ± 0.27		ab	0.79 ± 0.14		ns	111.5 ± 10.3		ns	8.44 ± 0.95		ns
	Sycamore	8	1.60 ± 0.05		ns	21.1 ± 1.7		ns	5.51 ± 0.57		b	0.71 ± 0.06		ns	120.8 ± 6.2		ns	8.14 ± 0.38		ns

* *n* is the same for all soil indicators in this table

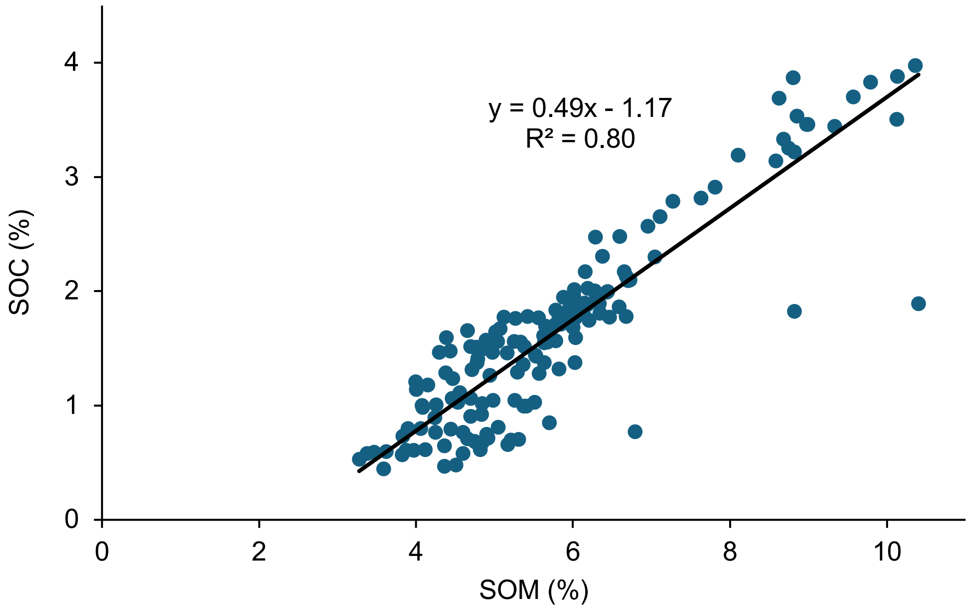
† Cumulative SOC stock values given, with depths representing 0-10 cm, 0-20 cm, 0-30 cm etc.

Supplementary Table A.3 – Further indicator means with standard error by treatment type and site for 0-10 cm soil, showing data for pH, NO_3-N, plant-available N (NO_3-N + NH_4-N), PO_4-P, and saturated hydraulic conductivity (measured at surface). Significant differences denoted by letters (note that significant differences are calculated within dotted boxes only); ns – not significant. Note that as saturated hydraulic conductivity is log-normally distributed, standard error bounds are given as an asymmetrical range.

		pH				NO_3-N				Plant-available N				PO_4-P				Sat. hydr. conductivity			
Depth	**Treatment**	n	value	Sig.		n	mg kg^{-1}	Sig.		n	mg kg^{-1}	Sig.		n	mg kg^{-1}	Sig.		n	mm hr^{-1}	Sig.	
	Control (arable)	3	7.49 ± 0.14	ns	ns	6	15.65 ± 4.19	a	a	6	18.30 ± 5.19	ns	ab	6	32.3 ± 3.0	a	a	23	3.24 (2.64, 3.99)	a	a
	All woodland	12	6.91 ± 0.15	ns		24	4.08 ± 0.95	b		24	9.33 ± 1.38	ns		24	15.7 ± 2.4	b		91	1.31 (1.15, 1.49)	b	
	Cherry	4	6.70 ± 0.27		ns	8	3.99 ± 1.21		b	8	7.72 ± 1.09		ab	8	21.0 ± 3.0		b	30	1.24 (0.99, 1.56)		b
	Hazel	4	7.00 ± 0.33		ns	8	2.15 ± 0.61		b	8	6.84 ± 1.27		a	8	8.2 ± 0.5		c	29	1.49 (1.19, 1.86)		b
0 - 10 cm	Sycamore	4	7.02 ± 0.24		ns	8	6.08 ± 2.45		ab	8	13.45 ± 3.50		b	8	17.9 ± 5.7		abc	32	1.19 (0.96, 1.49)		b
	Plot 1	3	7.51 ± 0.07	a		6	4.53 ± 1.34	ns		6	10.19 ± 1.95	ab		6	22.2 ± 7.6	ns		23	2.11 (1.69, 2.65)	a	
	Plot 2	3	6.46 ± 0.14	b		6	1.32 ± 0.67	ns		6	6.26 ± 1.49	ab		6	9.2 ± 1.7	ns		21	0.39 (0.29, 0.51)	b	
	Plot 3	3	6.64 ± 0.37	ab		6	7.42 ± 3.11	ns		6	13.95 ± 4.57	b		6	17.4 ± 3.8	ns		24	1.62 (1.36, 1.94)	a	
	Plot 4	3	7.02 ± 0.20	ab		6	3.03 ± 0.78	ns		6	6.95 ± 0.77	a		6	13.9 ± 3.0	ns		23	1.96 (1.62, 2.37)	a	



Supplementary Figure A.1 - Nutrient concentration and C:N scatterplots between forest floor litter/humus and surface soil for a. P, b. N and c. C:N. Dots are coloured by tree species (orange – Hazel, purple – Cherry, green – Sycamore).



Supplementary Figure A.2 – Comparison of soil organic carbon (SOC, determined via combustion) and soil organic matter (SOM, determined via loss on ignition) values for all samples used in the study.

Appendix B. Supporting Information for Chapter 3

Supplementary Table B.1 – Mean and standard error values for all soil physical properties, by a. treatment and measurement depth and b. alley width and depth (AF_{12}: 12 m agroforestry alleys, AF_{24}: 24m agroforestry alleys). ‘All AF’ is inclusive of both ‘Tree row’ and ‘Alley’. Significant differences within dotted boxes denoted by letters.

a.

			BD				**SOM**				**Moisture**				**Sat. hydr. conductivity (K_s)**				**Sand**			**Silt**			**Clay**		
Depth	**Treatment**	**n**	g cm^{-3}	Sig.		**n**	%	Sig.		**n**	%	Sig.		**n**	mm hr^{-1} *	Sig.		**n**	%	Sig.		%	Sig.		%	Sig.	
0-10 cm	Control	8	1.05 ± 0.02	a	a	8	6.89 ± 0.19	a	a	8	24.23 ± 0.37	ns	ns	12	3.57 (2.52, 5.05)	ns	a	8	42.56 ± 0.93	a	a	38.74 ± 0.57	ns	ns	18.71 ± 0.92	ns	a
	All AF	*32*	*0.96 ± 0.02*	b		*32*	*7.76 ± 0.17*	b		*32*	*25.38 ± 0.34*	ns		*41*	*7.98 (6.04, 10.54)*	ns		*32*	*45.00 ± 0.53*	b		*37.68 ± 0.51*	ns		*17.32 ± 0.31*	ns	
	Tree row	16	0.93 ± 0.03		b	16	7.79 ± 0.27		b	16	25.86 ± 0.54		ns	21	12.37 (8.64, 17.70)		b	16	45.90 ± 0.72		b	37.62 ± 0.75		ns	16.48 ± 0.43		b
	Alley	16	1.00 ± 0.03		ab	16	7.73 ± 0.22		b	16	24.89 ± 0.39		ns	20	5.04 (3.33, 7.61)		ab	16	44.11 ± 0.73		ab	37.73 ± 0.73		ns	18.16 ± 0.33		a
10-20 cm	Control	8	0.98 ± 0.03	ns	ns	8	5.92 ± 0.22	ns	ns	8	22.05 ± 0.30	ns	ns														
	All AF	*32*	*0.99 ± 0.02*	ns		*32*	*6.18 ± 0.10*	ns		*32*	*22.52 ± 0.20*	ns															
	Tree row	16	1.00 ± 0.03		ns	16	6.16 ± 0.13		ns	16	22.82 ± 0.29		ns														
	Alley	16	0.99 ± 0.02		ns	16	6.21 ± 0.16		ns	16	22.22 ± 0.26		ns														
20-50 cm	Control	8	1.20 ± 0.03	a	a	8	4.31 ± 0.41	ns	ns	8	17.75 ± 0.33	ns	ns														
	All AF	*32*	*1.07 ± 0.02*	b		*32*	*3.81 ± 0.12*	ns		*32*	*17.19 ± 0.30*	ns															
	Tree row	16	1.08 ± 0.02		b	16	3.73 ± 0.14		ns	16	17.19 ± 0.35		ns														
	Alley	16	1.06 ± 0.03		b	16	3.89 ± 0.20		ns	16	17.20 ± 0.50		ns														

b.

			BD			**SOM**			**Moisture**			**Sat. hydr. conductivity (K_s)**			**Sand**		**Silt**		**Clay**	
Depth	**Treatment**	**n**	g cm^{-3}	Sig.	**n**	%	Sig.	**n**	%	Sig.	**n**	mm hr^{-1} *	Sig.	**n**	%	Sig.	%	Sig.	%	Sig.
0-10 cm	Control	8	1.05 ± 0.02	a	8	6.89 ± 0.19	a	8	24.23 ± 0.37	ns	12	3.57 (2.52, 5.05)	ns	8	42.56 ± 0.93	a	38.74 ± 0.57	a	18.71 ± 0.92	a
	AF_{12}	16	1.04 ± 0.03	a	16	7.39 ± 0.19	a	16	25.18 ± 0.51	ns	17	8.07 (5.45, 11.96)	ns	16	46.14 ± 0.72	b	35.79 ± 0.50	b	18.06 ± 0.44	a
	AF_{24}	16	0.89 ± 0.02	b	16	8.14 ± 0.26	b	16	25.57 ± 0.45	ns	24	7.91 (5.34, 11.72)	ns	16	43.86 ± 0.68	a	39.57 ± 0.61	a	16.57 ± 0.35	b
10-20 cm	Control	8	0.98 ± 0.03	ab	8	5.92 ± 0.22	ns	8	22.05 ± 0.30	ns										
	AF_{12}	16	1.06 ± 0.02	a	16	6.13 ± 0.11	ns	16	22.43 ± 0.26	ns										
	AF_{24}	16	0.93 ± 0.02	b	16	6.23 ± 0.17	ns	16	22.62 ± 0.32	ns										
20-50 cm	Control	8	1.20 ± 0.03	a	8	4.31 ± 0.41	ab	8	17.75 ± 0.33	ns										
	AF_{12}	16	1.06 ± 0.02	b	16	4.11 ± 0.14	a	16	17.34 ± 0.40	ns										
	AF_{24}	16	1.09 ± 0.03	ab	16	3.51 ± 0.17	b	16	17.05 ± 0.46	ns										

* Saturated hydraulic conductivity is log-normally distributed and presented in the form *geometric mean* (*lower std. error bound*, *upper std. error bound*)

Supplementary Table B.2 – Mean and standard error values for all soil chemical properties, by a. treatment and measurement depth and b. alley width and depth (AF_{12}: 12 m agroforestry alleys, AF_{24}: 24m agroforestry alleys). 'All AF' is inclusive of both 'Tree row' and 'Alley'. Significant differences within dotted boxes denoted by letters.

a.

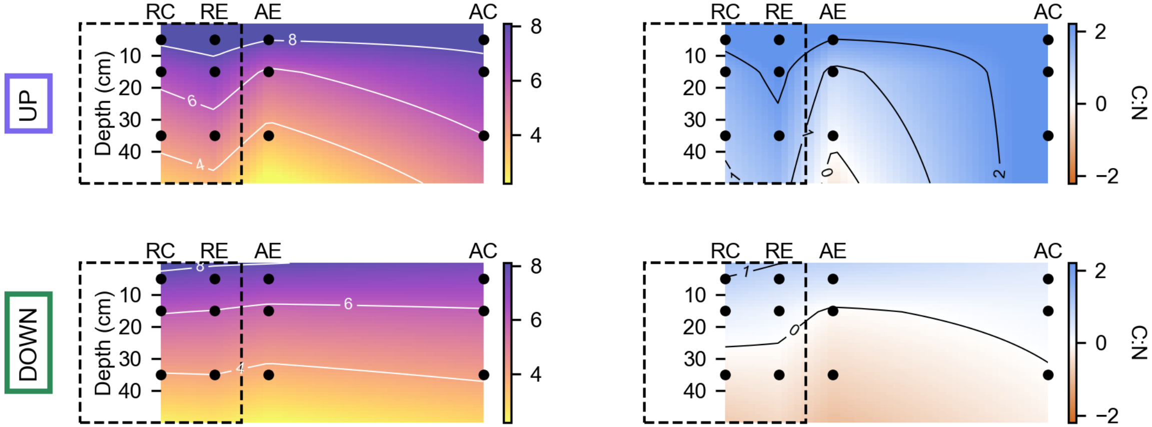
			SOC						Available N (NO_3-N + NO_2-N + NH_4-N)						Available P (PO_4-P)						C:N		
Depth	**Treatment**	***n****	%	Sig.		cum. t ha^{-1}	Sig.		mg kg^{-1}	Sig.		cum. kg ha^{-1}	Sig.		mg kg^{-1}	Sig.		cum. kg ha^{-1}	Sig.		ratio	Sig.	
0-10 cm	Control	8	2.69 ± 0.07	a	a	28.5 ± 0.7	a	a	6.00 ± 0.41	a	a	6.3 ± 0.4	a	a	42.5 ± 5.0	a	a	44.9 ± 5.3	a	a	6.36 ± 0.17	a	a
	All AF	*32*	*3.44 ± 0.12*	b		*36.8 ± 1.3*	b		*9.36 ± 0.72*	b		*10.0 ± 0.8*	b		*68.1 ± 3.3*	b		*72.7 ± 3.5*	b		*7.86 ± 0.23*	b	
	Tree row	16	3.49 ± 0.18		b	37.3 ± 2.1		b	10.47 ± 1.09		b	11.2 ± 1.2		b	66.6 ± 4.4		b	71.1 ± 4.6		b	8.05 ± 0.34		b
	Alley	16	3.39 ± 0.16		b	36.2 ± 1.7		b	8.25 ± 0.89		ab	8.8 ± 0.9		ab	69.7 ± 5.0		b	74.4 ± 5.3		b	7.68 ± 0.33		b
10-20 cm	Control	8	1.93 ± 0.06	a	a	47.3 ± 1.3	a	a	4.80 ± 0.23	a	a	11.0 ± 0.7	a	a	38.9 ± 5.3	a	a	82.9 ± 10.3	a	a	5.37 ± 0.16	ns	ns
	All AF	*32*	*2.29 ± 0.07*	b		*59.2 ± 1.9*	b		*6.61 ± 0.41*	b		*16.5 ± 1.1*	b		*63.5 ± 3.6*	b		*135.0 ± 7.0*	b		*6.34 ± 0.28*	ns	
	Tree row	16	2.28 ± 0.08		b	59.7 ± 2.8		b	7.30 ± 0.55		b	18.3 ± 1.7		b	62.2 ± 4.8		b	132.0 ± 9.2		b	6.47 ± 0.39		ns
	Alley	16	2.30 ± 0.13		ab	58.8 ± 2.7		b	5.92 ± 0.58		a	14.6 ± 1.4		a	64.9 ± 5.6		b	137.9 ± 10.7		b	6.21 ± 0.41		ns
20-50 cm	Control	8	0.96 ± 0.05	ns	ns	82.0 ± 2.8	a	a	2.98 ± 0.11	ns	ns	21.8 ± 0.9	a	a	32.4 ± 5.6	a	a	199.3 ± 29.7	a	a	3.81 ± 0.19	ns	ns
	All AF	*32*	*1.04 ± 0.06*	ns		*96.4 ± 3.3*	b		*3.65 ± 0.33*	ns		*29.6 ± 1.9*	b		*55.5 ± 4.5*	b		*334.3 ± 22.9*	b		*4.38 ± 0.34*	ns	
	Tree row	16	0.97 ± 0.04		ns	94.5 ± 2.9		b	3.73 ± 0.29		ns	31.7 ± 2.1		b	54.4 ± 6.1		b	327.3 ± 30.6		b	4.42 ± 0.48		ns
	Alley	16	1.10 ± 0.12		ns	98.3 ± 6.1		ab	3.58 ± 0.61		ns	27.5 ± 3.1		ab	56.6 ± 6.9		b	341.4 ± 35.1		b	4.33 ± 0.50		ns

b.

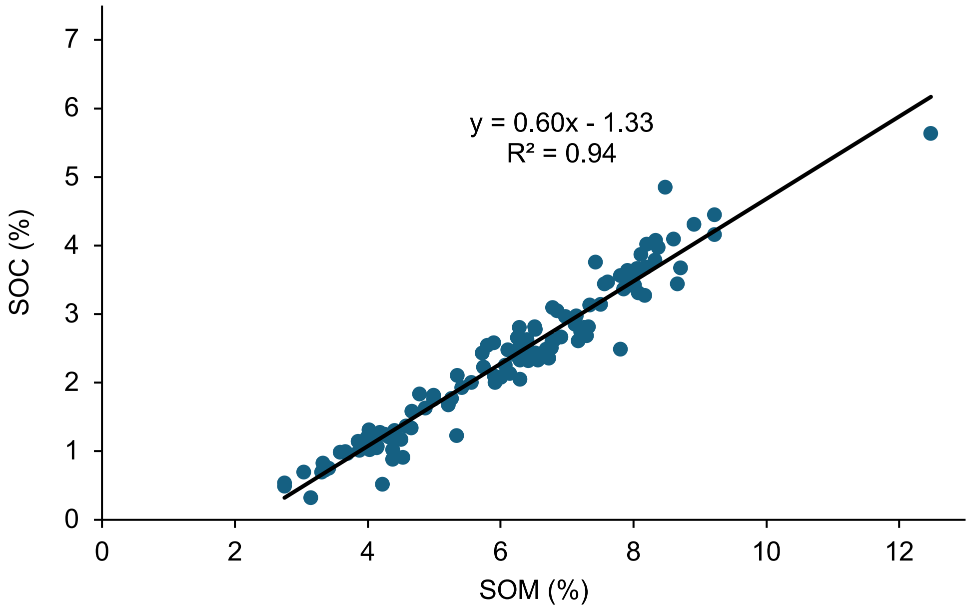
			SOC				Available N				Available P				C:N	
Depth	**Treatment**	***n****	%	Sig.	cum. t ha^{-1}	Sig.	mg kg^{-1}	Sig.	cum. kg ha^{-1}	Sig.	mg kg^{-1}	Sig.	cum. kg ha^{-1}	Sig.	ratio	Sig.
0-10 cm	Control	8	2.69 ± 0.07	a	28.5 ± 0.7	a	6.00 ± 0.41	a	6.3 ± 0.4	a	42.5 ± 5.0	a	44.9 ± 5.3	a	6.36 ± 0.17	a
	AF_{12}	16	3.03 ± 0.11	a	32.2 ± 1.2	a	10.03 ± 1.23	b	10.7 ± 1.3	b	78.6 ± 5.0	b	83.5 ± 5.3	b	8.60 ± 0.24	b
	AF_{24}	16	3.85 ± 0.15	b	41.4 ± 1.7	b	8.68 ± 0.76	b	9.3 ± 0.8	b	57.7 ± 2.2	c	61.9 ± 2.5	c	7.13 ± 0.31	a
10-20 cm	Control	8	1.93 ± 0.06	a	47.3 ± 1.3	a	4.80 ± 0.23	a	11.0 ± 0.7	a	38.9 ± 5.3	a	82.9 ± 10.3	a	5.37 ± 0.16	a
	AF_{12}	16	2.19 ± 0.07	b	53.7 ± 1.8	b	7.25 ± 0.67	b	17.8 ± 1.9	b	75.8 ± 5.5	b	157.7 ± 10.7	b	7.50 ± 0.23	b
	AF_{24}	16	2.39 ± 0.13	b	64.8 ± 2.8	c	5.96 ± 0.44	b	15.2 ± 1.1	b	51.2 ± 2.0	c	112.2 ± 4.2	c	5.18 ± 0.30	a
20-50 cm	Control	8	0.96 ± 0.05	ns	82.0 ± 2.8	a	2.98 ± 0.11	ns	21.8 ± 0.9	a	32.4 ± 5.6	a	199.3 ± 29.7	a	3.81 ± 0.19	a
	AF_{12}	16	1.13 ± 0.07	ns	94.4 ± 3.3	b	4.05 ± 0.50	ns	32.4 ± 3.0	b	70.4 ± 6.7	b	411.3 ± 34.5	b	5.74 ± 0.29	b
	AF_{24}	16	0.94 ± 0.10	ns	98.4 ± 5.9	ab	3.26 ± 0.43	ns	26.8 ± 2.0	b	40.6 ± 3.3	a	257.4 ± 13.9	a	3.01 ± 0.39	c

* *n* is the same for all indicators in Supplementary Table 2a and 2b

a. C:N



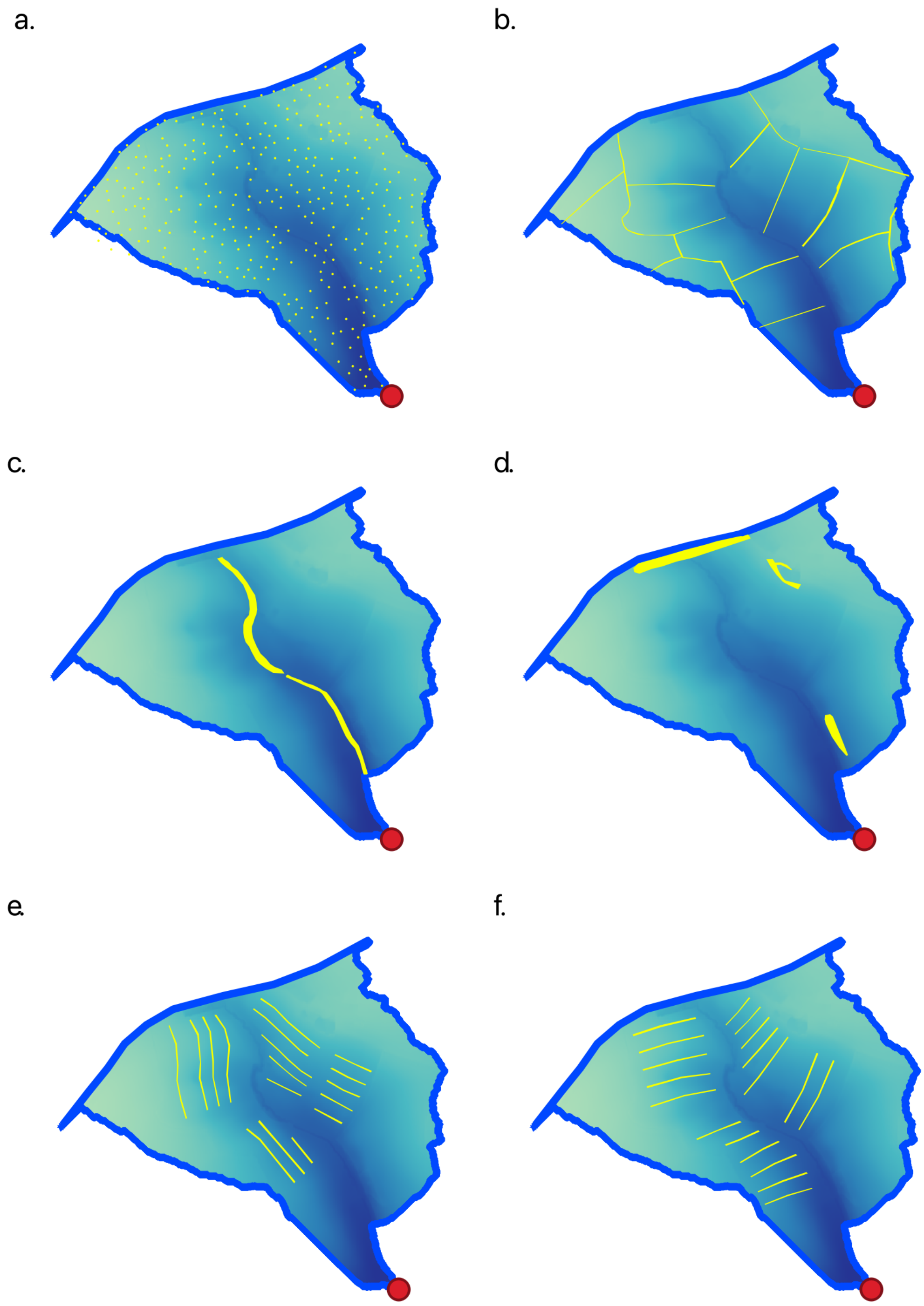
Supplementary Figure B.1 – Cross-sectional variability in C:N in upslope (UP - purple) and downslope (DOWN - green) positions, and row centre (RC), row edge (RE), alley edge (AE) and alley centre (AC) positions. Lateral distances between points are not shown to scale. Left column of plots shows variation in absolute values within combined agroforestry areas, right column shows differences between agroforestry and control areas. Data points are shown as black dots, with *n* = 4 for each point. Contours mapped using two-dimensional linear interpolation between data points.



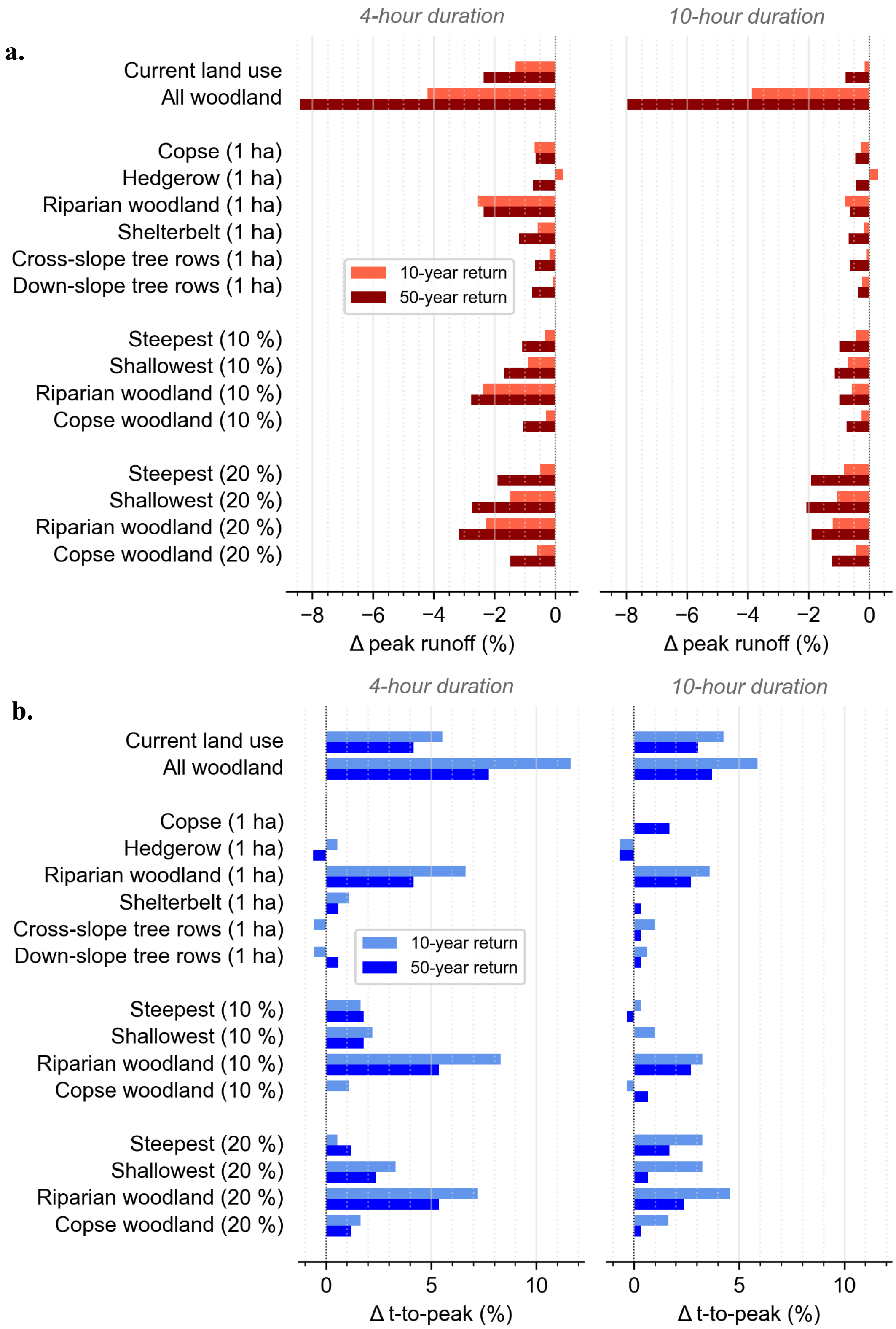
Supplementary Figure B.2 – Comparison of soil organic carbon (SOC, determined via combustion) and soil organic matter (SOM, determined via loss on ignition) values for all samples used in the study.

Appendix C. Supporting Information for Chapter 4

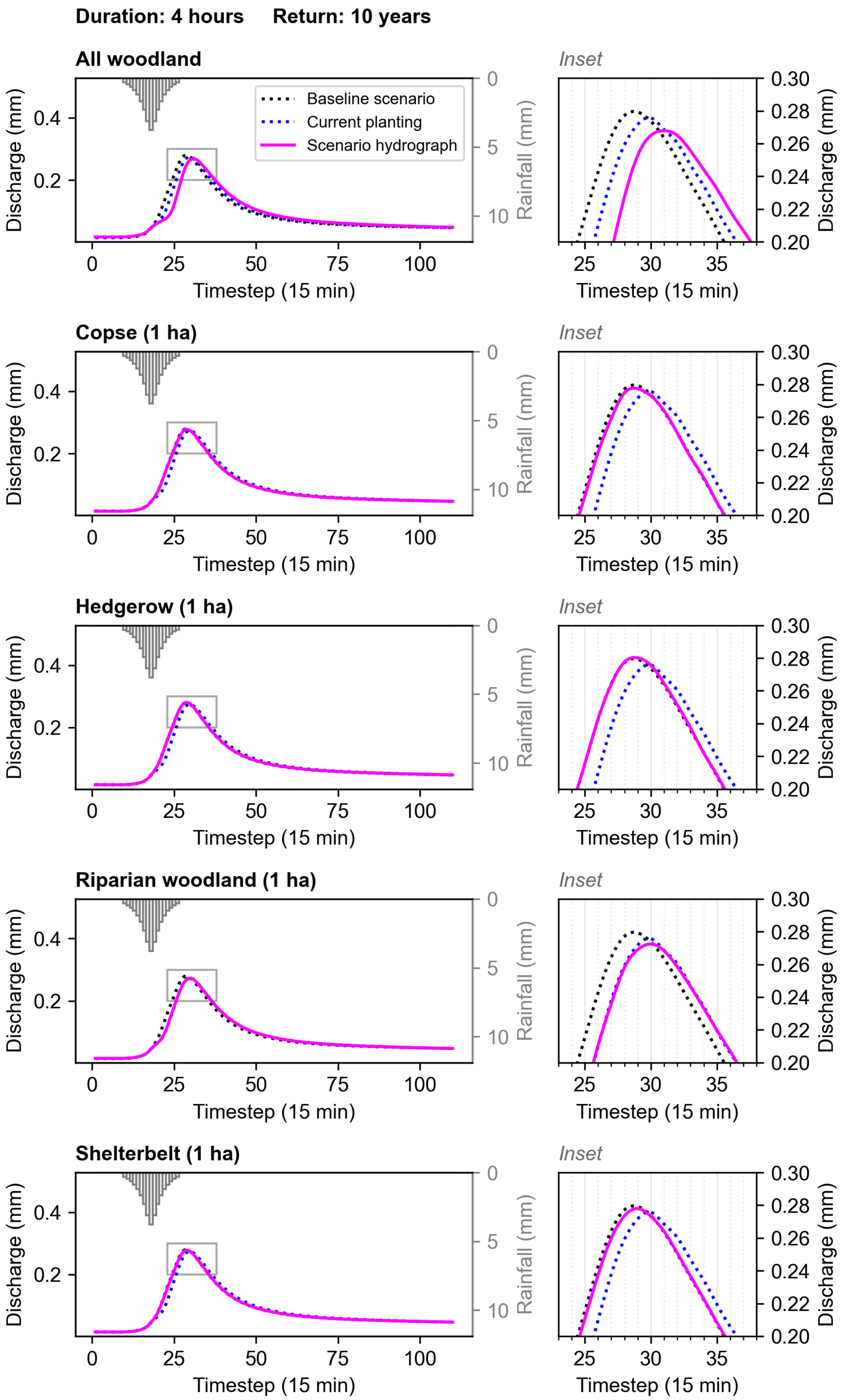
Supplementary Figure C.1 - Within-farm scenarios based on current planting for agroforestry, with each scenario adjusted to an equivalent 1 ha area. Agroforestry scenarios include a. copse planting, b. hedgerows, c. riparian woodland, d. shelterbelt, e. cross-slope tree rows and f. down-slope tree rows.

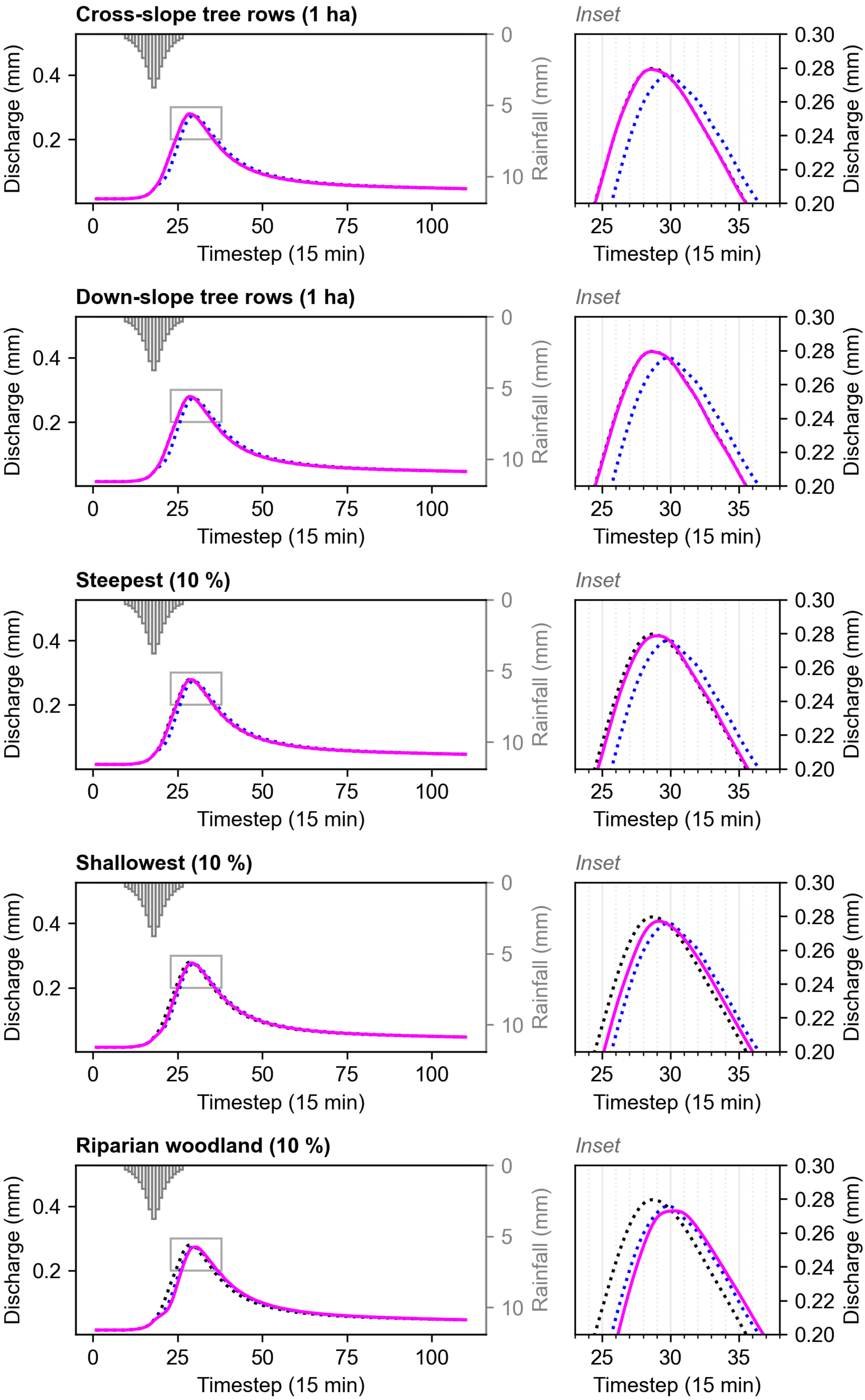


Supplementary Figure C.2 - Percentage changes in a. peak runoff and b. time-to-peak values compared with UKCEH baseline land use for all scenarios, for both 4-hour and 10-hour duration storms and 10- and 50-year return periods. 'All woodland' is a scenario in which the whole farmed area is converted to woodland. 'Steepest' and 'Shallowest' refer to the steepest and shallowest gradients within the farmed area, respectively. Percentage values in brackets indicate proportion of the farmed area converted to each land cover type. See main text (Fig. 4.6) for absolute differences.

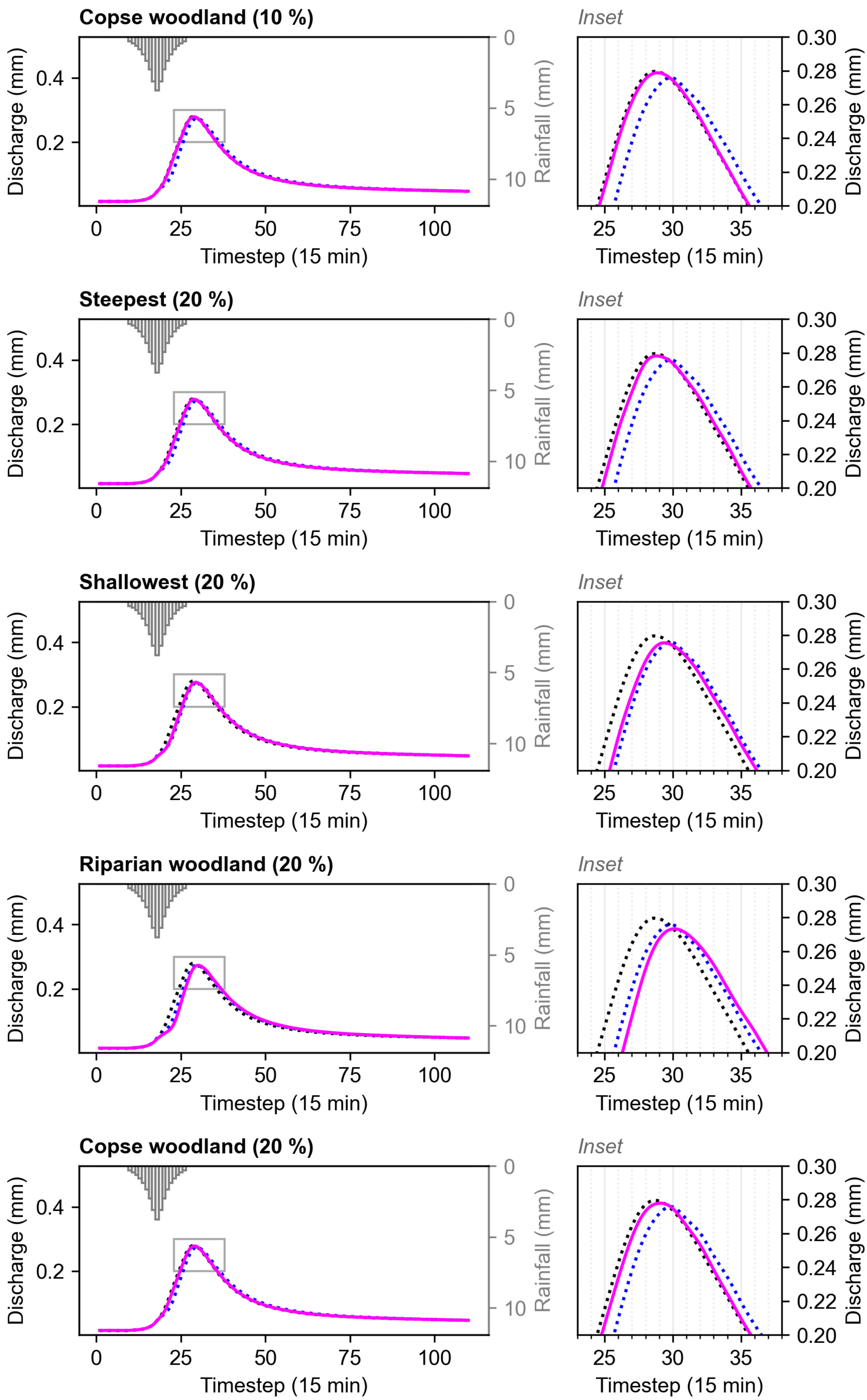


Supplementary Figure C.3 (following 12 pages) – hydrograph response for all agroforestry scenarios, storm durations and return periods. Scenario hydrograph shown as solid lines, with UKCEH baseline (black dotted) and all current agroforestry (blue dotted) hydrographs illustrated for comparison. Inset area corresponds to grey box on left plot for each storm. All insets shown to same scale.

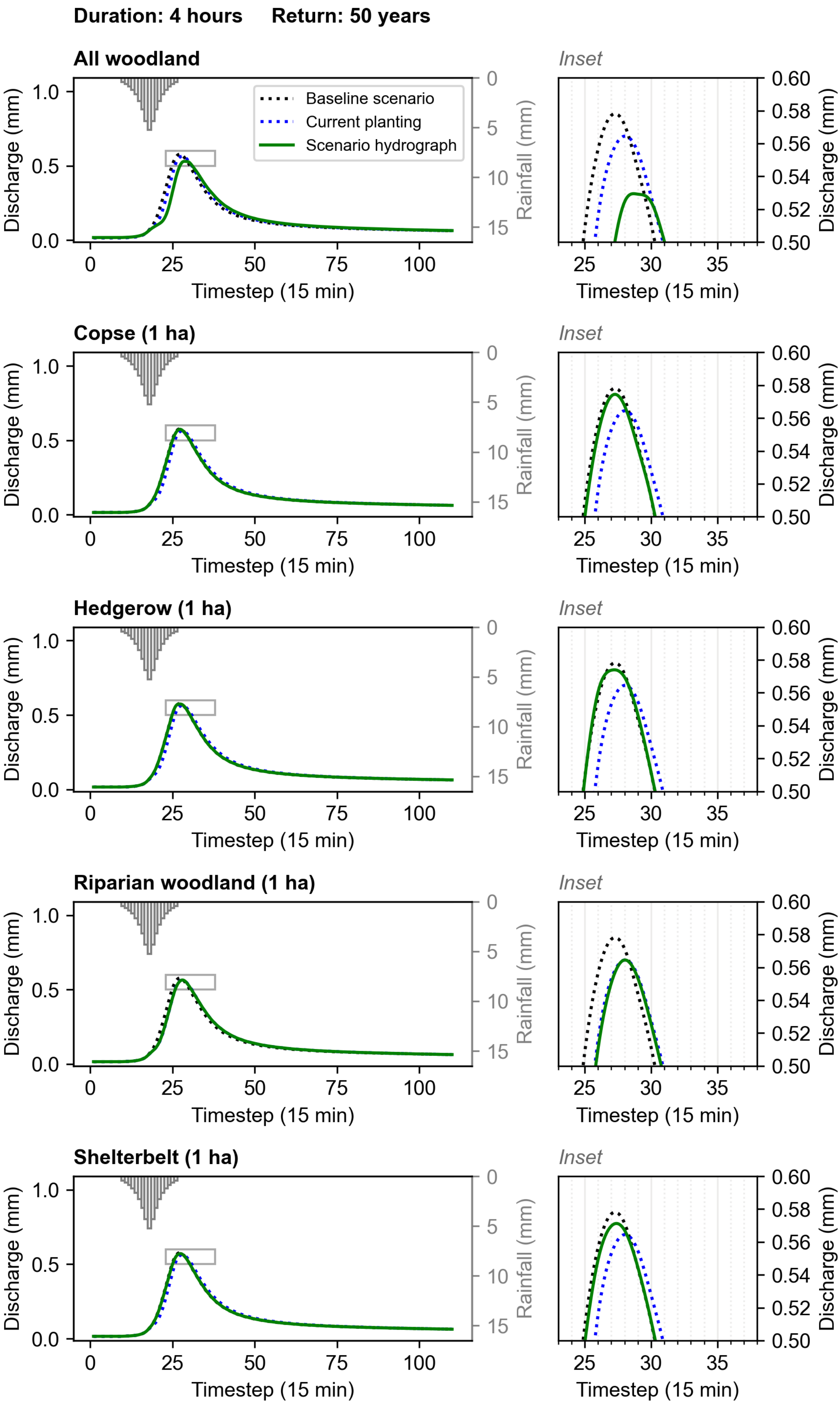


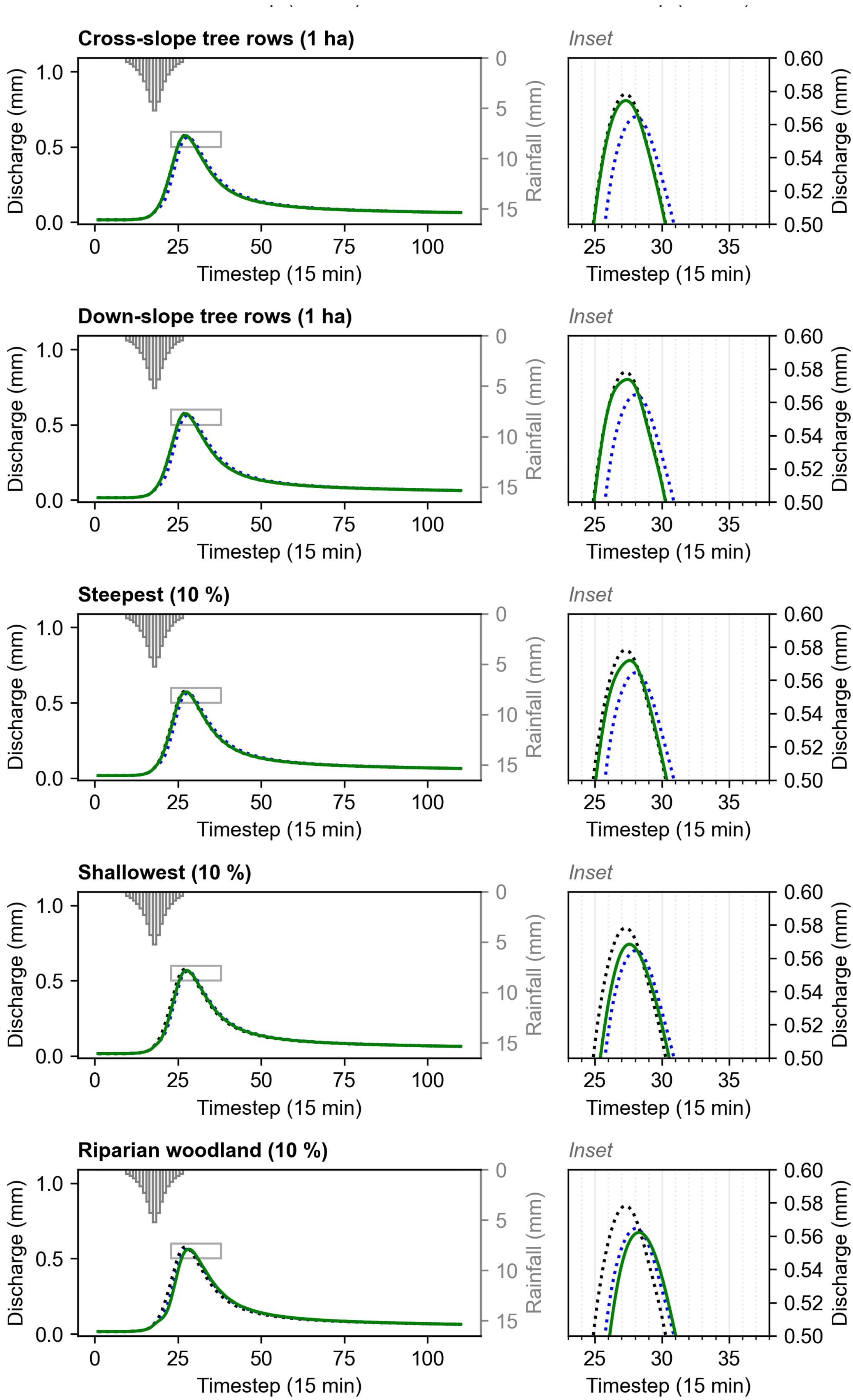


Cross-slope tree rows (1 ha)
Down-slope tree rows (1 ha)
Steepest (10 %)
Shallowest (10 %)
Riparian woodland (10 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)

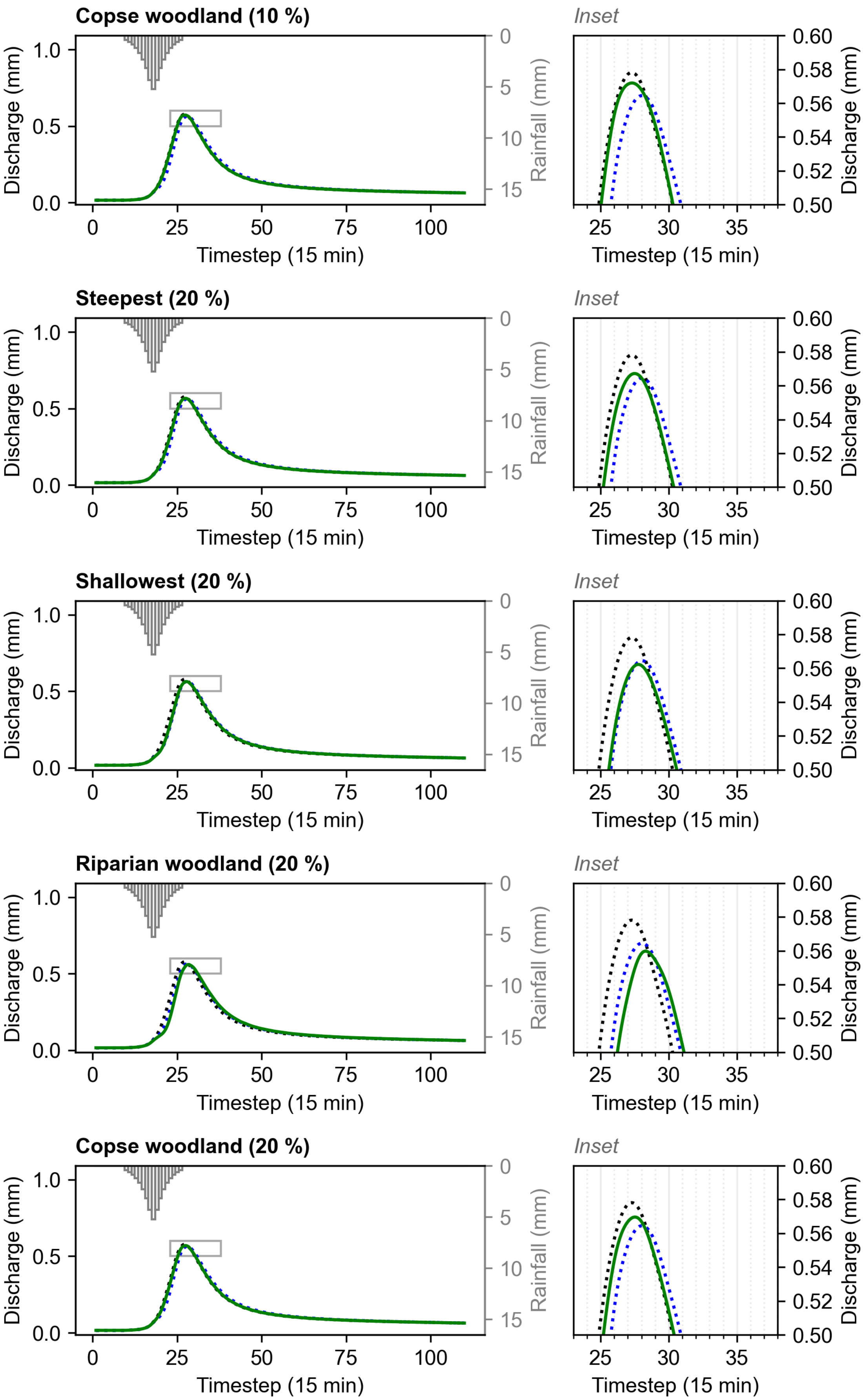


Copse woodland (10 %)
Steepest (20 %)
Shallowest (20 %)
Riparian woodland (20 %)
Copse woodland (20 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)

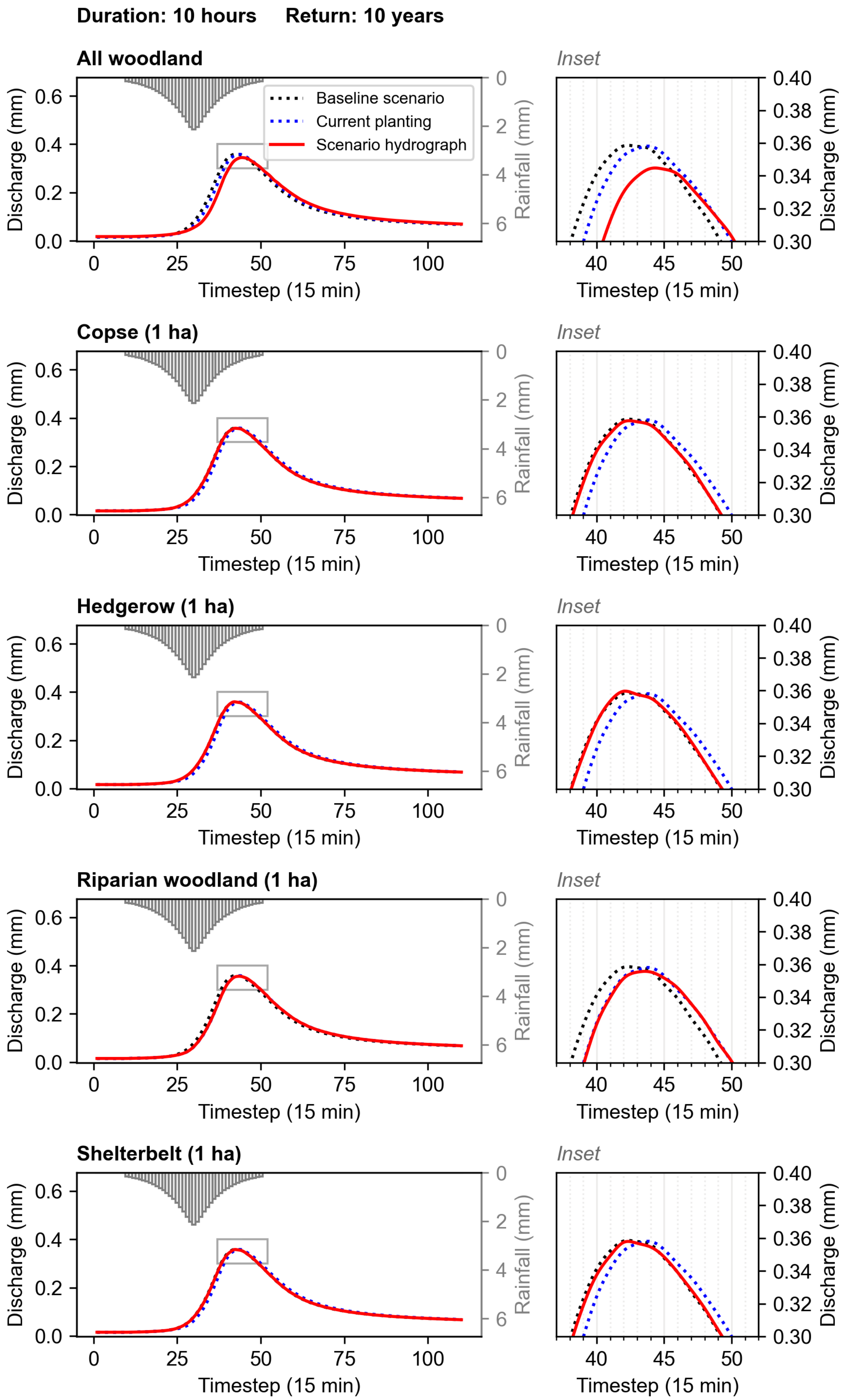

Duration: 4 hours
Return: 50 years
All woodland
Copse (1 ha)
Hedgerow (1 ha)
Riparian woodland (1 ha)
Shelterbelt (1 ha)
Inset
Baseline scenario
Current planting
Scenario hydrograph
Discharge (mm)
Rainfall (mm)
Timestep (15 min)

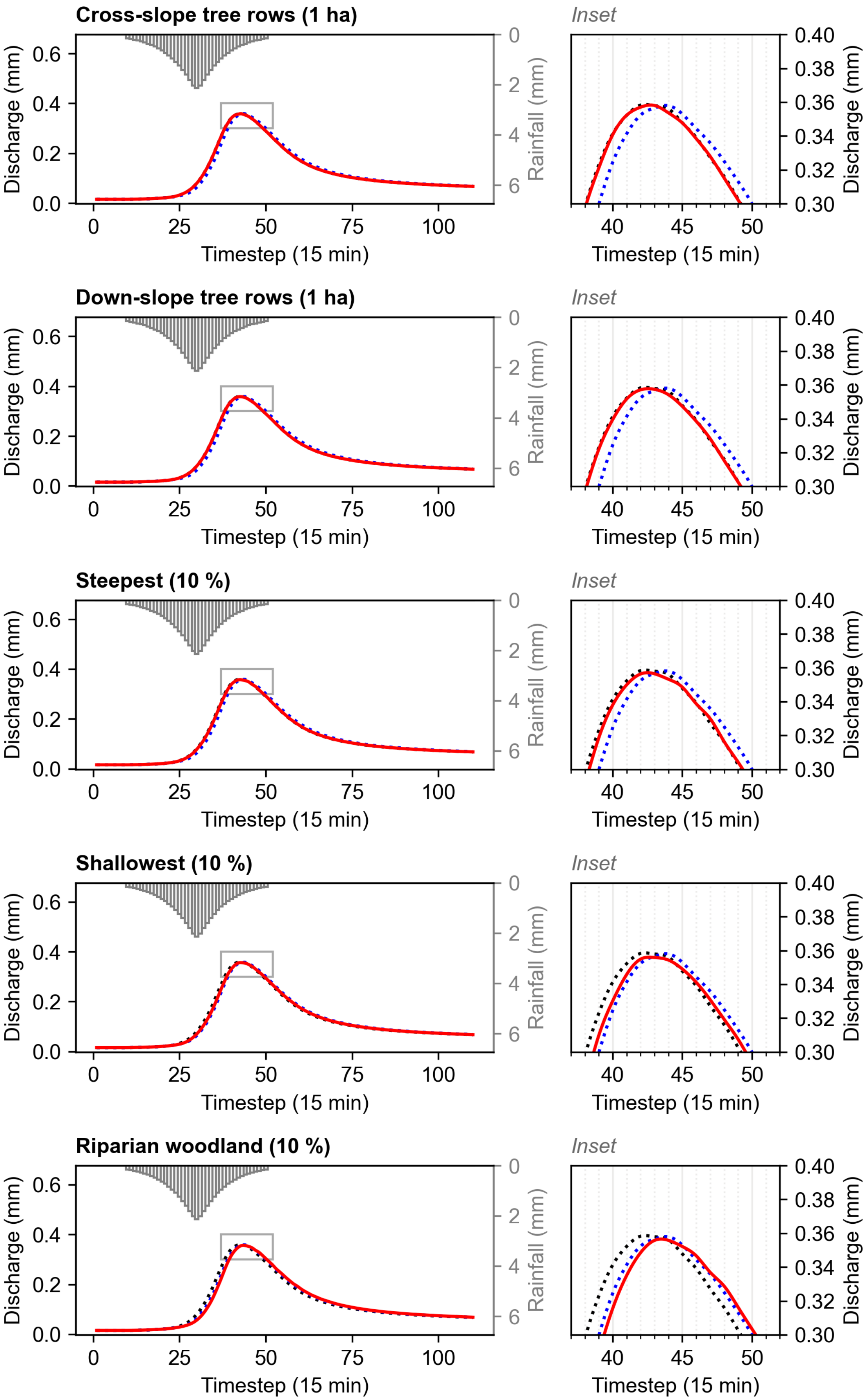


Cross-slope tree rows (1 ha)
Down-slope tree rows (1 ha)
Steepest (10 %)
Shallowest (10 %)
Riparian woodland (10 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)

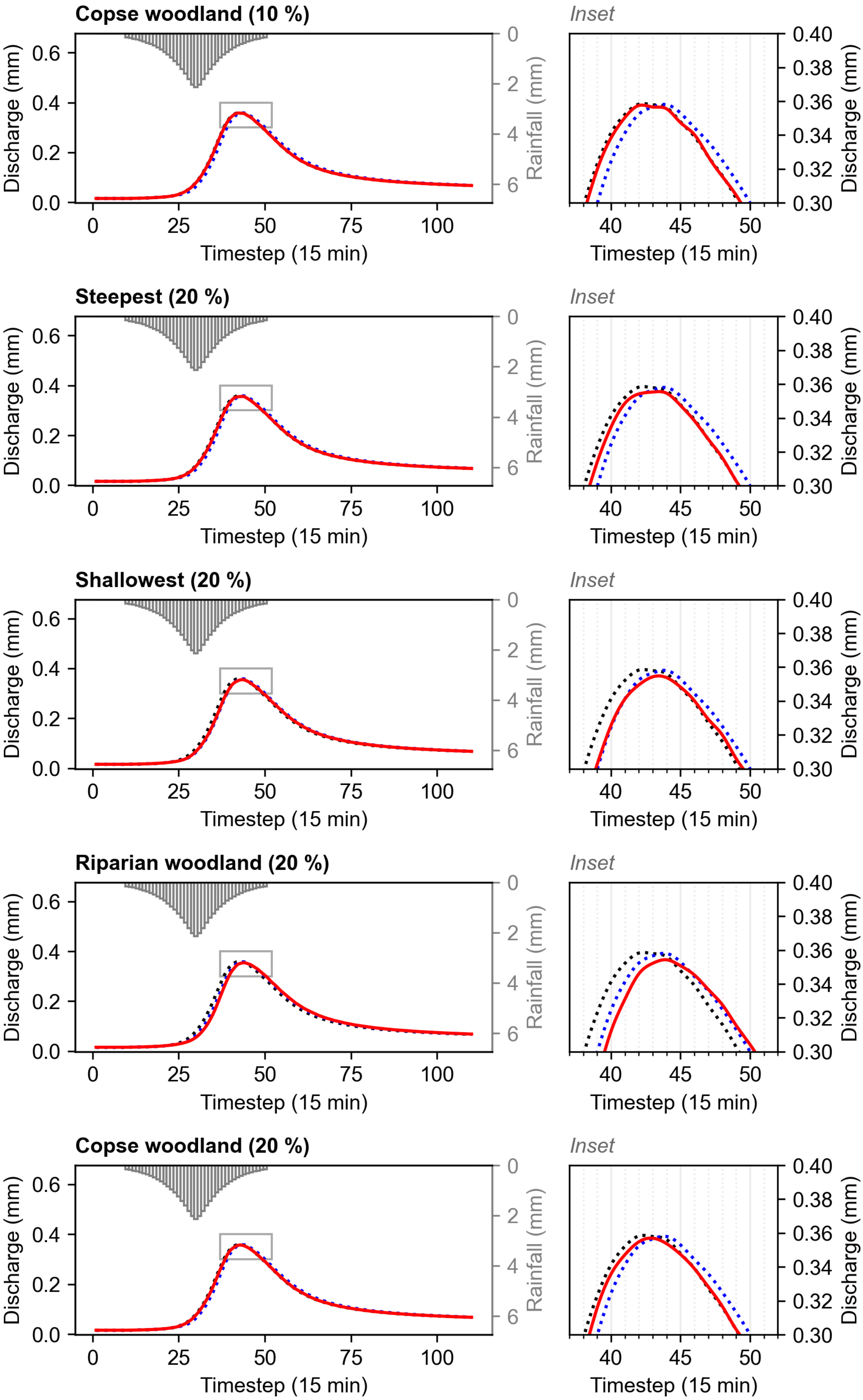


Copse woodland (10 %)
Inset
Steepest (20 %)
Inset
Shallowest (20 %)
Inset
Riparian woodland (20 %)
Inset
Copse woodland (20 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)
1.0
0.5
0.0
0
5
10
15
25
50
75
100
0.60
0.58
0.56
0.54
0.52
0.50
30
35

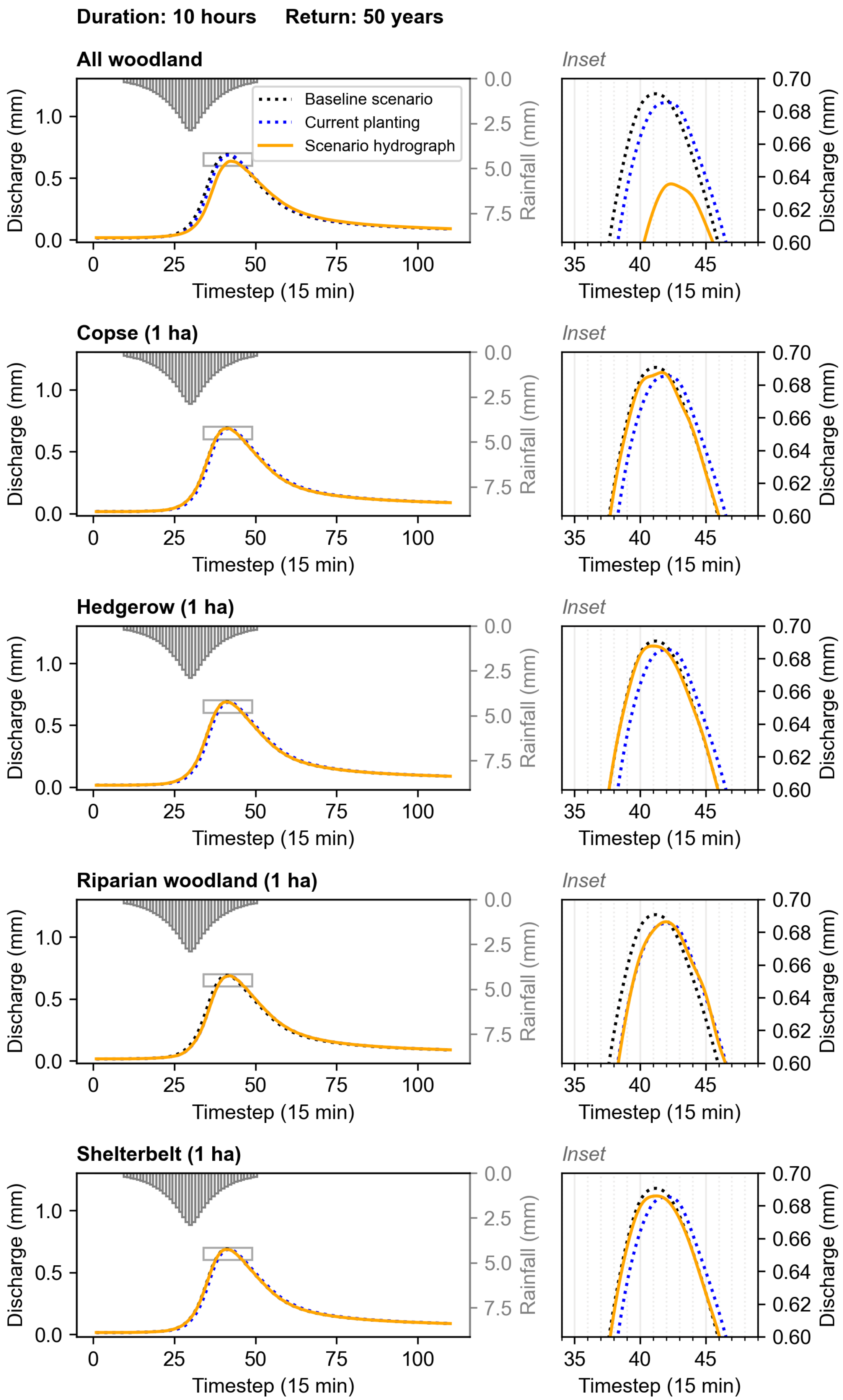

Duration: 10 hours Return: 10 years
All woodland
Copse (1 ha)
Hedgerow (1 ha)
Riparian woodland (1 ha)
Shelterbelt (1 ha)
Inset
Baseline scenario
Current planting
Scenario hydrograph
Discharge (mm)
Rainfall (mm)
Timestep (15 min)

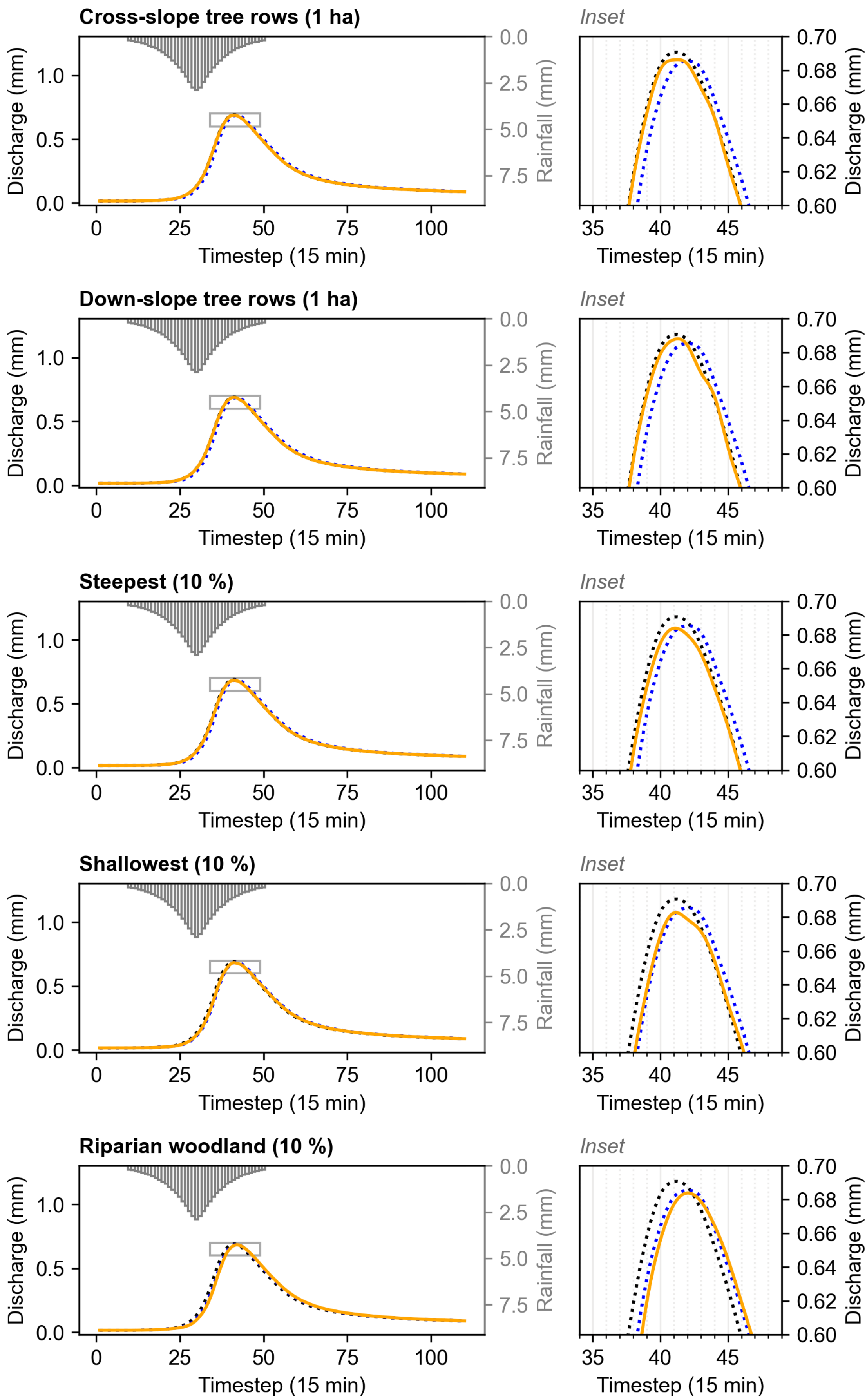


Cross-slope tree rows (1 ha)
Down-slope tree rows (1 ha)
Steepest (10 %)
Shallowest (10 %)
Riparian woodland (10 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)

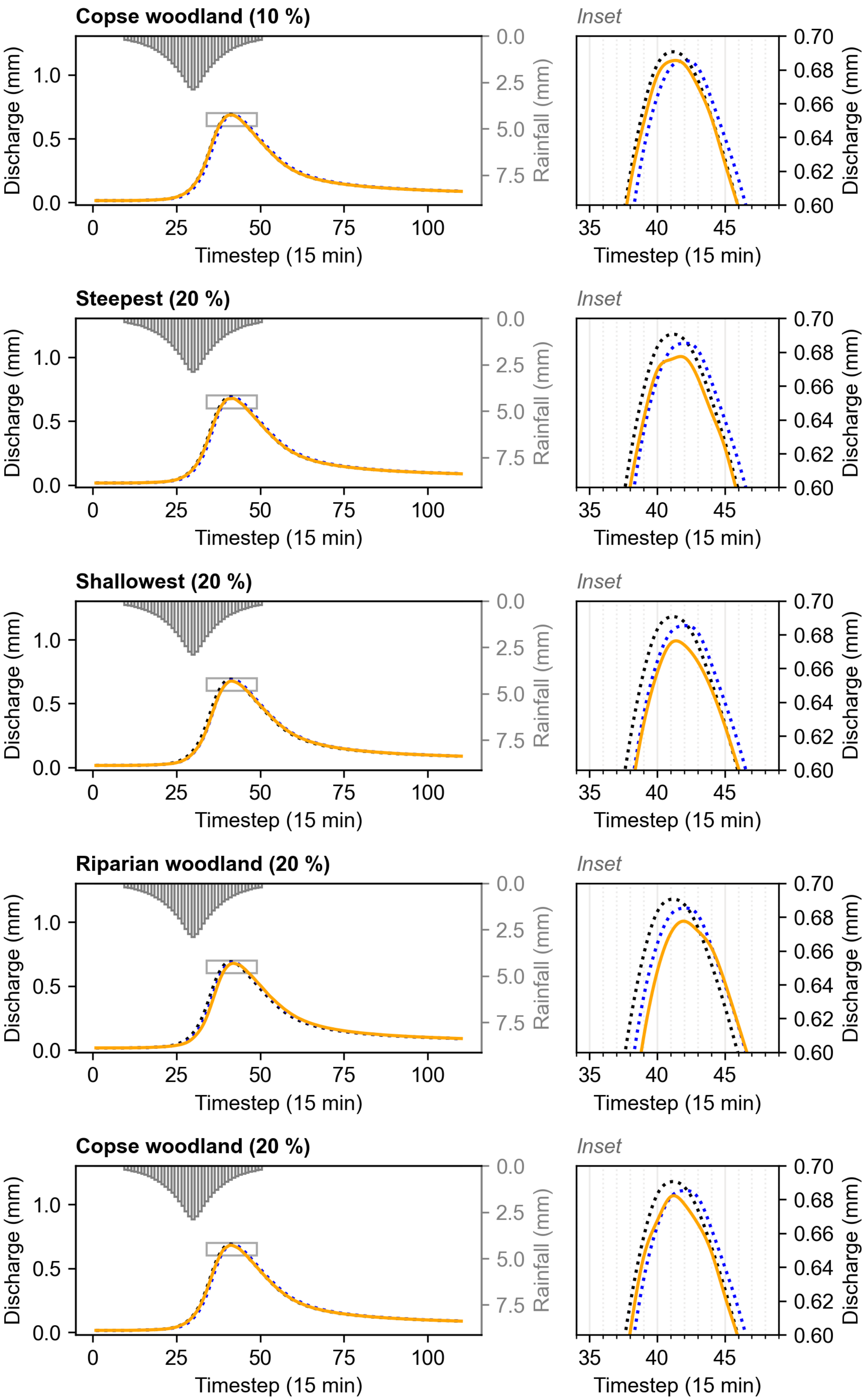


Copse woodland (10 %)
Inset
Steepest (20 %)
Inset
Shallowest (20 %)
Inset
Riparian woodland (20 %)
Inset
Copse woodland (20 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)
0.0
0.2
0.4
0.6
0
2
4
6
0
25
50
75
100
0.30
0.32
0.34
0.36
0.38
0.40
40
45
50


Duration: 10 hours Return: 50 years
All woodland
Copse (1 ha)
Hedgerow (1 ha)
Riparian woodland (1 ha)
Shelterbelt (1 ha)
Inset
Baseline scenario
Current planting
Scenario hydrograph
Discharge (mm)
Rainfall (mm)
Timestep (15 min)



Cross-slope tree rows (1 ha)
Down-slope tree rows (1 ha)
Steepest (10 %)
Shallowest (10 %)
Riparian woodland (10 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)


Copse woodland (10 %)
Inset
Steepest (20 %)
Inset
Shallowest (20 %)
Inset
Riparian woodland (20 %)
Inset
Copse woodland (20 %)
Inset
Discharge (mm)
Rainfall (mm)
Timestep (15 min)