The influence of trees on soil hydraulic properties: species effects and the provision of ecosystem services

Kathryn Ruth Chandler

BSc(Hons) MRes

Submitted for the degree of Doctor of Philosophy
Lancaster Environment Centre, Lancaster University
January 2014

Declaration

Except where reference is made to other sources, I declare that the work in this thesis is my own and has not been previously submitted, in part or in full, to any institution for any other degree or qualification.

Kathryn R. Chandler
Lancaster University, January 2014

Acknowledgements

The research presented in this thesis was funded by a BBSRC targeted priority studentship with additional financial support from the Centre for Ecology and Hydrology (CEH). I am very grateful to both for making this project possible. I would also like to thank the British Society of Soil Science and Lancaster University's Faculty of Science and Technology for providing grants that enabled me to present my work at the Eurosoil 2012 conference and the British Ecological Society for contributing towards the costs of attending a Field Studies Council course on sampling and identifying soil invertebrates.

I first conceived this project while undertaking an MRes at Lancaster University and I would like to thank Prof. Felix Wäckers for taking an interest in the research, for agreeing to be the principal supervisor for the BBSRC funding application and for supporting me through the application process. I am also very grateful to Dr. Carly Stevens, who subsequently became my principal supervisor following Felix's departure to take up a new post in Belgium; thank you for your support and encouragement, particularly through a difficult period during the PhD. I would like to thank Prof. Brian Reynolds for supporting the partnership between Lancaster University and CEH. I would also like to thank Dr. Aidan Keith, who took over as my CEH supervisor when Brian retired; I am very grateful for your advice, support and enthusiasm. Thanks also go to Prof. Andrew Binley and Prof. Richard Bardgett.

Much of the fieldwork was undertaken at the Glensaugh experimental agroforestry field site in Scotland and I would like to thank the James Hutton Institute for allowing me to work there. I would particularly like to thank the Glensaugh farm staff, Donald Barrie, Dave Hamilton, June and Jim Scott, Owen and the sheepdogs for making me so welcome. I really enjoyed my time spent at Glensaugh and I will miss my visits.

There was a great deal of field work involved in this project and I am very grateful to the many people who volunteered their time to help, either travelling up to Glensaugh with me, or just giving up an hour or two to assist with setting up and sampling the mesocosm experiment at Hazelrigg. They are, in alphabetical order: Calum Brown, Andy Cole, Anna England, Marina Dudley-Southern, Ahmed Elshenawy, Rachel

Eyre, Sam France, Iain Gould, James Heath, Carly Hoyle, Jenni Hughes, Soji Igunnugbemi, Vassil Karloukovski, John Keery, Dr Aidan Keith, Jonathan Kinnear, Eduardo Medina, Lakam Mejus, Peter Metcalfe, Dr. Trevor Piearce, Lawrence Pinell, Edmundo Placencia, Jos Ratcliffe, Isabel Rogers, Peter Shanahan, Dr Carly Stevens, Danny Tregidgo, Victor van Velzen and Mike Whitfield. I would particularly like to thank Nicola Thompson, who identified the species of ground flora in the treatments at Glensaugh and I would like to thank Ricky Dunn, Hazel Fielding and Hannah Wright who helped me to process some of the samples from the mesocosm experiment.

I am very grateful to Prof. Wilfred Otten and Dr Simona Hapca from the SIMBIOS Centre at the University of Abertay for their advice on image analysis and for using their software to segment my CT images.

I'd also like to thank Dr. Peter Henrys (CEH) for providing statistical advice and the staff in the Lancaster Environment Centre's workshop, finance office and postgraduate office for their assistance with practical matters.

Finally I'd like to thank my colleagues in the research groups that I was part of, especially Catherine Baxendale, Iain Gould and Helen Quirk, for providing encouragement, support, entertainment and a shoulder to cry on when needed!

Abstract

Trees contribute to a number of important ecosystem services through their influence on soil properties and functions, including water purification and regulation. Forest soils tend to have a higher saturated hydraulic conductivity (K_s) than soils under other vegetation, so the establishment of streamside tree buffer zones and strategically planted woodland is increasingly being promoted to reduce or intercept surface runoff from agricultural and urban environments that may contribute to diffuse pollution and flooding. Despite this, our understanding of how trees affect soil hydraulic properties is still extremely limited. The main aims of the work outlined in this thesis were to compare the effect of a broadleaf (sycamore) and a conifer (Scots pine) tree on K_s to determine if this differs between species type, to examine the effect of land use on K_s under forest cover and to investigate the key underlying mechanisms involved. Results showed significantly higher K_s under undisturbed mature Scots pine forest compared with sycamore forest, indicating a species effect. Forest soils also had significantly higher K_s than pasture grazed by sheep; however, when sheep were grazed under forest cover, there was no significant difference between forest and pasture, suggesting that the effect of land use masked the effect of trees on K_s. An investigation of soil flow pathways and pore characteristics showed that significantly higher K_s corresponded with significantly higher soil macroporosity and macropore flow. A mesocosm experiment, undertaken to investigate the influence of tree roots and associated soil macrofauna on K_s, was found to be of insufficient duration for significant changes to occur; however non-significant higher K_s observed under Scots pine compared with sycamore were consistent with the study of more mature trees. The results of a glasshouse experiment investigating the influence of leaf litter on K_s showed that leaf litter had a negative effect and litter type had more of an effect than litter quantity.

CONTENTS

List of figures

List of tables

List of symbols and abbreviations

θ	Porosity
ρ_b	Bulk density
CT	Computed tomography
HFA	Home field advantage
I_{max}	Maximum rainfall intensity
IOF	Infiltration excess overland flow
K_{-10}	Unsaturated hydraulic conductivity at a pressure head of -10 cm
K_{fs}	Field-saturated hydraulic conductivity
K_s	Saturated hydraulic conductivity
mF	Mean Ellenberg indicator value of soil moisture
MWD	Mean weight diameter
SOF	Saturation excess overland flow
SOM	Soil organic matter

CHAPTER 1

General Introduction

Trees contribute to a number of important ecosystem services through their influence on soil properties and functions, including water regulation and water purification. By enhancing soil hydraulic properties, trees can increase the infiltration of water into the soil and reduce surface runoff that may carry pollutants into streams and rivers. Consequently the establishment of streamside tree buffer zones and strategically planted woodland is increasingly being promoted to mitigate the environmental impacts of diffuse pollution and localised flooding.

1.1 Ecosystem services

For many decades it has been recognised that human beings derive multiple benefits from natural ecosystems, collectively referred to as ecosystem services. The importance of these services was highlighted by The Millennium Ecosystem Assessment (MEA, 2005), which was compiled to assess the state of the world's ecosystems, their link with human well-being and the consequences of change, in order to provide the information required by decision makers to manage ecosystems sustainably. The Millennium Ecosystem Assessment identified 21 ecosystem services and categorised them into four main groups: provisioning services were defined as 'the products obtained from ecosystems', such as food and fresh water; regulating services are the 'benefits obtained from the regulation of ecosystem processes', such as water regulation and water purification; and cultural services are the 'non-material benefits obtained from ecosystems'. The fourth category, supporting services, describes the 'services that are necessary for the production of all other ecosystem services' and includes soil formation, primary production and nutrient cycling (Alcamo *et al.*, 2003; MEA, 2005). Although decision makers tend to focus on the conservation and restoration of existing ecosystems to protect a range of ecosystem services, there is increasing interest in engineering ecosystems to provide specific services to manage environmental impacts (Palmer *et al.*, 2013).

1.2 Environmental impacts on water quality and quantity

Diffuse pollution and flooding, caused by surface runoff from agricultural and urban environments, has become an increasing problem in Europe and many other parts of the world as a result of agricultural intensification and urban expansion since around the mid-twentieth century. In agricultural environments, high pasture stocking rates and the use of heavy machinery can compact the soil, reducing soil hydraulic conductivity and water storage capacity (Drewry & Paton, 2005; Pietola *et al.*, 2005; Raper, 2005) and increasing the likelihood of overland flow that can lead to soil erosion and the transfer of sediments, nutrients and pesticides into watercourses. This, in turn, can lead to the eutrophication of rivers, lakes and coastal waters and can impact on drinking water supplies (Stoate *et al.*, 2001). Urban environments increase surface runoff by creating impermeable surfaces. Seto *et al.* (2011) estimated that the area of urban land in Europe expanded at a rate of 2.5% between 1970 and 2000 and Ellis (2013) reported that many inner London boroughs have lost an average of 10-20% of vegetated garden and open space to hard surfacing over the last decade. According to Ellis & Mitchell (2006), urban drainage is the third major source of water pollution in Scotland. In an attempt to improve water quality, the European Union issued the Water Framework Directive (EC, 2000), in October 2000. This directive requires that surface waters in all EU member states must achieve good ecological and chemical status within set timescales. This was followed in 2007 by the EU Floods Directive (EU, 2007), which requires member states to take ‘adequate and coordinated measures’ to reduce flood risk.

1.3 Using trees to provide ecosystem services

Vegetation plays an important role in the hydrological cycle through rainfall partitioning and evapotranspiration and by influencing soil hydraulic properties (figure 1.1). Forests are generally associated with higher rates of evapotranspiration (Thomas & Packham, 2007) and infiltration (Wood, 1977; Zimmermann *et al.*, 2006; Gonzalez-Sosa *et al.*, 2010; Agnese *et al.*, 2011; Archer *et al.*, 2013) and, consequently, lower surface runoff generation, than other land uses and vegetation types; trees can therefore contribute to the ecosystem services of water purification and water regulation and there is increasing interest in using them to tackle the problems of diffuse pollution and flooding. Forest buffers are now commonly used in many parts of the world to reduce surface runoff from agricultural land (Lowrance *et al.*, 1997; Borin *et al.*, 2005; Udawatta *et al.*, 2008) and it has been suggested that forest has an important part to play in meeting Water Framework Directive targets

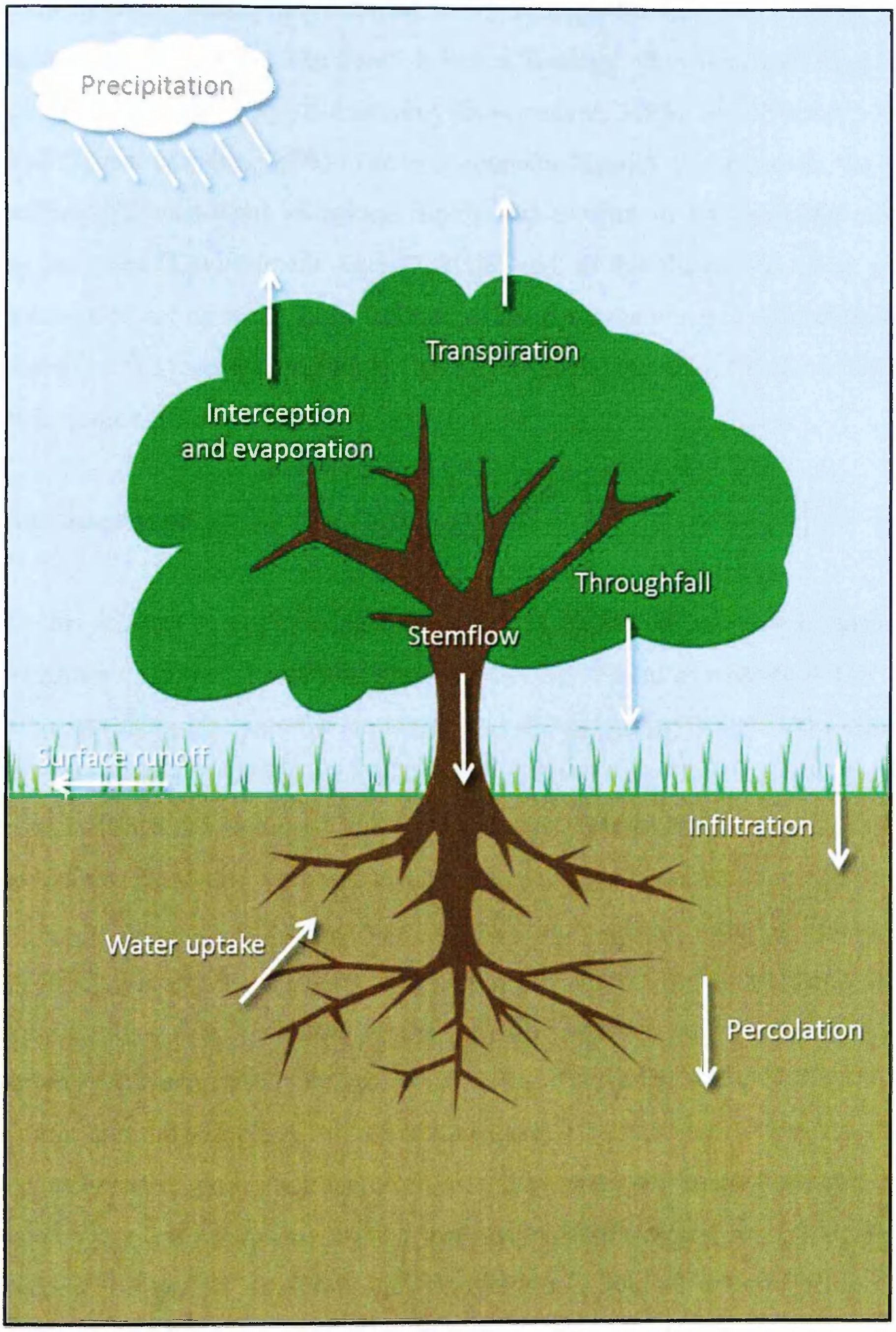


Figure 1.1. The role of vegetation in the hydrological cycle.

(Fogg, 2005; Defra, 2007). In the UK, government strategies to manage forest for the protection of water quality are outlined in ‘A Strategy for England’s Trees, Woods and Forests’ (Defra, 2007), ‘The Scottish Forest Strategy’ (Scottish Executive, 2006), ‘Woodlands for Wales’ (Welsh Assembly Government, 2009) and ‘Northern Ireland Forestry’ (Forest Service, 2006). The Environment Agency recommends the use of tree buffers and woodland to reduce runoff and erosion in its guidelines for best farming practices (Environment Agency, 2008) and, as this thesis was being written, former environment minister, Lord Rooker, called for large areas of upland pasture to be replaced with woodland to tackle flooding by delaying the arrival of rainfall to downstream areas (BBC, 2014).

1.4 Knowledge gaps

Despite this interest in engineering ecosystems to obtain the ecosystem services of water regulation and purification by strategic planting of trees as riparian buffer zones, or forest cover integrated into the landscape, our understanding of how trees affect soil hydraulic properties is still very limited. It has been suggested that soil hydraulic properties in forest are enhanced by larger and more extensive root systems, a higher diversity of soil fauna and a greater incorporation of organic matter from litterfall than under other land covers, which increases soil macroporosity (Pritchett & Fisher, 1987; Young, 1997; Archer *et al.*, 2013); however, while there is evidence that tree roots create preferential flow pathways in the soil that increase soil saturated hydraulic conductivity (Aubertin, 1971; Bartens *et al.*, 2008; Schwärzel *et al.*, 2012), the effect of leaf litter and the associated soil fauna is generally inferred rather than empirically tested. Furthermore, while root characteristics, associated soil fauna and litter quality and quantity are known to vary among tree species (Neirynck *et al.*, 2000; Reich *et al.*, 2005; Withington *et al.*, 2006; Kalliokoski *et al.*, 2008; Vesterdal *et al.*, 2008), few studies have investigated variation in tree species’ effects on soil hydraulic properties (*e.g.* Heiskanen & Mäkitalo, 2002; Eldridge & Freudenberger, 2005; Bartens *et al.*, 2008; Mishra & Sharma, 2010; Jost *et al.*, 2012). Another area of research that has received little attention is the effect of land use on soil hydraulic properties under trees. While there are many studies that report greater infiltration rates and saturated hydraulic conductivities under forest compared with other vegetation types (Wood, 1977; Zimmermann *et al.*, 2006; Gonzalez-Sosa *et al.*, 2010;

Agnese *et al.*, 2011; Archer *et al.*, 2013), the forests in these studies tend to be relatively undisturbed. The effect of trees on soil hydraulic properties may be much less under a more intense land use, however. Millward *et al.* (2011), for example, observed that infiltration rates were significantly lower in areas of an oak forest that were used for recreation than in public exclusion areas.

1.5 Thesis Outline

The work presented in this thesis attempts to address some of these knowledge gaps through a combination of field and laboratory based studies. The main aims were:

- to compare the effect of a broadleaf and a conifer tree species on soil surface hydraulic properties to determine if this differs between species type;

- to examine the effect of land use on soil surface hydraulic properties under forest cover;

- to investigate the key underlying mechanisms by which trees influence soil surface hydraulic properties and how this varies between tree species.

Chapter 2 describes a field based study in which the influence of tree species and land use on saturated hydraulic conductivity at the soil surface is investigated. Measured hydraulic conductivities were then compared with maximum rainfall intensities, estimated for various return periods from rainfall data collected at the field site for the Environmental Change Network, in order to determine the likelihood of surface runoff and, therefore, the effectiveness of the different vegetation covers and land uses in reducing surface runoff. The potential mechanisms by which trees influence soil surface hydraulic properties were investigated in a subsequent study described in chapter 3. In this chapter soil flow pathways were determined using a combination of ring and tension infiltrometer measurements and dye tracer tests, soil pore characteristics were quantified from x-ray computed tomography images of soil cores extracted at the field site and root characteristics, soil fauna populations and soil organic matter compared for the different treatments. The key mechanisms were then tested using a field based mesocosm experiment and a glasshouse based microcosm experiment. The mesocosm experiment, described in chapter 4, tested the contribution of tree roots and associated soil macrofauna on steady-state infiltration. The microcosm experiment, described in chapter 5, tested the contribution of leaf litter and the associated soil decomposer community on saturated hydraulic conductivity.

Although this work is presented as a traditional thesis, the individual data chapters have been written in paper format, therefore there is some repetition between the general introduction and the introductions to individual chapters; however this has been minimised as far as possible.

CHAPTER 2

Variation in soil hydraulic conductivity under different tree species and the impact of land use

2.1 Introduction

Forest soils are generally associated with higher rates of water infiltration (Wood, 1977; Zimmermann *et al.*, 2006; Gonzalez-Sosa *et al.*, 2010; Agnese *et al.*, 2011; Archer *et al.*, 2013) and, therefore, lower surface runoff (Huang *et al.*, 2003; Jordan *et al.*, 2008; Germer *et al.*, 2010; Alaoui *et al.*, 2011; Humann *et al.*, 2011; Dev Sharma *et al.*, 2013) than soils under other vegetation types; consequently trees have been identified as having a key role to play in the provision of the ecosystem services of water regulation and water purification as defined by the Millennium Ecosystem Assessment (Alcamo *et al.*, 2003). Strategic tree planting is now incorporated into many policies and guidelines for the mitigation of flooding, prevention of soil erosion and protection of watercourses from diffuse pollution from agricultural land and urban environments. In the UK, for example, the devolved governments set out strategies for forest management to protect water quality in 'A Strategy for England's Trees, Woods and Forests' (Defra, 2007), 'The Scottish Forest Strategy' (Scottish Executive, 2006), 'Woodlands for Wales' (Welsh Assembly Government, 2009) and 'Northern Ireland Forestry' (Forest Service, 2006) and the use of tree buffers and woodland to reduce runoff and erosion is recommended in the Environment Agency's guidelines for best farming practices (Environment Agency, 2008).

Despite this focus on utilising trees for improved water infiltration, our understanding of how trees affect soil hydraulic properties is still extremely limited. One area of research that remains largely neglected is the variation in species' effects. Although roots, soil fauna and soil organic matter, all of which affect soil hydrology (Aubertin, 1971; Eldridge, 1993; Edwards & Bohlen, 1996; Lado *et al.*, 2004; Schwärzel *et al.*, 2012), have been shown to vary between tree species (Neirynck *et al.*, 2000; Scheu *et al.*, 2003; Reich *et al.*, 2005; Kalliokoski *et al.*, 2008; Kasel *et al.*, 2011; Trum *et al.*, 2011; Tang & Li, 2013), studies that compare soil hydraulic properties under different tree species are still very few (table 2.1). In particular, a contrast in root characteristics, soil organic matter, and earthworm populations has previously been highlighted between broadleaved and conifer species (Ovington, 1953; Reich *et al.*, 2005; Withington *et al.*, 2006; Trum *et al.*, 2011), suggesting a possible contrasting influence on soil hydraulic conductivity between these species types.

Table 2.1. Published studies comparing soil hydraulic properties under different tree species.

Author(s) (year)	**Location of study site**	**Tree species studied and statistically significant difference**
Bartens *et al.* (2008)	Greenhouse experiment	Red maple = Black oak
Bens *et al.* (2007); Buczko *et al.* (2006); Wahl *et al.* (2003)	Kahlenberg, Germany	Scots pine = Beech
Eldridge and Freudenberger (2005)	NSW, Australia	Eucalyptus = White cypress pine
Heiskanen and Mäkitalo (2002)	Finland	Scots pine > Norway spruce
Johnson-Maynard *et al.* (2002)	California, US	Scrub oak > Chamise > Coulter pine
Jost *et al.* (2012)	nr. Kreisbach, Austria	Spruce = Beech
Mishra and Sharma (2010)	Uttar Pradesh, India	Mesquite/Forest red gum/Indian rosewood (significance not stated)
Sanou *et al.* (2010)	Burkina Faso, West Africa	Baobab = Néré

Forest management can also have an impact on soil hydrology, but this area of research has, likewise, received little attention. With widespread land use change occurring over the last few decades, particularly in tropical regions (Godsey & Elsenbeer, 2002), researchers have focused on the hydrological consequences of forest conversion to grazed pasture or arable land (Wood, 1977; Burch *et al.*, 1987; Lorimer & Douglas, 1995; Zimmermann *et al.*, 2010) and the effects of reforestation or afforestation (Messing *et al.*, 1997; Zimmermann *et al.*, 2006; Hassler *et al.*, 2011; Perkins *et al.*, 2012). Although these studies usually show much higher soil hydraulic conductivity under forest, the forests in these studies tend to be relatively undisturbed. More intensive forest use, however, may diminish the benefits of reduced runoff attributed to tree cover. Livestock grazing in forest and on wooded pasture (silvopasture) has been a common practice for many centuries and is widespread to this day (Sheldrick & Auclair, 2000). The Mediterranean dehesa, where livestock graze beneath scattered oak trees that provide wood, charcoal and cork, is one of the longest surviving and best known silvopastoral systems (Joffre *et al.*, 1988). More recently, integrated systems of livestock grazing with pine trees, grown to produce high-grade timber, have been developed in countries such as New Zealand, Chile and the United States (Knowles, 1991; Sheldrick & Auclair, 2000). Livestock were initially introduced to control the understorey that develops under these highly pruned and thinned trees, but the practice has gradually extended to incorporate low-density planting of trees into existing pasture (Knowles, 1991). There is, however, limited evidence to suggest that the trees in these systems do not enhance soil hydraulic properties. A study undertaken by Sharrow (2007) in an experimental agroforestry system planted with Douglas fir (*Pseudotsuga menziesii*) in the United States found no significant tree effect when infiltration rates in silvopasture were compared with those measured in pasture, and in New Zealand Yeates and Boag (1995) reported lower saturated hydraulic conductivity under radiata pine (*Pinus radiata*) silvopastures planted at various densities than under adjacent pasture.

The objectives of this study were to investigate the influence of tree species and the impact of the concurrent land use on the saturated hydraulic conductivity of the soil surface layer and to test the hypotheses that:

1) the presence of trees increases saturated hydraulic conductivity at the soil surface;

2) the influence of trees on saturated hydraulic conductivity at the soil surface is related to tree species;
3) the influence of trees on saturated hydraulic conductivity at the soil surface is modified by the concurrent land use.

Comparison between a broadleaf (sycamore) and a conifer (Scots pine) tree species was undertaken in order to determine if a species difference could be detected. The impact of land use was evaluated by comparing both silvopasture and ungrazed forest with grazed pasture, thereby testing the impact of trees under the same land use and for a less intense land use.

2.2 Methods

Field site

The field site (figures 2.1 and 2.2) used in this study is located at Glensaugh in Scotland (56° 54′ N, 02° 33′ E) and is owned and managed by the James Hutton Institute. Established in 1988, it originally formed part of the UK's National Network of Silvopastoral Experiments. These experimental sites (six in total) were created to investigate livestock productivity of an integrated sheep grazing and woodland pasture system (*i.e.* silvopasture) in a UK setting, with timber providing a potential alternative source of income. Three tree species were planted at the site, which was previously used as grazed pasture: sycamore (*Acer pseudoplatanus*), Scots pine (*Pinus sylvestris*) and hybrid larch (*Larix eurolepsis*). In addition to the silvopasture treatments, grazed pasture and ungrazed forest treatments were set up as controls. All treatments were replicated three times in a randomised block design. Although the original experiment has now ended, the majority of the treatments at the Glensaugh field site remain intact and the site continues to be maintained for scientific study. Silvopasture and pasture plots are grazed from April to October by sheep and, since 2010, occasionally by cattle, while forest plots are fenced to prevent access. Altitude across the site ranges from 140 m to 205 m, mean annual rainfall is 1168 mm and mean annual temperature is 8.0° C (2006 to 2011). Soils at the site, classed as leptic podzols or cambisols (dystric) (IUSS Working Group WRB, 2007), developed primarily on glacial drifts derived from quartz-mica-schist and are generally quite stony.

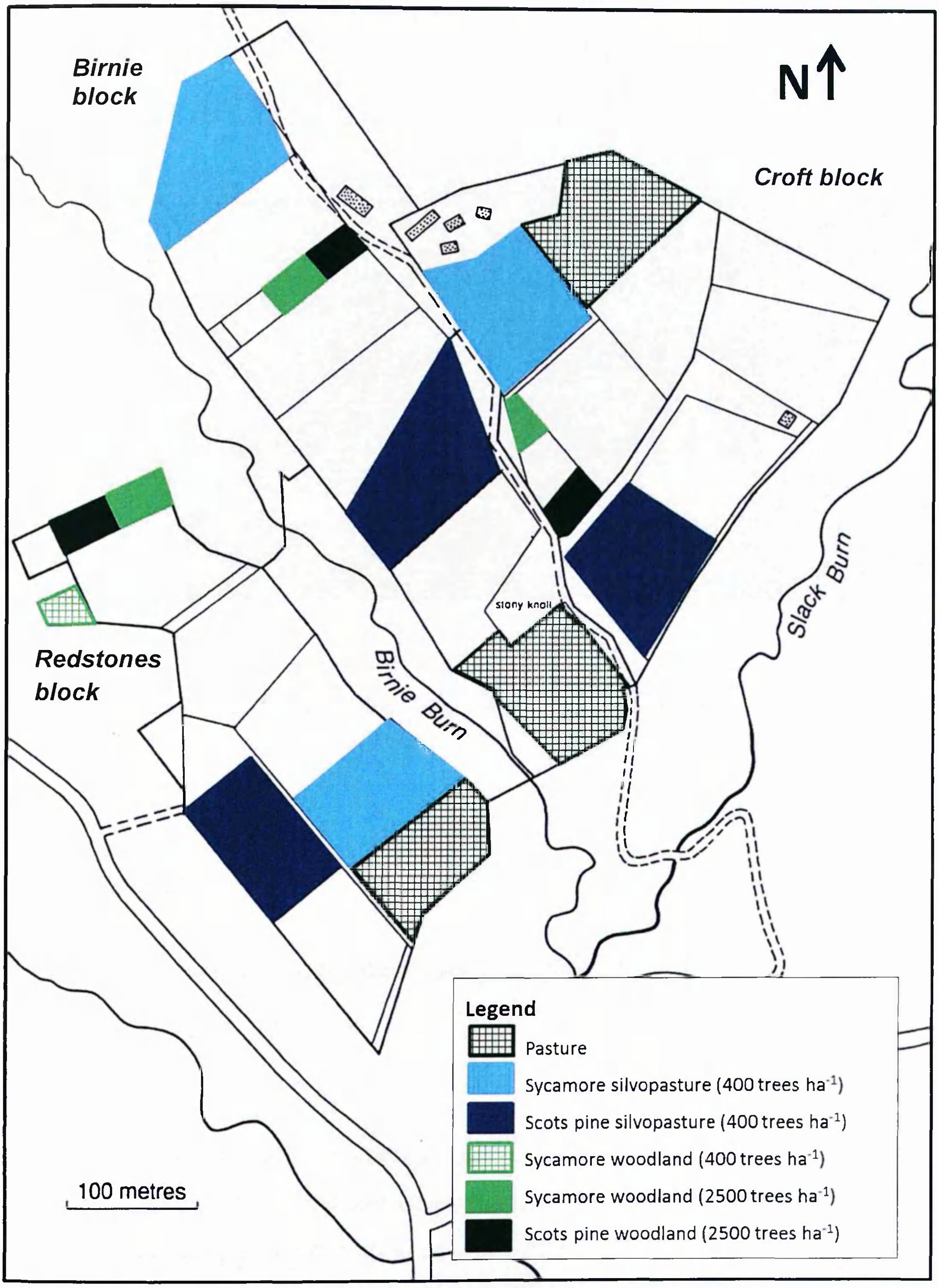


Figure 2.1. Map of Glensaugh experimental agroforestry site showing the treatments studied.

Figure 2.2. A view of the Glensaugh field site with Croft block in the foreground.

Sampling design

Sampling was initially undertaken during summer 2011 in the following five treatments:

1) grazed pasture;
2) grazed silvopasture planted with sycamore at 400 trees ha^{-1};
3) grazed silvopasture planted with Scots pine at 400 trees ha^{-1};
4) ungrazed forest planted with sycamore at 2500 trees ha^{-1};
5) ungrazed forest planted with Scots pine at 2500 trees ha^{-1}.

Trees in the silvopasture and forest plots had been planted in equally spaced rows, oriented in a north-south direction, forming a grid pattern comprising squares of 5 x 5 m and 2 x 2 m squares respectively. For the purpose of this study, these squares were defined as potential sampling locations. To minimise edge effects, squares with one or more sides on the boundary of the plot were excluded. Squares with one or more trees missing, either because they had failed to grow, had fallen or were felled for another experiment, were also excluded. Within pasture plots a virtual grid pattern, comprising squares of 5 x 5 m, was defined to identify potential sampling locations. Once potential sampling squares were identified, six squares were randomly selected for sampling from each plot in each of the three blocks (*i.e.* eighteen sample squares per treatment). Field saturated hydraulic conductivity (K_{fs}) in the upper few centimetres of the soil profile was then determined from infiltration measurements undertaken in the centre of each sample square.

Silvopasture and forest treatments were planted at different densities, so a follow up study was carried out in April 2012 to separate the influence of land use (*i.e.* grazed vs. ungrazed) from the influence of tree density, using the following treatments:

1) grazed silvopasture planted with sycamore at 400 trees ha^{-1};
2) ungrazed forest planted with sycamore at 400 trees ha^{-1};
3) ungrazed forest planted with sycamore at 2500 trees ha^{-1}.

Since the ungrazed forest planted with sycamore at 400 trees ha^{-1} treatment had been removed from one block and was considered unsuitable for study in a second, measurements (six per plot) were undertaken in a single block (Redstones) only. Without block replication this second study lacked the robustness of the first; however it provided useful additional information to aid interpretation of those results.

Field saturated hydraulic conductivity

K_{fs} was determined from measurements obtained using a small single ring infiltrometer and the pressure infiltrometer method described by Reynolds and Elrick (2002). Infiltration rate was timed until a quasi-steady rate was achieved and then K_{fs} (m s^{-1}) was derived using a single-head analysis (Reynolds & Elrick, 1990) as follows:

$$K_{fs} = \alpha^{*}GAR_1/[a(\alpha^{*}H_1 + 1) + Ga^{*}\pi a^2]$$

where α^* is a soil texture-structure parameter, estimated to be 12 m^{-1} from Reynolds *et al.* (2002), A (m^2) is the cross-sectional area of the infiltrometer reservoir, R_1 (m s^{-1}) is the quasi-steady rate of fall of water in the reservoir, a (m) is the inner radius of the ring (minimum 10 cm, maximum 20 cm), H_1 (m) is the head of water maintained in the ring (at least 5 cm) and G is a dimensionless shape factor. G is calculated using the equation (Reynolds & Elrick, 1990):

$$G = 0.316(d/\mathrm{a}) + 0.184$$

where d (m) is the depth of insertion of the ring into the soil (minimum 3 cm, maximum 10 cm). Two ring sizes (inner diameters 10 and 11 cm) were used in this study, inserted up 7 cm into the soil profile. Since K_{fs} is dependent on both the properties of the soil and the viscosity of the fluid (Hillel, 1998), and the viscosity of water varies with temperature, the temperature of the water was measured at the time of each test and a correction factor applied to determine K_{fs} at 20° C. If no drop in the water level in the infiltrometer reservoir was observed within the first hour of observation then measurement was abandoned and K_{fs} recorded as too low to be measureable.

Statistical analysis of K_{fs} data

K_{fs} data were analysed using R version 3.0.0 (R Core Team, 2013). The raw data were transformed in order to satisfy the assumption of normality (confirmed by the Shapiro-Wilk test) required by the statistical tests for differences between treatments, ensuring that observations within each treatment met this assumption Zuur (2010). A logarithmic transformation was applied to the data from the initial study, while a square root transformation was applied to the data from the follow up study. A linear mixed-effects model was used to test data from the initial study as the hierarchical structure, created by grouping observations by block, can be easily accounted for by setting block as the random factor. This type of model is also able to account for the heteroscedasticity (unequal variance) and missing values in this dataset. The follow up study, undertaken in a single block, was tested with a linear model. The *post hoc* Tukey test provided *p*-values for comparison between individual treatments. K_{fs} values collected in the follow up study were also compared with those previously collected in the same plots, using the Mann-Whitney test, in order to test for temporal differences.

Rainfall intensity-frequency analysis

Meteorological measurements are recorded at Glensaugh by the Environmental Change Network. Rainfall is recorded in hourly intervals by tipping bucket gauge, enabling a rainfall intensity-frequency analysis to be carried out for comparison with K_{fs}. Maximum rainfall intensities (I_{max}) for 2, 5, 10 and 50 year return periods were estimated from annual maximum hourly rainfall values recorded between 1995 and 2010 (Environmental Change Network, 2012) using the Weibull formula (Shaw, 1994).

Estimation of soil moisture patterns

Initial soil moisture is also a key driver of surface runoff during storm events. Prevailing soil moisture conditions for each of the treatments was estimated by determining ecological indicator values from species data collected during a ground flora survey undertaken in summer 2010. 2 x 2 m quadrats were placed approximately in the centre of each plot and all species within the quadrat identified. Mean Ellenberg indicator values for soil moisture (mF) were then calculated by averaging the indicator values of each species present in the quadrat (Hill *et al.*, 1999). Mean values are sometimes weighted to take account of the abundance of species present; however there is often little difference in estimated values between methods (Ertsen *et al.*, 1998; Schaffers & Sýkora, 2000) and, according to Diekmann (2003), weighted averages may be more affected by random spatial variations in species composition than by differences in underlying conditions.

2.3 Results

Distribution of the K_{fs} data

Distributions of the K_{fs} data demonstrated a positive skew, in common with many other studies (Talsma & Hallam, 1980; Chappell & Franks, 1996; Zimmermann *et al.*, 2006; Bonell *et al.*, 2010). The degree of skew varied between treatments (table 2), with forest treatments showing less skew than silvopasture treatments planted with the same tree species. A logarithmic distribution was found to best describe the K_{fs} data collected in the initial study, while a square root distribution was found to be more probable for the data collected in the follow up study. Treatments also showed differences in variance, both in the initial and follow up study, with lower variance in K_{fs} observed in the grazed treatments than the ungrazed treatments (figure 2.3). Application of Levene's test showed that, in the case of the initial study, this difference was highly significant ($p = 0.0009$).

Comparison of mean K_{fs} values

In the initial study K_{fs} was found to be significantly enhanced by the forest treatments (table 2.2), with mean values one and two orders of magnitude higher in the sycamore and Scots pine forests respectively compared with pasture ($p < 0.00001$). Forest treatments were also found to be significantly different from each other ($p < 0.00001$). However, no tree effect was observed in the silvopasture treatments, which had a similar mean K_{fs} to pasture and also significantly lower mean K_{fs} than the forest treatments ($p < 0.00001$).

The silvopasture treatments differ from the forest treatments in both more intense land use and lower planting density, so the effect of both were investigated in the follow up study (table 2.2). While no difference was observed between ungrazed sycamore plots planted at 400 trees ha^{-1} (equivalent to the silvopasture planting density) and 2500 trees ha^{-1} (forest), the ungrazed plot planted at 400 trees ha^{-1} had significantly greater K_{fs} ($p < 0.05$) than the grazed plot planted at the same density (silvopasture), with the same order of magnitude difference that was previously observed between silvopasture and forest (table 2.2).

Although saturated hydraulic conductivity can exhibit temporal as well as spatial variation (Bonell *et al.*, 2010), comparison of the K_{fs} data collected in the sycamore forest and silvopasture treatments for the follow up study were not found to be significantly different from the observations recorded in the same plots in the previous year.

Table 2.2. Summary of statistics for K_{fs} recorded at the soil surface at Glensaugh, Scotland.

	n	**Mean (mm hr^{-1})**	**Variance (x 10^{-9} mm hr^{-1})**	**Skewness**
Initial study				
Pasture	18	32 ^{a}	0.55	1.57
Sycamore silvopasture	15	24 ^{a}	0.40	2.03
Scots pine silvopasture	17	19 ^{a}	0.37	1.32
Sycamore forest	18	379 ^{b}	47.17	1.80
Scots pine forest	18	1239 ^{c}	98.15	0.63
Follow up study in plots planted with sycamore				
Grazed (400 stems ha^{-1})	6	46 ^{a}	1.80	1.62
Ungrazed (400 stems ha^{-1})	6	433 ^{b}	6.21	0.43
Ungrazed (2500 stems ha^{-1})	6	386 ^{b}	6.01	-0.44

Mean values given are the geometric mean for the initial study and the back transformed mean from the square root transformed data for the follow up study.
Significant differences in mean K_{fs} ($p < 0.05$) within each study are indicated by different letters.

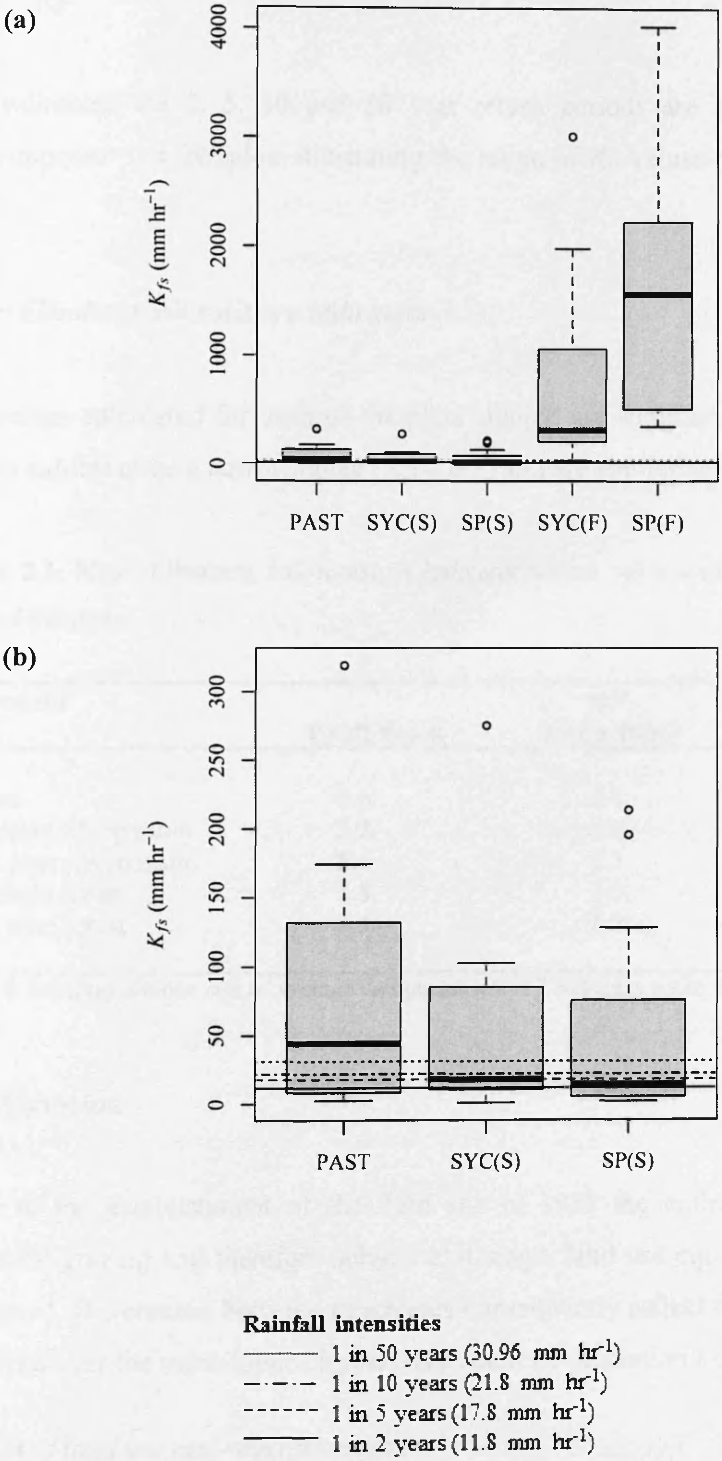


Figure 2.3. Boxplots showing the median and range of K_{fs} recorded at Glensaugh in the initial study for (a) pasture [PAST], sycamore silvopasture [SYC(S)], Scots pine silvopasture [SP(S)], sycamore forest [SYC(F)] and Scots pine forest [SP(F)] treatments and (b) the grazed treatments only. Superimposed lines show rainfall intensities for selected return periods.

I_{max} vs. K_{fs}

I_{max} estimated for 2, 5, 10 and 50 year return periods are shown in figure 2.3, superimposed over boxplots illustrating the range of K_{fs} values recorded in the initial study.

Mean Ellenberg soil moisture indicators

mF values calculated for each of the plots studied are summarised in table 2.3. The values exhibit quite a narrow range (5.2 – 6.3) and are similar across treatments.

Table 2.3. Mean Ellenberg soil moisture indicator values calculated for species present in each of the plots.

Treatment		**mF**	
	Croft Block	**Birnie Block**	**Redstones Block**
Pasture	5.6	5.4	5.5
Sycamore silvopasture	5.2	5.2	6.0
Scots pine silvopasture	5.4	5.3	5.4
Sycamore forest	5.8	5.5	5.8
Scots pine forest	6.3	5.3	5.6

mF = 5 indicates a moist site of average dampness; mF = 7 indicates a site that is constantly moist or damp.

2.4 Discussion

Prior to the establishment of the field site in 1988 the entire area was used for livestock grazing and therefore subject to a single land use equivalent to the pasture treatment. Differences between treatments consequently reflect the changes that have occurred over the subsequent 23 years as a result of vegetation and land use effects.

Impact of land use and vegetation type

While mean K_{fs} in forest soils was at least an order of magnitude greater than in pasture soils, trees in the silvopasture treatments had no observable effect on K_{fs}. In addition, variance of K_{fs} was found to be significantly lower in silvopasture and pasture compared with forest. An increase in saturated hydraulic conductivity under ungrazed forest, but not silvopasture, was also observed by Sharrow (2007), who attributed the difference to the presence of livestock grazing in the silvopastures. Grazing livestock can exert considerable pressure on the soil surface, causing compaction, resulting in reduced porosity and infiltration (Abdel-Magid *et al.*, 1987; Wheeler *et al.*, 2002; Castellano & Valone, 2007). Willatt and Pullar (1984), for example, recorded a hoof pressure of 83 kPa for sheep and estimated that this could rise to 200 kPa when the animal is walking and only two or three hooves are in contact with the ground. An order of magnitude difference in mean K_{fs} between forest and pasture soils, both in temperate and tropical regions, has been reported by other studies (Alegre & Cassel, 1996; Zimmermann *et al.*, 2006; Gonzalez-Sosa *et al.*, 2010; Agnese *et al.*, 2011; Archer *et al.*, 2013); however, this order of magnitude difference has similarly been observed between grazed and ungrazed grassland (Willatt & Pullar, 1984). This suggests that, while a greater influence of land use may mask any tree influence on soil hydraulic properties, it is possible that increased K_{fs} in the forest merely reflects the recovery of soil properties in the absence of grazing. A number of studies have shown improvements in infiltration following cessation of grazing (Nie *et al.*, 1997; Castellano & Valone, 2007; Hassler *et al.*, 2011); however, there are very few studies that compare the influence of trees with other vegetation under a similar and less intense land use and these have had mixed results. Chandler and Chappell (2008), for example, found an increase in K_{fs} around isolated oaks (*Quercus robur*) compared with surrounding parkland, whereas Kumar *et al.* (2012), comparing tree and grass buffers found that saturated hydraulic conductivity was higher under the grass buffer. More work is needed to understand vegetation effects under different land uses and how vegetation and land use effects interact; nevertheless, when considering the influence of forest on soil saturated hydraulic conductivity and, indeed, other soil properties, it is clearly important to differentiate undisturbed forest as a land use from forest as the vegetation cover.

Influence of tree species

Although there was no difference in K_{fs} between tree species in silvopastures, K_{fs} under Scots pine forest was significantly higher than under sycamore forest. This species difference in the absence of grazing lends support to the idea that the effect of land use masks the influence of trees in silvopasture. If higher K_{fs} in forest was only the result of recovery after grazing exclusion, then similar values would be expected, regardless of tree species. While there are few studies for comparison, this observation of higher K_{fs} under a coniferous species is in accordance with a study undertaken in Cambodia by Toriyama *et al.* (2011), who found that K_{fs} was generally higher under evergreen compared with deciduous forests. Wahl *et al.* (2003), Bens *et al.* (2007) and Buczko *et al.* (2006), all studying the soil hydraulic properties for a Scots pine-beech transition in the Kahlenberg forest in Germany did not find a difference between the conifer and broadleaved species; however, the historical effect of Scots pine may still be reflected in the soil properties under the replacement beech forest, making species differences more difficult to detect.

Potential for surface runoff generation

When rainfall intensity exceeds infiltration rate surface runoff is generated by infiltration excess overland flow (IOF). Soils with low surface K_{fs} are therefore more likely to generate surface runoff. Reduced variance in K_{fs}, as observed in the grazed pastures and silvopastures, also has important implications for runoff generation since, once rainfall intensity exceeds the lower end of the range, small increases affect a much wider area. Superimposing estimated I_{max} values on measured K_{fs} values (figure 2.3) illustrates that, although not widespread, IOF is likely to be generated by high intensity rainfall events at the field site. Even during a 1 in 2 year storm, estimated I_{max} exceeds K_{fs} at the lower end of the range measured within pasture and silvopasture. In contrast, it is evident that IOF is extremely rare in the forest treatments. The range of measured K_{fs} values in Scots pine forest far exceed I_{max} for a 1 in 50 year storm and, although sycamore forest may experience some runoff during such a high intensity event, it is likely to affect only a small area.

Surface runoff is also generated when the ground becomes saturated as saturation excess overland flow (SOF). Direct measurement of soil moisture or water table level to infer the occurrence of SOF requires intensive sampling over a long time period, which was not possible for this study; however, in the absence of direct measurements, ecological indicators have proved a useful tool for summarising environmental conditions over an integrated period of time (Schaffers & Sýkora, 2000). Ellenberg indicator values of soil moisture, based on the environmental response of plant species, have been shown to correlate well with field measurements of water table level both in the Netherlands (Schaffers & Sýkora, 2000) and in the UK (Mountford & Chapman, 1993). While these values cannot be directly compared with K_{fs} values, they do allow comparison between treatments. Evapotranspiration from forests is generally higher than from pastures and evapotranspiration from conifer forests higher than broadleaved forests; according to Thomas and Packham (2007) evapotranspiration is in the order of 20% of precipitation in grasslands, 30-60% in deciduous forests and 60-70% in coniferous forests. We might therefore expect lower soil moisture under tree cover; however mF values were similar across treatments. The plots in this study covered only a small proportion of the hillslope and it may be that the water table level is determined more by the vegetation and land use characteristics of the whole hillslope, rather than a small part of it; however, these results suggest that, at this field site, differences in vegetation and land use are more likely to influence surface runoff by IOF than SOF.

Implications for land management

The results of this study demonstrate that undisturbed forest has the potential, not only to reduce surface runoff, but also the capacity to 'soak up' runoff generated further up the hillslope. However, tree planting alone may not be sufficient to mitigate overland flow. The absence of a tree effect in the silvopasture treatments suggests that forest management also needs to be considered in order to obtain the ecosystem services of water regulation and water purification. Land use can have a profound impact on soil hydraulic properties, which may mask vegetation effects. Millward *et al.* (2011) found that even recreational use lowered infiltration rates of an oak forest in Toronto, Canada by an order of magnitude. Although differences in K_{fs} between tree species were significant, comparison between K_{fs} and predicted rainfall intensities indicate that surface runoff generation in undisturbed forest is rare, regardless of species and that tree species is less important than land use.

2.5 Conclusions

Strategic tree planting to provide the ecosystem services of water regulation and water purification is based on the concept that trees enhance infiltration of water into the soil, thereby reducing surface runoff that may occur during storm events. Many studies have reported higher soil hydraulic properties under forest compared with other vegetation types. Undisturbed forest in this study also had significantly higher K_{fs} than pasture; however, trees on grazed pasture (*i.e.* silvopasture) had no effect on K_{fs}. This demonstrates the importance of differentiating between undisturbed forest as a land use and forest as a land cover when considering the influence of forest on soil properties. It also highlights the importance of taking forest management into consideration if tree buffer zones and small scale woodland are to be effective in tackling diffuse pollution and localised flooding. In undisturbed forest plots K_{fs} was found to be significantly different between tree species, suggesting that the influence of livestock masked the influence of trees in silvopastures, rather than higher K_{fs} in forest simply reflecting recovery of soil properties in the absence of grazing. Comparison with predicted rainfall intensities for storms with return periods of up to 1 in 50 years demonstrated that, by increasing K_{fs}, trees in undisturbed forest do have the potential to mitigate surface runoff and that tree species is less important than land use.

CHAPTER 3

The influence of tree species and land use on soil structure and soil flow pathways

3.1 Introduction

Trees contribute to a number of important ecosystem services through their influence on soil properties and functions. These include water regulation and water purification. Forest soils are generally associated with higher rates of water infiltration and lower surface runoff than soils supporting other vegetation types (Gonzalez-Sosa *et al.*, 2010; Agnese *et al.*, 2011; Humann *et al.*, 2011; Dev Sharma *et al.*, 2013; Archer *et al.*, 2013), and the establishment of tree buffer zones and small-scale woodland, integrated into the landscape, is now widely promoted as a tool to mitigate localised flooding and diffuse pollution from agricultural land and urban environments (Lowrance *et al.*, 1997; Defra, 2007; Environment Agency, 2008). Nevertheless, our understanding of how trees affect soil hydraulic properties and, in particular, how this varies between tree species is still very limited (chapter 2).

Soil hydraulic properties are regulated by the underlying soil structure. Pore characteristics such as size, tortuosity and connectivity all affect the rate of water flow through the soil; although it has long been recognized that pore size is of particular importance because large pores, or macropores, act as preferential pathways for water flow (Beven & Germann, 1982; Allaire-Leung *et al.*, 2000a; b). When the soil is unsaturated, water is transported through smaller pores by capillary forces, while the macropores remain filled with air (Hillel, 1998). It is only as the surrounding soil approaches saturation that the water pressure becomes high enough for water to enter the larger pores (Jarvis, 2007) and for macropore flow, which is mainly driven by the force of gravity, to begin. Macropores have a considerable impact on saturated hydraulic conductivity, even though they may make up only a small percentage of the overall porosity of the soil, because the rate of flow through them can be orders of magnitude higher than through the rest of the soil matrix. Watson and Luxmoore (1986), for example, observed that nearly 73% of saturated flow was conducted by pores greater than 1 mm in diameter in their study, although these pores accounted for only 0.04% of the total soil volume. The whole of the soil profile does not need to be saturated for macropore flow to occur, however. Localised saturation, which may be a layer just a few millimetres thick at the soil surface, will generate macropore flow (Jarvis, 2007). While the exact definition of a macropore has been considerably debated since the early 1980s (Beven & Germann, 1982), experimental evidence has shown that water pressures greater than -10 cm H_2O are required to initiate preferential flow through individual pores, which corresponds with pores sizes of an 'equivalent cylindrical diameter' > 0.3 mm (Jarvis, 2007).

Soil structure is directly influenced by trees via their root systems. Decaying tree roots create macropores, often lined with bark, that may persist for many decades (Hagedorn & Bundt, 2002), while root shrinkage and wind-induced rocking creates gaps around living roots (Carminati & Vetterlein, 2012). Roots also act as binding agents of soil aggregates, promoting a better soil structure with increased porosity (Tisdall & Oades, 1982). However, while roots generally increase soil porosity, root growth displaces soil particles, causing compaction, and there is some evidence that tree roots can increase the bulk density of the soil around them (Greacen & Sands, 1980). Trees indirectly influence soil structure by contributing large amounts of organic material to the soil in the form of litter and root exudates, which promotes soil aggregation and improves aggregate stability (Tisdall & Oades, 1982; Chaney & Swift, 1984; Bronick & Lal, 2005). The influence of trees on the soil fauna and flora (Scheu *et al.*, 2003; Reich *et al.*, 2005; Churchland *et al.*, 2013) may also indirectly affect soil structure, since soil bacteria and fungi also have a positive effect on aggregate structure and stability (Tisdall & Oades, 1982; Bronick & Lal, 2005), while the burrowing activity of larger organisms, such as earthworms, increases the macroporosity of the soil (Lee & Foster, 1991).

In a previous study, at a field site in Scotland, both tree species and land use was shown to have a significant influence on field saturated hydraulic conductivity (K_{fs}) at the soil surface (chapter 2). The aim of this study was to investigate the underlying soil structural properties contributing to the variation between tree species and land uses and to explore the possible causes of variation. Specific objectives were: (i) to characterise flow pathways under the two tree species (sycamore and Scots pine) and different land uses (grazed pasture versus undisturbed forest) using a combination of infiltration measurements and dye tracer tests; (ii) to quantify the soil pore characteristics of the different treatments based on image analysis of 3-D x-ray computed tomography (CT) scans; and (iii) to investigate the possible mechanisms involved in macropore creation by evaluating root characteristics, soil macrofauna populations and soil organic matter content.

3.2 Methods

Field site and sampling design

The study was undertaken at an experimental field site at Glensaugh in Scotland (56° 54′ N, 02° 33′ E, 140-205 m a.s.l.) in undisturbed monocultures of sycamore (*Acer pseudoplatanus*) and Scots pine (*Pinus sylvestris*) and in grazed pasture plots. The history and management of the site is described in more detail in chapter 2; however, briefly, experimental plots were established in 1988 on previously grazed pasture and replicated three times in a randomised block design. The soils at the site, which are generally quite stony and have a mainly sandy loam or sandy clay loam texture, developed primarily on glacial drifts and are classed as leptic podzols or cambisols (dystric) according to the World Reference Base for soils (IUSS Working Group WRB, 2007). The mean annual rainfall at the site is 1168 mm, mean annual temperature is 8.0° C (2006 to 2011) and altitude ranges from 140 m to 205 m.

Within the silvopasture and forest plots trees had been planted in equally spaced rows, oriented in a north-south direction, forming a grid pattern comprising squares of 5 x 5 m and 2 x 2 m squares respectively. These squares were defined as potential sampling locations for the study. To minimise edge effects, squares with one or more sides on the boundary of the plot were excluded. Squares with one or more trees missing were also excluded. Within pasture plots a virtual grid pattern, comprising squares of 5 x 5 m, was used to define potential sampling locations. Squares were randomly selected for sampling and all sampling was undertaken in the centre of each square

Dye tracer tests were carried out at the end of April 2012 in a single square within each treatment in a single block (Birnie). Infiltration measurement and soil sampling (including the extraction of intact soil cores for CT analysis) was undertaken between mid-July and early September 2012 in a total of nine sampling locations per treatment (three in each block). Soil macrofauna samples were extracted from an additional twelve sampling locations per treatment (four in each block) in May 2012 and root characteristics were determined from soil cores that had been extracted during summer 2011 in eighteen sampling locations per treatment (six in each block).

Visualisation of flow pathways using a dye tracer

A qualitative analysis of soil flow pathways was undertaken using dye tracer tests. A thin-sided metal ring, 36 cm in diameter, was carefully inserted 7 cm into the soil profile with surface vegetation and/or litter left in place. 10 litres of clean water was ponded in the ring and allowed to infiltrate fully, then 8 litres of dye solution added. The dye chosen for these tests was food grade Brilliant Blue FCF (also known as the food dye E133), diluted to 4 g l^{-1} (Flury & Flühler, 1994). This dye was chosen because it has low toxicity and is easily visible in the soil (Flury & Flühler, 1994); consequently it is now routinely used as a tracer of soil flow pathways (*e.g.* Alaoui & Goetz, 2008; Capuliak *et al.*, 2010; Alaoui *et al.*, 2011). After infiltration the dye solution was left to continue percolating through the soil for 24 hours, and then a vertical pit dug to expose the stained flow pathways in the soil profile. The pit wall was carefully cleaned with a brush to remove any smearing, a frame mounted around the profile to provide scale and the profile photographed with a digital camera.

Determination of soil hydraulic properties

K_{fs} and macropore flow were determined from infiltration measurements undertaken using single-ring and tension infiltrometers. At each sampling location the surface was prepared by trimming the vegetation with scissors to < 3 mm and/or carefully removing loose litter. A thin layer (~ 3 mm) of fine-grain moist sand was applied over a circular area equivalent to the tension infiltrometer base (8 cm diameter) in order to provide a flat surface and maintain good contact between the soil surface and infiltrometer membrane. Infiltration measurements were then undertaken using the tension infiltrometer (purchased from Soil Measurement Systems, USA) at pressure heads of -10 and -6 cm, to exclude pores with equivalent cylindrical diameter > 0.3 mm (*i.e.* macropores) and 0.5 mm respectively, until a steady-state was reached. The tensions required to exclude these pore sizes were determined from the capillarity equation (Watson & Luxmoore, 1986). The second tension was applied in order to provide the additional data required to calculate unsaturated hydraulic conductivity at the -10 cm pressure head (K_{-10}) using the method of Reynolds & Elrick (1991), with the lower pressure head being applied first to avoid errors caused by hysteresis (Reynolds & Elrick, 1991). After tension infiltration measurements were completed, the tension infiltrometer was replaced with a small, thin-sided single ring, carefully inserted into the soil. K_{fs} was then determined from steady-state ponded infiltration measurements using the single-head analysis of Reynolds & Elrick (1990). This method can be used to calculate K_{fs} using a range of ring sizes between 10 and 20 cm diameter and ring insertion depths between 3 and 10 cm. Two ring sizes (inner diameters 10 and 11 cm) were used and the depth of insertion varied between 3 and 7 cm. Since the viscosity of water varies with temperature and K_{fs} is dependent on both the properties of the soil and the viscosity of the fluid (Hillel, 1998), a viscosity correction factor was applied to determine K_{fs} at 20° C. Macropore flow, as the percentage of flux, was determined from K_{fs} and K_{-10} (Watson & Luxmoore, 1986).

Quantification of pore characteristics using x-ray computed tomography

Following infiltration testing, undisturbed soil cores were extracted from the surface of the soil profile for x-ray CT scanning. Soil cores were extracted from points immediately adjacent to the location of each infiltration test. A section of plastic drainpipe (inner diameter 65 mm, depth 70 mm) was carefully pushed into the soil, using a small hydraulic jack braced against a metal frame that was anchored to the ground with marquee ground anchors (figure 3.1). The intact soil core, encased in the plastic pipe, was then gently dug out of the soil with a hand trowel. Soil cores were secured by placing plastic caps over the ends of the pipe and taping them in place with duct tape. After transporting the soil cores to the laboratory they were stored in a refrigerator at ~ 4° C until they were scanned.

Soil cores were scanned with an X-TEK Benchtop CT 160Xi scanner set to 105 kV and 127 μA using a 5 mm copper filter. Images of each core were obtained from 2073 angular projections around the centre of rotation, with 4 frames taken at each projection to reduce noise. These images were reconstructed using CT-Pro reconstruction software (XTEK Systems Ltd, 2004) to produce a 3-D greyscale volume with a voxel resolution of 74 x 74 x 74 μm, which was then converted to a set of image stacks, representing horizontal slices through the soil core, in BMP format using the image analysis software VGStudio-Max v2.1 (Volume Graphics, Germany).

Figure 3.1. The equipment used to extract intact soil cores for x-ray CT scanning.

In order to analyse pore characteristics the image stacks were segmented, using the segmentation algorithm developed by Hapca *et al.* (2013), to produce binary images of pores and solids. Using public domain image analysis software, ImageJ version 1.47 (Rasband, 2002), the images were cropped to obtain sample volumes 512^3 voxels in size (representing a 54.4 cm^3 cube of soil from the top of the soil profile) and converted to 8-bit before the segmentation algorithm was applied. Analysis of pore characteristics was then carried out in ImageJ, using the BoneJ plugin (Doube *et al.*, 2010). First the Particle Analyser tool was used to exclude particles (*i.e.* pores) that intersected the boundaries of the image stack and this image then subtracted from the original (segmented) image stack to create a stack of images representing the continuous pore networks connected to the rest of the soil. Examples are shown in figure 3.2. Connected meso- and macroporosity (pores > 74 μm diameter) was then determined using the Histogram tool and multiplying the number of voxels

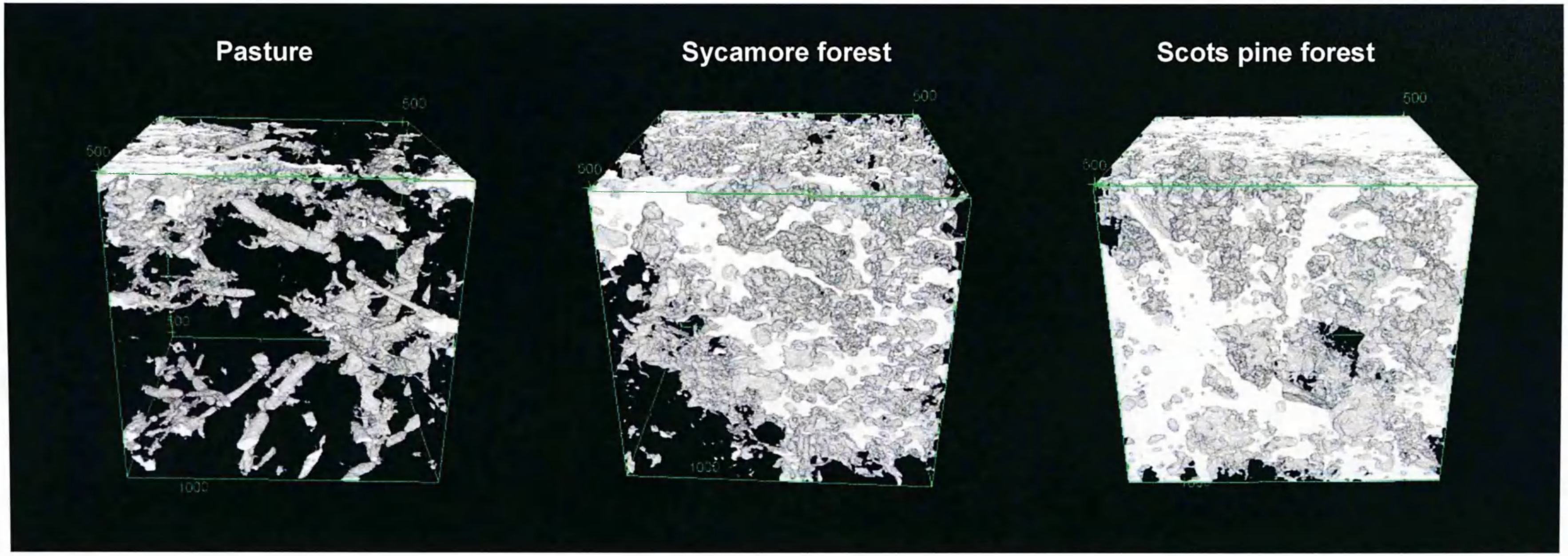


Figure 3.2. Examples of 3-D representations of connected pore networks in each of the treatment types studied, with air filled pores shown in white. Each image is 512 x 512 x 512 voxels, representing a cube of soil 38 x 38 x 38 mm.

representing pore space by the voxel size. The length of individual branches in the pore network and the number of junctions (*i.e.* points at which one branch connects with another) were determined using the Skeletonise and Analyse Skeleton tools. Pore length density and junction density were calculated by dividing the total length of branches and total number of junctions by the volume of the image stack. The mean equivalent cylindrical diameter of pores was calculated from the connected meso- and macroporosity and total length of the pore networks in the sample.

Soil sampling and analysis

A second soil core (diameter 68 mm, depth 60 mm) was extracted from the soil surface, adjacent to the first, for determination of bulk density (ρ_b) by oven-drying at 105° C for 48 hours. A sub-sample of this soil was used to determine soil organic matter (SOM) by loss on ignition (560° C overnight). Porosity (θ) was then calculated from ρ_b and SOM assuming a mineral particle density of 2.65 g cm^{-3} and SOM density of 1.35 g cm^{-3} (Rowell, 1994). An additional loose sample was obtained and transported in a rigid container to the laboratory to protect the aggregates. This sample was air-dried and aggregate stability determined following the methods described by Le Bissonnais (1996, see appendix).

Soil macrofauna sampling

Soil macrofauna were sampled using the hand sorting method. A block of soil (25 x 25 x 25 cm) was excavated with a spade, placed on a plastic sheet and hand sorted to extract earthworms and other macrofauna. Immediately after excavating the soil block, a solution of allyl-isothiocyanate (mustard oil) was sprinkled over the bottom of the remaining hole with a watering can in order to extract any remaining earthworms. The allyl-isothiocyanate solution was prepared by mixing 1 ml of allyl-isothiocyanate in 20 ml of ethanol and mixing this thoroughly in 10 litres of tap water. Sampled macrofauna were placed in a solution of 85% ethanol and taken back to the laboratory for identification and counting.

Analysis of root characteristics

Root characteristics were determined from soil cores (diameter 65 mm, depth 70 mm) extracted from within the top 10 cm of the soil profile. Roots were separated from the soil by washing in tepid water and then stored in 15% ethanol solution prior to analysis. Images of the roots were subsequently obtained by suspending them in de-ionised water on a clear acrylic tray, which was scanned on a flatbed scanner at 300 dpi. The images were analysed using WinRHIZO software (Regent Instruments Inc., Canada) to determine the total length and mean diameter of the sampled roots, and then root length density determined from the sample root length and core volume.

Statistical analysis

All statistical analyses were performed using R version 3.0.0 (R Core Team, 2013). A logarithmic transformation was applied to the root data and hydraulic conductivity data after confirming normality with the Shapiro-Wilks test. Macropore flow as a percentage of flux was negatively skewed, so values were subtracted from 100 before applying a logarithmic transform to achieve normality. Linear mixed effects models and the *post hoc* Tukey test were then used to ascertain the significance of treatment differences.

3.3 Results

Soil physical and hydrological properties

The soil physical and hydrological properties determined at the surface of the soil profile for the treatments studied are summarised in table 3.1. K_{fs} and macropore flow (as a percentage of flux) were significantly influenced ($p < 0.001$ and $p < 0.00001$ respectively) by both land use and tree species, both being higher in sycamore forest than pasture and higher in Scots pine forest than sycamore forest. However, while ρ_b and K_{-10} were significantly lower ($p < 0.01$ and $p < 0.05$ respectively) and θ significantly higher ($p < 0.01$) under Scots pine than either sycamore forest or pasture, these properties were not significantly different ($p > 0.05$) between sycamore forest and pasture.

Table 3.1. Soil physical and hydrological properties at the surface of the soil profile at Glensaugh, Scotland.

	Bulk Density (g cm^{-3})	**Porosity (m^3 m^{-3})**	K_{fs} **(mm hr^{-1})**	K_{-10} **(mm hr^{-1})**	**Macropore flow (% of flux)**
Pasture	0.70 ± 0.02^a	0.70 ± 0.01^a	8 (2-30)^a	0.37 (0.11-0.93)^a	95.13 (55.49-99.48)^a
Sycamore forest	0.67 ± 0.03^a	0.71 ± 0.02^a	665 (73-2920)^b	0.39 (0.11-1.12)^a	99.94 (99.85-99.98)^b
Scots pine forest	0.53 ± 0.04^b	0.77 ± 0.02^b	1723 (691-4277)^c	0.13 (0.01-0.62)^b	99.99 (99.93-100.00)^c

Values given are the arithmetic mean ± the standard error for bulk density and porosity, the geometric mean followed by the range for K_{fs} and K_{-10} and the backtransformed mean from the normalised distribution followed by the range for macropore flow.
Significant differences between treatments ($p < 0.05$) are indicated by different letters.

Dye tracer tests

Visual inspection of the soil profiles stained with Brilliant Blue FCF dye showed very different patterns of staining between the pasture (figure 3.3) and forest (figures 3.4 and 3.5) treatments. In the pasture the dye was restricted to a layer a few millimetres deep at the soil surface, a single isolated earthworm burrow deeper in the soil and a few fine roots below it. In the forest treatments the dye formed a heterogeneous pattern that extended over a greater proportion of the soil profile than in the pasture and to a greater depth. In both forest profiles the dye followed pathways formed by interaggregate voids and some fine roots, although larger roots (> 2 mm) and some larger biopores remained unstained. In the sycamore forest profile the dye formed distinct individual vertical pathways from the soil surface, which spread out deeper in the soil, while vertical plumes extending from the soil surface were evident in the Scots pine profile.

CT measured soil pore characteristics

Soil pore characteristics (pores > 74 μm equivalent cylindrical diameter) determined from image analysis of x-ray CT scans of soil cores are summarised in table 3.2. Connected meso- and macroporosity in forest treatments was significantly higher than in pasture (sycamore: $p < 0.01$; Scots pine: $p < 0.001$) and Scots pine forest had significantly higher connected meso- and macroporosity than sycamore forest ($p < 0.001$). The mean size of pores, based on equivalent cylindrical diameter, was also significantly higher in forest than in pasture (sycamore: $p < 0.05$; Scots pine: $p < 0.01$), although there was no significant tree species effect ($p > 0.05$). Pore length density in sycamore forest was similar to pasture ($p > 0.05$); however, tree species had a significant effect, with significantly higher pore length density in Scots pine forest than either sycamore forest or pasture ($p < 0.01$). Although junction density was higher in Scots pine forest than sycamore forest and higher in sycamore forest than in pasture, differences were not significant ($p > 0.05$).

Biopores, created by roots or the burrows of soil macrofauna, tend to be more circular, follow less tortuous pathways and have more regular surfaces than inter-aggregate pores. Visual inspection of 3-D volumes (examples shown in figure 3.2) and 2-D

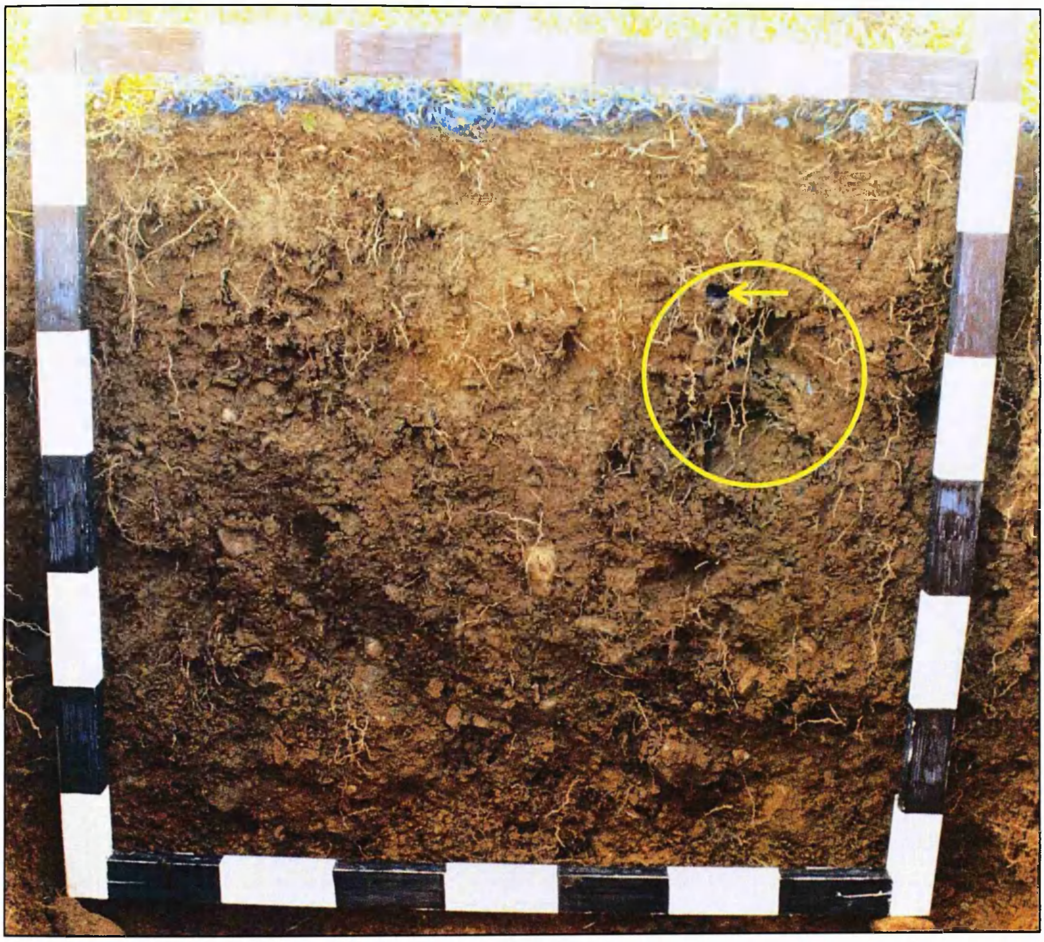


Figure 3.3. Photograph showing the pattern of dye staining of the soil profile exposed in the pasture treatment, with areas of dyed soil circled in yellow and a stained earthworm burrow indicated by the yellow arrow. The surrounding frame is marked in 5 cm increments for scale.

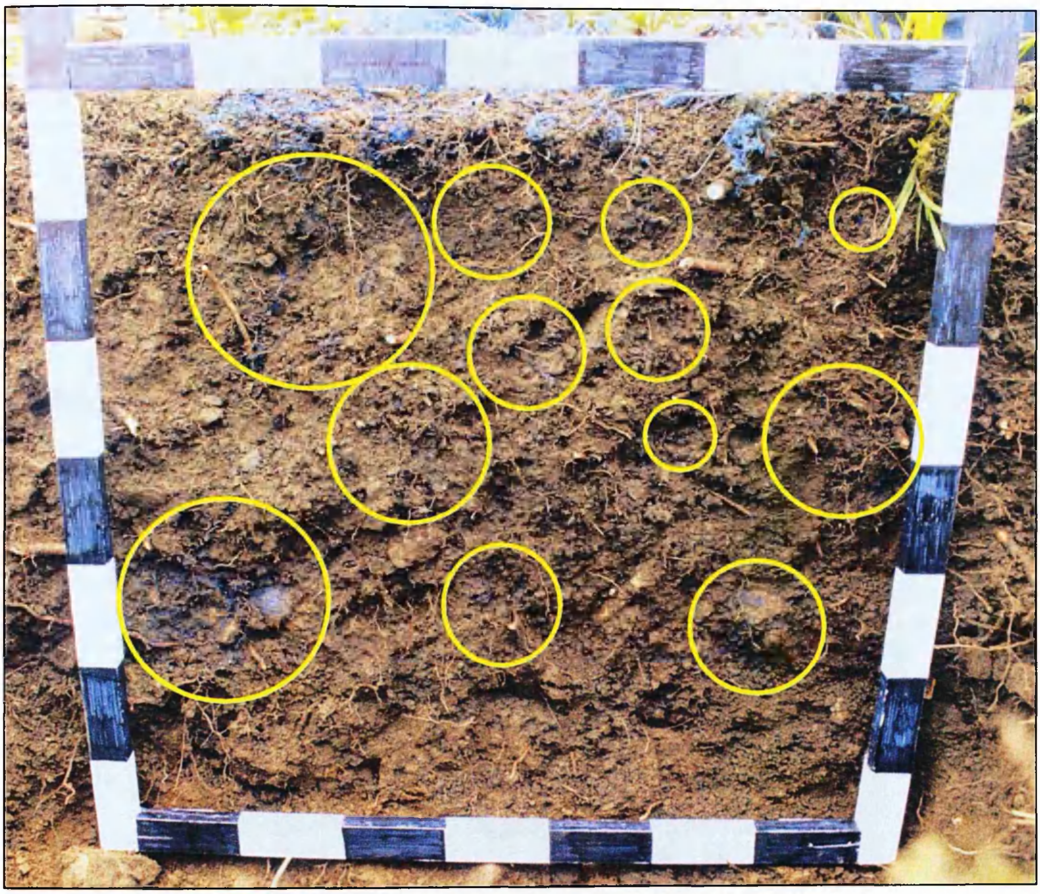


Figure 3.4. Photograph showing the pattern of dye staining of the soil profile exposed in the sycamore forest treatment, with areas of dyed soil circled in yellow. The surrounding frame is marked in 5 cm increments for scale.

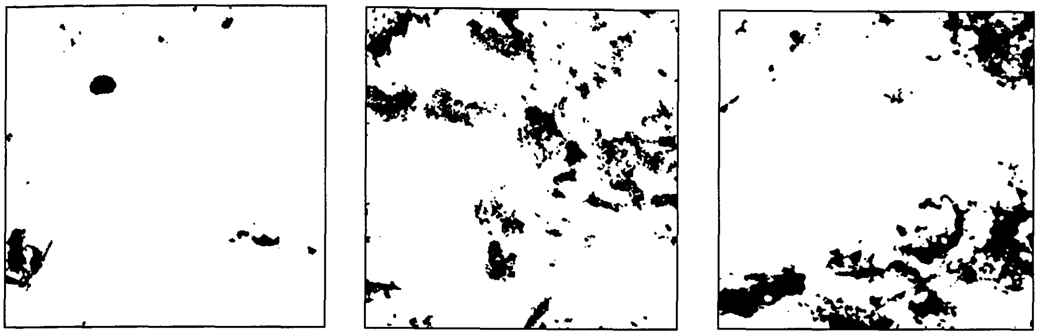
Figure 3.5. Photograph showing the pattern of dye staining of the soil profile exposed in the Scots pine forest treatment, with areas of dyed soil circled in yellow. The surrounding frame is marked in 5 cm increments for scale.

Table 3.2. CT measured pore characteristics (pores > 74 µm diameter) of soil cores extracted from the surface of the soil profile at Glensaugh, Scotland.

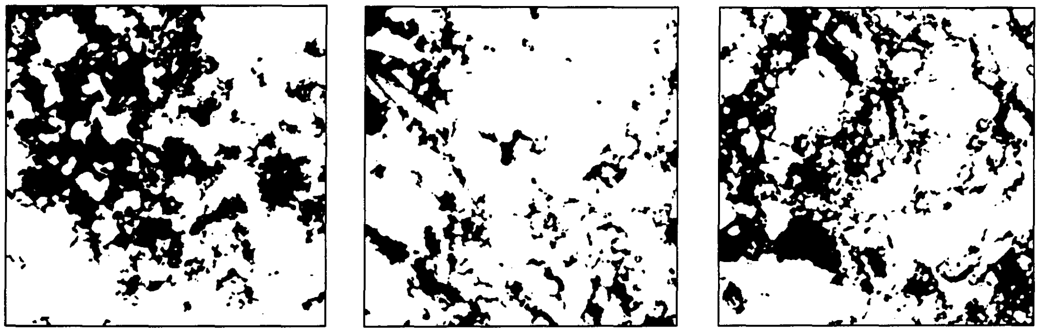
	Connected meso- and macroporosity (m^3 m^{-3})	**Mean equivalent cylindrical pore diameter (mm)**	**Pore length density (m m^{-3})**	**Junction density (number m^{-3})**
Pasture	0.12 ± 0.03^a	0.47 ± 0.02^a	810 ± 216^a	341 910 077 ± 69 953 028
Sycamore forest	0.20 ± 0.02^b	0.56 ± 0.02^b	845 ± 136^a	449 147 302 ± 87 224 946
Scots pine forest	0.32 ± 0.02^c	0.54 ± 0.02^b	1470 ± 169^b	510 044 857 ± 50 057 422

Values given are the arithmetic mean ± the standard error (the mean equivalent cylindrical pore diameter is the mean of sample means).
Significant differences between treatments ($p < 0.05$) are indicated by different letters.

Pasture



Sycamore forest



Scots pine forest

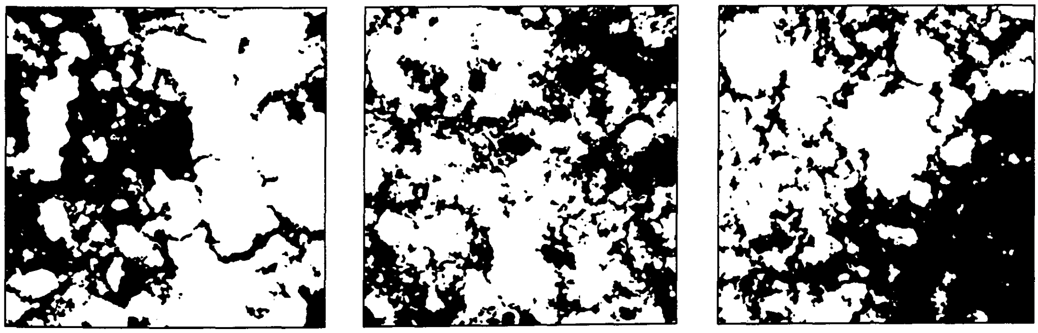


Figure 3.6. 2-D image slices (3.79 x 3.79 cm and ~ 1 cm below the soil surface) created from x-ray CT scans of the soil cores extracted from one of the experimental blocks (Birnie) at Glensaugh, Scotland. Air filled pore space is shown in black.

Table 3.3. Soil organic matter and root characteristics in the near surface of the soil profile at Glensaugh, Scotland.

		Root characteristics	
	SOM (%)	**Root Length Density (m m^{-3})**	**Mean diameter (mm)**
Pasture	13.7 ± 0.7^{a}	579 122 (362 781 - 821 641)^{a}	0.23 (0.21 - 0.25)^{a}
Sycamore forest	14.2 ± 0.5^{a}	93 559 (43 683 - 241 764)^{b}	0.30 (0.24 - 0.35)^{b}
Scots pine forest	18.9 ± 1.3^{b}	33 366 (2 357 - 133 182)^{c}	0.41 (0.30 - 1.00)^{c}

Values given are the arithmetic mean ± the standard error (the mean root diameter is the mean of sample means determined from WinRHIZO).
Significant differences between treatments ($p < 0.05$) are indicated by different letters.

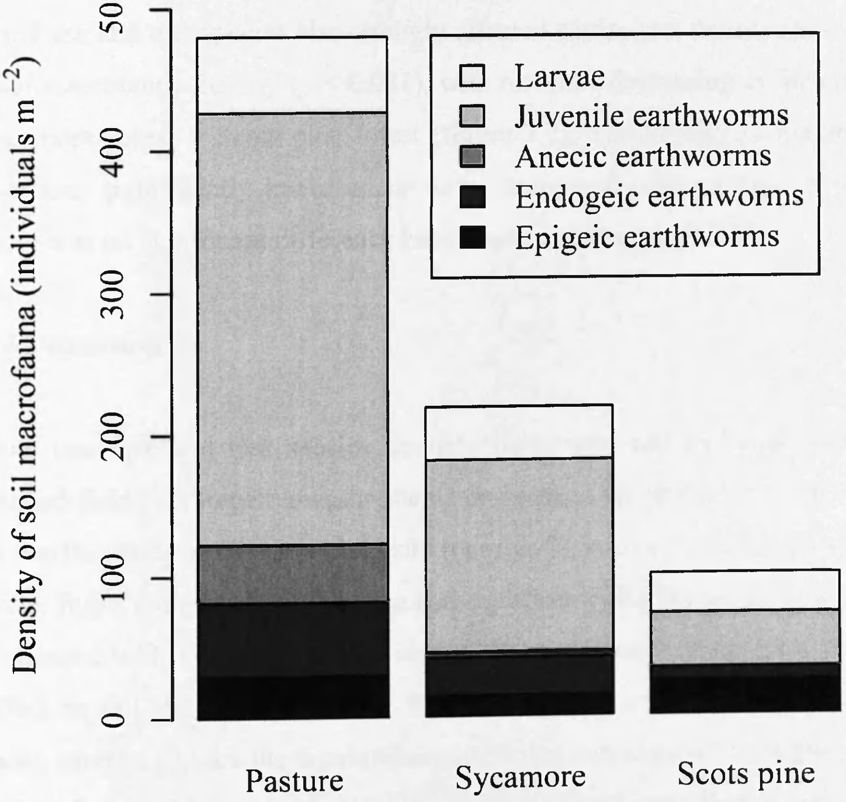


Figure 3.7. Mean density of soil macrofauna groups present in the treatments studied at Glensaugh, Scotland.

image slices (examples shown in figure 3.6) revealed that much of the pore space in the pasture samples originated from biopores, while forest treatments contained a high degree of inter-aggregate pore space.

SOM, aggregate stability, root characteristics and soil macrofauna populations

SOM (table 3.3) was significantly higher in Scots pine forest than in either sycamore forest or pasture ($p < 0.01$), but there was no significant difference between the sycamore forest and pasture treatments ($p > 0.05$). Soil aggregates were classified as very stable for all samples; therefore there was no difference between treatments. Root characteristics (table 3.3) were strongly affected by the vegetation type in the different treatments. Root length density decreased ($p < 0.001$) and root mean diameter increased ($p < 0.0001$) in the order: pasture > sycamore forest > Scots pine forest. Land use and tree species also strongly affected earthworm density ($p < 0.001$) and the total macrofauna density ($p < 0.001$), with numbers decreasing in the order: pasture > sycamore forest > Scots pine forest (figure 3.7). The density of mature earthworms was also significantly lower under forest compared with pasture ($p < 0.0001$), but there was no significant difference between tree species ($p > 0.05$).

3.4 Discussion

Both land use and tree species strongly influenced soil hydraulic properties at the studied field site. Repeat measurements undertaken for this study were consistent with the earlier study at the same field site (chapter 2), with a significantly higher mean K_{fs} under forest compared with pasture and significantly higher mean K_{fs} under Scots pine compared with sycamore. Other authors have similarly observed a strong land use effect on soil hydraulic properties, often reporting an order of magnitude difference in mean saturated hydraulic conductivity or infiltration capacity between grazed pasture and undisturbed forest (table 3.4). Fewer studies have investigated tree species effects (chapter 2, table 2.1); however Toriyama *et al.* (2011) also observed a generally higher K_{fs} under evergreen than deciduous forests in their study in Cambodia.

Soil hydraulic properties are regulated by soil pore characteristics, which influence soil flow pathways. In the following discussion the causes of variation in hydraulic

Table 3.4. Published values of mean soil bulk density (ρ_b) and infiltrability / saturated hydraulic conductivity (K_s) recorded in the topsoil of grazed pastures and undisturbed forests.

Author(s) (year)	**Location**	**Pasture**		**Forest**	
		ρ_b	**Infiltrability/K_s (mm h^{-1})**	ρ_b	**Infiltrability/K_s (mm h^{-1})**
Agnese *et al.* (2011)*	Corleone, Sicily	1.22	403	1.04	836
		1.67	40	1.24	140
		1.20	78	1.16	896
		1.46	39	1.61	131
		1.31	316	1.11	639
Archer *et al.* (2013)*		-	39	-	174
		-	21	-	119
		-	35	-	42
		-	1	-	8
Bormann & Klaassen (2008)#	Lower Saxony, Northern Germany	1.35	38	0.70	21
		1.55	6	1.00	10
Jusoff (1989)	Selangor, Malaysia	-	-	0.99	252
Millward *et al.* (2011)	Toronto, Canada	-	-	1.15	470
Sharrow (2007)	Oregon, USA	0.97	102	0.94	216
Willatt & Pullar (1984)	Victoria, Australia	1.04	22	-	-

---

[*] Study of different forest types paired with adjacent pastures at multiple locations.

[#] Study of forest on different soil types paired with adjacent pastures.

properties between the studied land uses and tree species are explored by comparing soil flow pathways, soil pore characteristics and the potential sources of macropores in the three treatments.

The effect of land use and tree species on soil flow pathways

Although they may make up only a tiny proportion of the overall soil porosity, preferential flow through macropores can dominate saturated flow (Watson & Luxmoore, 1986). In both pasture and forest treatments flow through macropores, estimated from ring and tension infiltrometer measurements, accounted for more than 95% of total flux.

Soil porosity was not significantly different between pasture and sycamore forest; however, the significantly higher K_{fs} observed in forest compared to pasture can be explained by the significantly higher proportion of saturated flow attributable to macropores. The pattern of dye staining in the excavated pasture profile indicated that flow penetrated only a few millimetres into the surface of the soil matrix and that biopores created by earthworms are the main pathways for flow deeper into the soil. There was little evidence of flow through pores created by roots near the soil surface; however, staining of fine roots beneath a stained earthworm burrow indicated that connected pathways were created by roots that intercepted the burrow deeper in the soil. In contrast, flow patterns in the forest profiles indicated a large number of connected macropores, created by interaggregate voids, decaying organic matter and some smaller roots, that carried flow from the soil surface; however, although there was some evidence of staining around smaller roots, larger roots (> 2 mm diameter) were unstained. Several unstained biopores were visible deeper in the soil profile, indicating that these pathways did not connect with the network of flow pathways at the soil surface and therefore did not contribute to K_{fs} at the soil surface, although they may become active during extended periods of rainfall as the rest of the soil profile wets up, contributing to K_{fs} deeper in the soil.

Higher K_{fs} in Scots pine forest soils compared with sycamore may be explained by both higher total porosity and greater macropore flow. The pattern of dye staining in both forest profiles indicated a large number of connected flow pathways created by interaggregate voids and some smaller roots; however, in the Scots pine a greater proportion of the upper part of the soil profile was stained, indicating a greater number of active flow pathways connected to the soil surface. Staining around partially decayed organic material incorporated into the upper part of the profile suggested that this contributed to the creation of a generally more open soil structure within the Scots pine forest soil. In the sycamore profile there was evidence that a smaller number of pathways carried flow from the soil surface deeper into the profile, where it spread out, suggesting that these pores connected with a more extensive network of macropores deeper in the soil.

The effect of land use and tree species on soil pore characteristics

Soil pore characteristics were determined from CT images of soil cores extracted from the soil surface. Image resolution restricted the size of pores detectable in the images to pores greater than 74 μm in diameter, which includes the larger capillary pores, described here as mesopores, as well as macropores (> 300 μm diameter). The volume of soil analysed was limited by the segmentation algorithm used to create the binary images that distinguish pore space from soil and organic material, which was based on a volume made up of 512^3 voxels. Although the resulting cube of soil represented by these images (with dimensions 38 x 38 x 38 mm) was quite small, results showed that it was of sufficient size to detect statistically significant differences between treatments and therefore suitable for considering differences in pore characteristics that may influence K_{fs} in the surface of a soil profile.

Both pasture and forest soils contained highly connected networks of meso- and macropores, indicated by a high density of junctions, where individual branches connect with the rest of the pore network. After isolated pore space was removed from the images, the mean connected meso- and macroporosity was found to comprise 17, 28 and 42% of overall mean porosity for pasture, sycamore and Scots pine soils respectively. Several authors have reported a significant positive relationship between saturated hydraulic conductivity and macroporosity (Mbagwu, 1995; Luo *et al.*, 2010; Agnese *et al.*, 2011) and, in this study, significantly higher K_{fs} was also associated with significantly higher meso- and macroporosity. Differences in macroporosity between land uses are unsurprising, since many authors have reported a reduction in macroporosity as a result of livestock trampling (Drewry & Paton, 2005; Houlbrooke *et al.*, 2009; Cournane *et al.*, 2011). The influence of tree species on soil macroporosity is less well studied; however, significantly higher meso- and macroporosity in the Scots pine soil compared with sycamore was consistent with a generally higher coarse pore volume observed under evergreen forest compared with deciduous forest by Toriyama *et al.* (2011). A comparison of mean pore diameters and pore length density between land uses showed that, while pore length density was not significantly different between pasture and sycamore forest, forest soils had significantly greater mean pore diameter, suggesting that the difference in meso- and macroporosity between land uses was the result of generally larger pore sizes. A comparison of forest soils, however, showed that, while mean pore sizes were similar, Scots pine forest soils had significantly higher pore length density than sycamore forest soils, suggesting that differences in meso- and macroporosity between tree species occurred as a result of a greater density of large pores in Scots pine forest.

Potential sources of macropores in pasture and forest soils

There is much evidence that both living and decayed roots can create soil macropores that form preferential flow pathways (Aubertin, 1971; Noguchi *et al.*, 1999; Devitt & Smith, 2002); however, while gaps may form around living roots (Carminati & Vetterlein, 2012), decayed roots are considered to be more important in creating macropores (Barley, 1954; Mitchell *et al.*, 1995; Perillo *et al.*, 1999). Patterns of dye staining in the soil profiles at the study site supported this hypothesis, with very little staining observed around living roots, especially in the pasture soil. While root decay can leave voids behind in the soil, the mean diameter of roots sampled in pasture was less than the diameter of pores shown to support preferential flow, suggesting that, in pasture soils roots play a limited role in creating macropores. In contrast, the significantly higher mean root diameters in the forest treatments equalled or exceeded the diameter of pores defined as macropores, indicating the greater potential for decaying roots to create macropores in forest soils. Both measured root characteristics differed significantly between tree species, with Scots pine having significantly greater mean root diameter, but significantly lower root length density than sycamore, suggesting the potential for decaying roots in sycamore forest to create more macropores but decaying roots in Scots pine forest to create larger pores. In addition to root diameter and root length density, root turnover rate will also influence the density of pores created by decaying roots in the soil. Although not measured in this study, Withington *et al.* (2006) found that roots of mature sycamore trees had a median root lifespan more than three times greater than Scots pine roots in a common garden study in Poland.

Macropores are also created by burrowing soil fauna. Earthworms can create some of the largest pores in the soil, with burrows generally measuring between one and ten millimetres in diameter (Lee & Foster, 1991). Vegetation may consequently have an indirect influence on soil macroporosity via an influence on soil macrofauna populations. Earthworm populations are generally reported to be higher in pasture than in temperate forest, and deciduous forests tend to support larger earthworm populations than coniferous forests (Edwards & Bohlen, 1996; Spurgeon *et al.*, 2013). Significantly greater earthworm density in pasture than in forest, and under sycamore compared with Scots pine, is therefore consistent with other studies and a comparison of all soil macrofauna showed the same trend. This indicates that the role of soil macrofauna in enhancing soil macroporosity is likely to be greater in pasture than in forest and that the differences in the meso- and macroporosity of the different treatments cannot be explained by differences in earthworm density or overall macrofauna density.

Organic matter, in the form of leaf and root litter, root exudates, living roots, fungal hyphae, bacteria and animal faeces, influences soil structural properties by enhancing soil aggregation and increasing aggregate stability (Tisdall & Oades, 1982; Chaney & Swift, 1984; Bronick & Lal, 2005). Organic matter can therefore affect the size of interaggregate voids formed within the soil and their destruction by compaction or aggregate breakdown during wetting (Le Bissonnais, 1996; Arvidsson, 1998). The generally high organic matter content of soils at the study site (> 13%) may therefore explain the extremely high overall soil porosities (≥ 70%) and meso- and macroporosities (12-32% of total soil volume) observed there. Values of bulk density, from which overall soil porosity was determined, typically fall within the range of 1.1 to 1.8 g cm^{-3} (McKenzie *et al.*, 2002; table 3.4); however, mean bulk densities recorded at 0-6 cm depth this site were much lower (0.70, 0.67 and 0.53 g cm^{-3} for pasture, sycamore forest and Scots pine forest respectively). Similarly low values were recorded over 0-15 cm depth at the same field site (0.55, 0.54 and 0.52 g cm^{-3} for pasture, sycamore forest and Scots pine forest respectively) by Keith *et al.* (2014). Organic matter was not significantly different between pasture and sycamore forest and therefore does not contribute to the difference in K_{fs} between land uses; however, significantly higher organic matter content in Scots pine forest soils compared with sycamore forest soils, together with visual observation of a proportion of interaggregate pores in the forest soils, suggests that higher organic matter in the Scots pine forest soil contributes the difference in meso- and macroporosity between the two tree species.

Whereas the creation of macropores is strongly influenced by vegetation, land use strongly influences the destruction of macropores. Drewry & Paton (2005), for example, reported a 15% reduction in soil macroporosity as a result of treading by sheep stocked at 1800 sheep ha^{-1} compared with an unstocked control treatment. Kulli *et al.* (2003) also observed a reduction in macroporosity as a result of compaction by farm traffic and in addition, found that the ratio of macroporosity to total porosity decreased, indicating that compaction had a greater effect on the destruction of large pores. Differences in meso- and macroporosity between pasture and forest are therefore likely to be the result of a more intense land use in the pasture. Visual inspection of both the dye stained profiles and CT images revealed a marked reduction in interaggregate voids in the pasture compared with the forest soils, even though the organic matter content of the pasture soil was similar to that in sycamore forest. In addition, the Scots pine forest soil was observed to compress underfoot (personal observation), highlighting the potential for macropores to be destroyed under a more intense land use.

3.5 Conclusions

Significantly higher saturated hydraulic conductivities at the soil surface under undisturbed forest mean that, incorporated into appropriate land management plans, the establishment of forested areas have the potential to reduce surface runoff and therefore help reduce diffuse pollution and localised flooding. Furthermore, it has been shown that tree species can significantly influence soil surface saturated hydraulic conductivity and therefore the potential of trees to infiltrate surface runoff.

This study set out to investigate potential causes of variation between land uses and tree species by studying soil flow pathways, soil pore characteristics and the potential sources of macropores in pasture, sycamore forest and Scots pine forest treatments.

Soils in all treatments were found to have a high overall porosity, containing highly connected networks of meso- and macropores. K_{fs} was found to be strongly influenced by macropore flow, with significantly higher K_{fs} corresponding with significantly higher meso- and macroporosity and macropore flow. Forest soils contained larger pores than pasture, which appear to originate from a combination of interaggregate voids and decayed roots. Large pores in pasture appear to originate mainly from earthworm burrows. It is likely that many large pores in pasture are destroyed by livestock trampling, which compacts the soil surface. Scots pine had pores of a similar size to sycamore, but a greater density of meso- and macropores, causing an overall greater meso- and macroporosity. Differences between species seem to result from larger roots and more organic matter in the Scots pine forest soil.

CHAPTER 4

Tree species influence on soil hydraulic properties: the contribution of roots and associated soil macrofauna

4.1 Introduction

It is often suggested that trees enhance soil hydraulic properties because large and extensive tree roots and higher diversity of soil fauna increases the macroporosity of forest soils (Pritchett & Fisher, 1987; Archer *et al.*, 2013). The large pores, or biopores, that are created by living and decayed roots and the activity of burrowing soil fauna can substantially increase soil saturated hydraulic conductivity and infiltration capacity because they form preferential flow pathways that transmit water many times faster than through the rest of the soil profile. Peterson and Dixon (1971, cited in Beven & Germann, 1982), for example, observed a 65% increase in infiltration capacity after the opening of a single macropore in a plot they were studying, even though it increased soil porosity by just 0.002%.

The idea of higher macroporosity in forest soils is supported by a number of studies, using several different methods. Using a dye tracer, Shougrakpam *et al.* (2010) observed a greater number of macropores under forest compared with a neighbouring paddy field; Messing *et al.* (1997) inferred higher macroporosity under forest compared with agricultural crops from the soil water retention characteristic; and, using x-ray computed tomography, Udawatta and Anderson (2008) measured significantly more macropores in undisturbed soil cores extracted from under trees than from under grass. However, while several studies have observed preferential flow pathways related to tree roots in forest soils (Aubertin, 1971; Noguchi *et al.*, 1999; Schwärzel *et al.*, 2012), the contribution of soil invertebrates to the hydraulic properties of forest soils has been less well studied (Schutz *et al.*, 2008).

Increases in soil hydraulic properties have been linked to several groups of soil macrofauna, including ants, termites and earthworms (Eldridge, 1993; Edwards & Bohlen, 1996; Léonard *et al.*, 2004). Increases in infiltration rates of between two and ten times have commonly been attributed to earthworms in grassland and arable soils (Lee & Foster, 1991), although at a reclaimed site in the Netherlands, where earthworms were previously absent, Hoogerkamp *et al.* (1983) reported an increase in infiltration capacity from just 2 mm hr^{-1} to 267 mm hr^{-1} eight to ten years after their introduction. Schutz *et al.* (2008) found a significant correlation between earthworm numbers and infiltration rates in deciduous forest in Switzerland; however, according to Edwards and Bohlen (1996) grasslands tend to support higher populations of earthworms than temperate forests so, while earthworms may contribute to the hydraulic properties of forest soils, it does not follow that improved infiltration in forest is the result of greater earthworm activity. Moreover, there is evidence that the abundance and diversity of soil fauna varies between tree species (Muys *et al.*, 1992; Neirynck *et al.*, 2000; Scheu *et al.*, 2003; Reich *et al.*, 2005). Populations of earthworms are generally reported to be lower in coniferous than in deciduous forests (Edwards & Bohlen, 1996; Spurgeon *et al.*, 2013). Likewise, differences in root architecture have been reported for coniferous and deciduous trees (Withington *et al.*, 2006), suggesting that the influence of trees on soil hydraulic properties varies between tree species and, in particular, between these species types. Despite this, the influence of tree species on soil hydraulic properties has rarely been tested (chapter 2, table 2.1).

The aim of this study was to investigate the separate and combined effects of tree roots and soil macrofauna on steady-state infiltration and to determine if these effects contribute to variation between a deciduous and coniferous species. Using soil mesocosms a field experiment was set up to test the following hypotheses:

1) The effect of roots on steady-state infiltration differs between tree species;
2) soil macrofauna contribute to higher steady-state infiltration under trees;
3) the contribution of soil macrofauna to steady-state infiltration differs between tree species.

The experiment was conducted in parallel with a study of more mature trees at a field site in Scotland (chapter 2), so the species used in this study, sycamore (*Acer pseudoplatanus*) and Scots pine (*Pinus sylvestris*), were selected to allow comparison.

4.2 Methods

Experimental design and construction

Soil mesocosms were installed at Hazelrigg environmental research station, Lancaster University (54°1′ N, 2°46′ W), which has a mean annual rainfall of 1110 mm and mean annual temperature of 9.3° C (1974 - 2011). Odum (1984) suggested the use of the term mesocosm to describe 'bounded and partially enclosed outdoor experimental setups', being intermediate between the simplified enclosed model ecosystems commonly described as microcosms and complex, real, world ecosystems as macrocosms. Unfortunately the term has become somewhat confused in the scientific literature, often being used to describe large (indoor or outdoor) microcosms (Teuben & Verhoef, 1992; Al-Maliki & Scullion, 2013). However, by allowing partial exchange with the surrounding environment, mesocosms as 'outdoor semi-controlled ecosystems' (Boyle & Fairchild, 1997) are designed to mimic natural ecosystem functioning and behaviour as closely as possible, while allowing the researcher to subtract the variables of interest. Using a factorial design, six treatments were prepared in order to test the influence of soil macrofauna and roots for two tree species. These were: sycamore, Scots pine and no tree, each either accessible or inaccessible to soil macrofauna. The mesocosms were positioned 1.5 m apart in five rows, with each row containing one complete set of randomly positioned treatments.

Mesocosms were constructed from large plastic tubs 42 cm in depth with an inner diameter of 50 cm at the top, reducing to 44 cm at the base. A series of evenly spaced openings were cut in the side, comprising three groups of eleven circular holes (48 mm diameter), inter-spaced with three groups of six vertical slits (5 mm wide x 130 mm deep), designed to allow maximum contact with the surrounding soil, while maintaining the structural integrity of the tub. Openings were offset from each other so that they would extend continuously from 4 cm below the soil surface to a point close to the base of the tub. Holes were drilled in the base for drainage. Soil fauna classified as micro-organisms have a body width < 100 µm (Bardgett, 2005), so the openings in half of the tubs were covered with a 75 µm nylon mesh to prevent the entry of meso- and macro-fauna, while the holes in the remaining tubs were covered with a 5 mm mesh, allowing access to all the soil fauna. Each tub was then partially buried in holes that had been pre-prepared with a layer of gravel in the bottom to facilitate drainage, leaving the upper rim exposed (figure 4.1). Another layer of gravel was placed in the base of the tub, over which heat sterilised Kettering loam soil (pH 7.1, 5% organic matter content and sieved to 5 mm) was packed by hand in 8 x 4 cm deep layers to a dry bulk density of 1.2 g cm^{-3}. This value of bulk density represents the upper end of the range typically reported for grassland soils and is slightly less than the density at which root penetration starts to be inhibited (Brady & Weil, 2002). After each layer was packed, the surface of the soil was lightly scarified to ensure hydraulic connectivity between the layers (Lewis & Sjostrom, 2010). When fully packed the soil in the mesocosms was level with the surface of the surrounding soil. One year old bare root sycamore trees (40 – 60 cm height) and cell grown Scots pine trees (15 – 30 cm height) were planted by removing a small plug of soil from the centre of the mesocosm, inserting the tree roots and then carefully packing some of the removed soil in the remaining gaps. Finally, gravel collars were placed around the tubs to restrict weed growth and plastic fencing erected around each tub to keep out rabbits.

Figure 4.1. Photograph showing a partially buried mesocosm prior to filling with soil. The sides are covered with a fine mesh to prevent the entry of meso- and macrofauna.

Maintenance of the experiment

The mesocosms were set up at the field site during May 2011 and remained in the ground until the end of October 2012. During the first six weeks a period of very warm, dry weather was experienced, so all of the mesocosms, including those without trees, were occasionally watered by hand, to prevent the root zone from drying out before the trees had established. Following this, the only water supply was from rainfall and groundwater. Any weeds that appeared in the mesocosms were cut at the soil surface to prevent growth while avoiding disturbance of the soil surface. Any leaf litter was removed. Some moss growth occurred, so the thickest patches were carefully lifted in February 2012. Other than this, the mesocosms remained undisturbed until sampling at the end of the experiment.

Sampling

Measurements of vertical steady-state infiltration were recorded in mid-October 2012. Bentonite clay was used to seal any gaps that had formed between the soil and the tub edge then, using the exposed upper rim of each tub as a large single ring, water was ponded to a depth of 3 cm and the time for the water level to drop by 1 cm recorded. This was repeated until a consistent time was recorded for at least three consecutive drops, indicating that a steady-state had been reached. Although some lateral divergence of flow may have occurred through the holes in the side of the tub, the error caused by lateral divergence under the sides of ring infiltrometers is usually considered negligible for large rings anyway (Bouwer, 1986).

Once infiltration measurements were complete, soil samples were taken from the near surface of the soil profile for the determination of bulk density, soil organic matter content (SOM), pH and aggregate stability, as well as classification of the soil structure. Trees were then carefully removed from the mesocosms with roots intact and the soil sorted by hand to extract the soil macrofauna.

Laboratory analysis

In the laboratory, bulk density was determined from a small, undisturbed soil core (68 mm diameter x 60 mm depth) by oven-drying at 105° C for 48 hours and a sub-sample then used to determine SOM by loss on ignition (560° C overnight). pH was measured using a soil suspension of 10 g of soil in 25 ml of de-ionised water that was allowed to settle for 12 hours. Aggregate stability was determined by following the fast wetting method described in Le Bissonnais (1996, see apendix) and aggregate structure classified according to the FAO 'Guidelines for Soil Description' (FAO, 2006). Each tree was cut at the base of the trunk and aboveground biomass ascertained from the dry weight after oven drying at 60° C until mass loss ceased. A single lateral branch root system was cut from the main root system of each tree and these roots suspended in water on a clear acrylic tray to enable them to be scanned on a flatbed scanner at 300 dpi. Root images were analysed using WinRHIZO software (Regent Instruments Inc., Canada,) to determine total root length and mean root diameter, then all lateral roots were suspended in water and displacement used to determine volume. Using this volume and the total root length determined for a single lateral branching system, the total length of all branching roots were estimated for the tree. The volume of the tap root, or root cluster of the original cell for the Scots pine trees, was then determined and total root volume calculated. All roots were oven dried at 60° C to determine total belowground biomass. Soil macrofauna were initially divided into earthworms and larvae and then earthworms identified and sorted by functional group.

Statistical analysis

Data were analysed in R version 3.0.0 (R Core Team, 2013). Normality was tested using the Shapiro-Wilks test and a logarithmic transformation applied to the K_s data before differences between treatments were tested using linear models with the *post-hoc* Tukey test. Where interaction terms were not found to be significant, model simplification was performed (Crawley, 2013). Regression analysis was also carried out using linear models to investigate relationships between variables.

4.3 Results

Soil macrofauna

Sampling of soil macrofauna (figure 4.2) revealed that a small number of individuals were present in mesocosms designed to exclude them; however, their number were significantly reduced in comparison to the accessible treatments, allowing their effect to be investigated. Within all treatments the community was dominated by juvenile earthworms, with a smaller number of mature earthworms and larvae. Endogeic species (*A. chlorotica*, *A. caliginosa*, *A. rosea*, *E. tetraedra*, *L. castaneus* and *O. cyaneum*) and epigeic species (*L. rubellus* and *S. mammalis*) were present in all treatments, while anecic species (*A. longa*) were found only in sycamore and no tree treatments. There were marginally fewer mature earthworms present under trees (p = 0.0789 for sycamore; p = 0.0695 for Scots pine) and fewer juveniles under Scots pine than in the no tree treatment (p = 0.0449).

Tree properties

Tree properties (table 4.1) demonstrated a significant species difference, with Scots pine trees having greater aboveground biomass ($p < 0.0001$), belowground biomass (p = 0.0276), total root volume (p = 0.0071) and mean root diameter (p = 0.0003) than sycamore trees, although estimated total root length was similar ($p > 0.05$) for both species. The presence of soil macrofauna had no observable effect on tree properties.

Steady-state infiltration and other soil properties

Steady-state infiltration was log-normally distributed. Variation was high for all treatments, with values ranging from tens to hundreds of millimetres per hour (figure 4.3) and differences were not found to be significant, even though the highest median value (261 mm hr^{-1}), was nearly nine times greater than the lowest (30 mm hr^{-1}). Soil macrofauna significantly increased bulk density ($p < 0.0001$; table 4.2) and altered soil structure. Mesocosms accessible to soil macrofauna all exhibited a blocky aggregate structure, while the aggregate structure in the treatments inaccessible to

Table 4.1. Tree properties (mean ± standard error).

Treatment	**Aboveground Biomass (g)**	**Belowground Biomass (g)**	**Root Volume (cm^3)**	**Estimated Total Root Length (m)**	**Mean Root Diameter (mm)**
Sycamore	24.3 ± 5.8^a	25.2 ± 7.1^a	78.0 ± 24.8^a	334.6 ± 85.5	0.41 ± 0.03^a
Scots pine	77.7 ± 18.0^b	41.6 ± 11.0^b	150.5 ± 33.3^b	350.5 ± 84.6	0.55 ± 0.06^b
Sycamore and macrofauna	23.4 ± 4.3^a	26.8 ± 8.3^a	64.4 ± 24.1^a	297.4 ± 135.2	0.42 ± 0.02^a
Scots pine and macrofauna	111.6 ± 16.2^b	49.3 ± 5.7^b	144.0 ± 17.7^b	215.3 ± 69.2	0.58 ± 0.03^b

Significant differences between treatments ($p < 0.05$) are indicated by different letters

Table 4.2. Soil properties (mean ± standard error) determined for the near surface soil of the different treatments.

Treatment	**Bulk Density ($g\ cm^3$)**	**SOM (%)**	**pH**	**MWD**
No tree	1.19 ± 0.01^a	8.76 ± 0.14	8.04 ± 0.08	0.76 ± 0.13
Sycamore	1.17 ± 0.02^a	8.51 ± 0.12	8.07 ± 0.10	0.75 ± 0.08
Scots pine	1.15 ± 0.01^a	8.99 ± 0.08	8.09 ± 0.09	0.67 ± 0.04
No tree and macrofauna	1.27 ± 0.02^b	8.58 ± 0.20	8.09 ± 0.09	0.83 ± 0.20
Sycamore and macrofauna	1.31 ± 0.03^b	8.83 ± 0.28	8.14 ± 0.06	0.59 ± 0.10
Scots pine and macrofauna	1.24 ± 0.02^b	8.87 ± 0.20	7.96 ± 0.07	0.54 ± 0.06

Significant differences between treatments ($p < 0.05$) are indicated by different letters

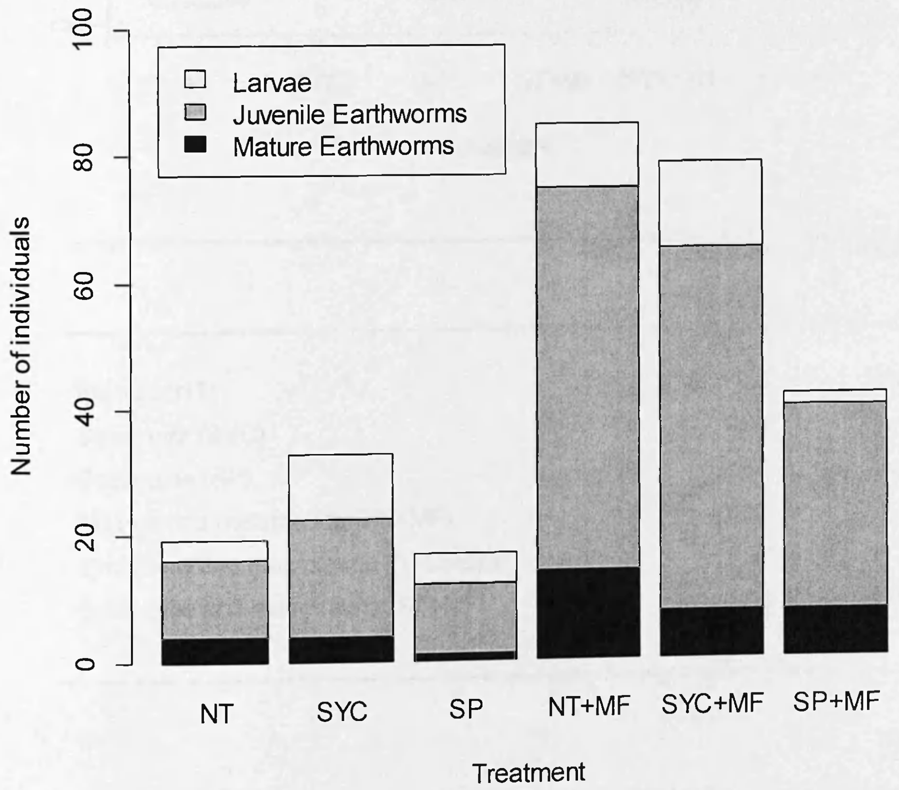
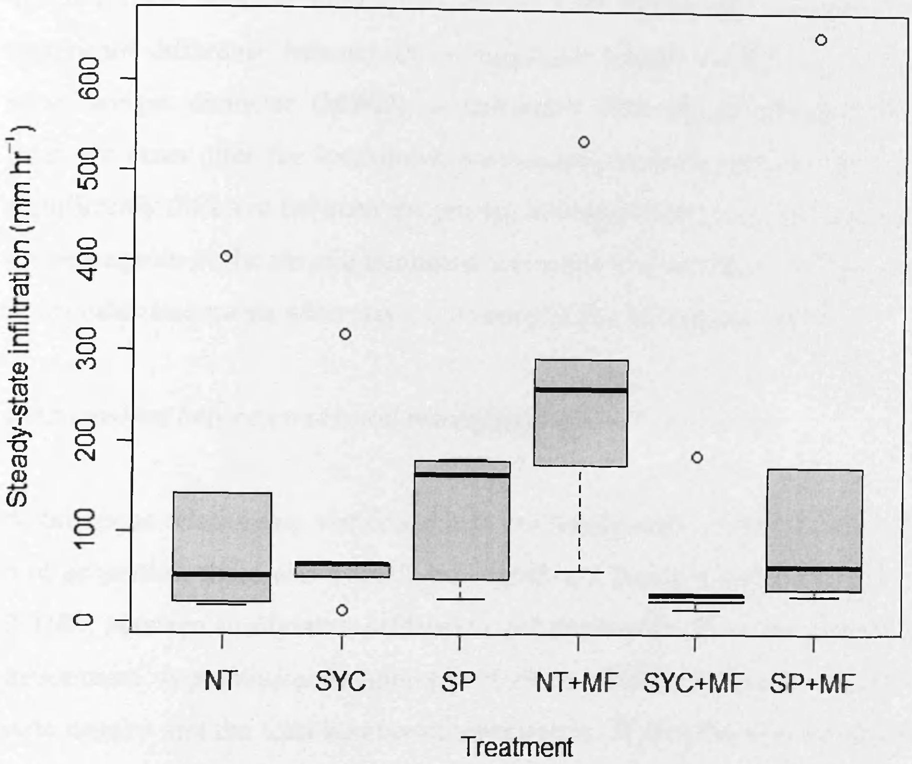


Figure 4.2. Mean number of groups of soil macrofauna present in the no tree (NT), sycamore (SYC), Scots pine (SP), no tree and macrofauna (NT+MF), sycamore and macrofauna (SYC+MF) and Scots pine and macrofauna (SP+MF) treatments (October 2012).



	Min	Median (mm hr^{-1})	Max
No tree (NT)	18	51	404
Sycamore (SYC)	13	64	319
Scots pine (SP)	26	164	181
No tree and macrofauna (NT+MF)	57	261	537
Sycamore and macrofauna (SYC+MF)	16	30	187
Scots pine and macrofauna (SP+MF)	30	63	655

Figure 4.3. Boxplots showing median and range of steady-state infiltration rates recorded for the studied treatments (October 2012).

macrofauna could be described as either granular or crumbly, having changed very little since the soil was sieved prior to the start of the experiment. Soil organic matter (table 4.2) had increased from around 5% to over 8% and pH (table 4.2) from a neutral 7.1 to a more alkaline value of around 8 in all treatments, although there was no significant difference between them. Aggregate stability (table 4.2), expressed as a mean weight diameter (MWD), is calculated from the distribution of remaining fragment sizes after the breakdown mechanism has been applied. This too was not significantly different between treatments, although medium stability was determined for aggregates in the no tree treatment accessible to macrofauna, whereas aggregates in the other treatments were classed as unstable (Le Bissonnais, 1996).

Relationships between trees, soil macrofauna and soil properties

Although no relationship was found between steady-state infiltration and either soil or root properties, there was a small, but significant, positive correlation ($r^2 = 0.16$, $p = 0.0288$) between steady-state infiltration and the number of mature earthworms in the mesocosms. A positive correlation ($r^2 = 0.66$, $p < 0.0001$) was also observed between bulk density and the total number of earthworms. Within the treatments accessible to macrofauna, the number of mature and juvenile earthworms, although not larvae, correlated positively with pH and negatively with root volume (table 4.3).

Table 4.3. Results of correlation tests for soil macrofauna.

	pH		**Total root volume**		**pH+Total root volume**	
	r^2	p	r^2	p	r^2	p
Mature earthworms	0.37	0.0158	0.29	0.0384	0.35	0.0309
Juvenile earthworms	0.39	0.0130	0.40	0.0110	0.44	0.0119
All earthworms	0.46	0.0052	0.46	0.0057	0.54	0.0038

4.4 Discussion and conclusions

Soil mesocosms were utilised by this study to investigate the separate and combined effects of tree roots and soil macrofauna on steady-state infiltration. By using different sizes of mesh to cover the sides of mesocosms, macrofauna could be prevented from re-colonising half of the treatments but, by otherwise allowing free movement of soil fauna, both the influence of tree species on soil macrofauna and the contribution of soil macrofauna to steady-state infiltration could be investigated, something that would not have been possible using a fully enclosed system. Mesocosms with a coarse mesh, and therefore designed to be accessible to the macrofauna, were successfully recolonized, with treatment differences observed and, although macrofauna were not completely excluded from mesocosms with a fine mesh, numbers were significantly reduced. While macrofauna were prevented from entering mesocosms through the soil, earthworms are known to disperse over the soil surface and there is also evidence of climbing behaviour (Eijsackers, 2011), suggesting that earthworms in the 'inaccessible' mesocosms entered over the side. Larvae populations are more difficult to manipulate as many larvae originate from eggs laid by flies and beetles. Kandeler *et al.* (1999) similarly found that mesocosms designed to exclude mesofauna reduced numbers rather than completely excluded them; however, by significantly reducing the population, it is possible to assess the effects of the organisms in question.

Root presence did not have a significant impact on steady-state infiltration, despite both tree species having produced extensive root systems. Studies have previously shown that tree roots can increase soil hydraulic properties by creating macropores that form preferential flow pathways through the soil (Aubertin, 1971; Noguchi *et al.*, 1999; Schwärzel *et al.*, 2012); however, less than a year and a half after planting, these young trees were still in the early stages of growth and it is possible that the mechanisms responsible for macropore generation were still developing. Although macropores are generated by both living and decayed roots, according to Perillo *et al.* (1999) the channels formed by decayed roots are more important. Black *et al.* (1998) suggested that root lifespan in young trees may be as little as a few days but, in a common garden study in Poland, Withington *et al.* (2006) estimated the median root lifespan of several tree species to be of the order of hundreds of days, therefore the duration of the experiment may have been insufficient for root decay to have a significant impact. Macropores associated with living roots form around the root surface (Lange *et al.*, 2009); however, juvenile trees grow rapidly, investing energy in growth rather than reproduction (Thomas, 2000), so any gaps that form around the root surface may be quickly filled by the root as it grows.

Differences in steady-state infiltration between tree species, although not significant, exhibited the same trend observed for more mature trees at Glensaugh in Scotland (chapter 2), where mean saturated hydraulic conductivity under Scots pine was an order of magnitude higher than under sycamore. Therefore, while the hypothesis that that the effect of roots on steady-state infiltration differs between tree species is not supported by the data in the present study, this non-significant difference may reflect the early stages of a developing species effect mediated by roots. Since root decay is an important source of macropores that enhance soil hydraulic properties, a possible reason for higher infiltration under Scots pine may be a faster root turnover rate. Withington *et al.* (2006) found that the median root lifespan of the Scots pine trees in their study was 236 days, whereas the median lifespan of sycamore roots was more than three times that. There was no evidence of a significant relationship between any of the measured root properties and steady-state infiltration; however, the coarser Scots pine roots, with larger diameter, may create larger, more persistent macropores. It is interesting to note that Bartens *et al.* (2008), comparing infiltration rates under young red maple and black oak trees over a period of several months also found consistently higher (although not significant) infiltration rates under the coarse-rooted black oak.

Soil macrofauna presence did not have a significant effect on steady-state infiltration either, therefore the hypothesis that soil macrofauna contribute to higher steady-state infiltration under trees was not supported and neither was the hypothesis that the contribution of soil macrofauna to steady-state infiltration differs between tree species. Nevertheless a slight, but significant, positive correlation between the number of mature earthworms present in the mesocosms and steady-state infiltration suggests that some of the variation in infiltration rates may be attributable to the burrowing activity of earthworms. In fact, the 5.1 times higher steady-state infiltration in the no tree treatment accessible to macrofauna compared with the inaccessible treatment, although not significant, is consistent with increases in infiltration of two to ten times typically attributed to earthworm activity (Lee & Foster, 1991). This increase was not observed between the tree treatments accessible and inaccessible to soil macrofauna however, which may be explained by the significantly fewer mature earthworms found under the trees. Earthworm abundance was found to be positively related to soil pH; however, while there is much evidence that earthworms are sensitive to pH, most species can tolerate wider ranges than were observed within this study (Edwards & Bohlen, 1996). A negative relationship between root volume and earthworm abundance suggests that the presence of tree roots may have a negative effect on earthworms, possibly by altering the habitat. Although soil moisture was not measured, the uptake of water by tree roots has been shown to reduce the soil moisture around individual trees (Ziemer, 1968), which may create an unfavourable environment for earthworms (Edwards & Bohlen, 1996). Tree roots may also obstruct movement of earthworms through the soil.

Despite having no significant effect on steady-state infiltration, soil macrofauna presence was found to be associated with significantly higher soil bulk density accompanied by a change in aggregate structure from granular or crumbly to blocky. This was surprising for two reasons. Firstly, bulk density is closely related to soil porosity, so an increase in bulk density is usually associated with a reduction in steady-state infiltration; however, preferential flow through the macropores created by earthworm burrows can be many times faster than through the smaller pores that make up the rest of the soil matrix (Beven & Germann, 1982), so a loss in total porosity may be offset by an increase in macroporosity. Secondly, it is generally accepted that the burrowing activity of earthworms improves soil structure, reducing bulk density and increasing porosity (Edwards & Bohlen, 1996). While there are studies that support this idea of a reduction in bulk density associated with the presence of earthworms (Edwards & Bohlen, 1996), there are an increasing number of studies that report the opposite effect (Alegre *et al.*, 1996; Hale *et al.*, 2005; Milleret *et al.*, 2009; Kohler-Milleret *et al.*, 2013). Although earthworm burrows can increase soil macroporosity, displacement of soil particles by earthworms during burrowing may compact soil around the burrow. Rogasik *et al.* (2014) found that the burrowing activity of the anecic earthworm species, *L. terrestris*, could increase the bulk density of the soil around the burrow by more than 30% compared with the rest of the soil matrix. Capowiez *et al.* (2011) observed a similar effect for *A. nocturna*, also an anecic species. In addition, they found that the endogeic species, *A. chlorotica*, created zones of higher bulk density in the soil by backfilling burrows with cast material. This habit of backfilling by some species may explain the change in soil structure observed in this study, resulting from increased homogenisation of the soil as it is mixed by the burrowing and casting activity of earthworms. A species-specific influence of earthworms on bulk density was demonstrated by Derouard *et al.* (1997), who suggest that earthworms should be broadly divided into groups of "compacting" and "decompacting" species. Scharenbroch & Johnston (2011) also demonstrated that the effect of earthworms on bulk density depended on initial soil compaction. In their study *L. terrestris* decreased the bulk density of a compacted soil with an initial bulk density of 1.95 g cm^{-3}, but had no effect on a less compacted soil with an initial bulk density of 1.48 g cm^{-3}.

In conclusion, while the duration of this experiment (less than one and a half years) was found to be insufficient to capture the significant changes in soil hydraulic properties normally associated with the presence of tree roots and soil macrofauna, there were early signs of a tree species effect on steady-state infiltration mediated by roots, while a positive relationship between the number of mature earthworms and steady-state infiltration, together with a negative impact of trees on earthworm population size, suggests that the contribution of earthworms to soil hydraulic properties is negatively affected by the presence of trees.

CHAPTER 5

Tree species influence on soil hydraulic conductivity: the contribution of leaf litter and the associated soil decomposer community

5.1 Introduction

Organic matter is an important agent for soil structural formation, improving porosity and promoting aggregate stability (Tisdall & Oades, 1982; Bronick & Lal, 2005), both of which positively influence soil hydraulic properties. This has led to the suggestion that enhanced incorporation of organic matter into the soil from litterfall contributes to higher soil saturated hydraulic conductivity (K_s) under forest compared with other land covers (Pritchett & Fisher, 1987; Young, 1997; Archer *et al.*, 2013). However, while there have been many studies documenting the positive effects of organic matter on soil structural properties (*e.g.* Chaney & Swift, 1984; Six *et al.*, 2000, 2002; Tang *et al.*, 2011), the relationship between organic matter and K_s remains poorly understood. Studies to date, which have primarily focused on the effects of adding compost or organic fertilizers to agricultural soils, have had inconsistent results. While some have shown improvements in infiltration following applications of organic matter, others have shown no effect, or even the reverse, depending on the type of organic material added (Boyle *et al.*, 1989; Martens & Frankenberger, 1992; Whelan *et al.*, 2013). To the author's knowledge, the effects of tree leaf litter on K_s have never been specifically tested.

A key driver of soil organic matter (SOM) dynamics in forest ecosystems is tree species. Inputs related to litterfall have been shown to vary among tree species by several common garden experiments (Binkley, 1994; Reich *et al.*, 2005), while variation in leaf chemistry (Vivanco & Austin, 2008; Vesterdal *et al.*, 2008; Trum *et al.*, 2011) and the associated decomposer community (Neirynck *et al.*, 2000; Scheu *et al.*, 2003; Keith *et al.*, 2009) accounts for much of the variation in decomposition rates (Aerts, 1997; Cornwell *et al.*, 2008). Contrasting effects of broadleaved and coniferous species, with broadleaf litter decaying faster than coniferous needles, results from variation in litter quality factors, particularly the C:N ratio and lignin:N ratio (Zhang *et al.*, 2008). Furthermore, leaf chemistry has an indirect effect on decomposition via its influence on the decomposer community. It has been suggested that competition for nutrients from litter among soil biota creates a selective pressure that favours those that are most efficient at breaking it down (Ayres *et al.*, 2009b). It would therefore be expected that leaf litter decomposes faster under the tree species from which it derives than under a different tree species, an effect that has become known as 'home-field advantage' (HFA). HFA has been reported by a number of studies (Ayres *et al.*, 2009a; Milcu & Manning, 2011), although the effect is not always observed (Ayres *et al.*, 2006; St John *et al.*, 2011).

Based on the contribution of leaf litter and the decomposer community to SOM and the perceived effects of organic matter on K_s it was hypothesised that:

1) K_s will be enhanced by the addition of leaf litter;
2) leaf litter from a coniferous tree species will increase K_s more than leaf litter from a deciduous tree species;
3) higher quantities of leaf litter will increase the effect of that litter on K_s;
4) soil biota will have a negative effect on K_s.

These hypotheses were tested using leaf litter and associated soil biota of Scots pine (a coniferous species) and sycamore (a deciduous species) in two simultaneous experiments established using soil microcosms under glasshouse conditions. One was set up to test the influence of litter type and soil biota in a full factorial design, while additional treatments were used to test the effect of increasing litter quantity.

5.2 Methods

Experimental set-up

Microcosms were constructed using sections of standard PVC waste pipe with an inner diameter of 103 mm and height approximately 250 mm. A 1 mm aperture nylon mesh was attached to the base to keep the soil in place while avoiding any restriction to flow during hydraulic conductivity testing. Soil was collected from Hazelrigg environmental research station (54°1′ N, 2°46′ W) at Lancaster University, air-dried and sieved using a 9 mm mechanical sieve to break down large aggregates and to remove stones and large organic residues, then steam-sterilised at 121° C using an autoclave. Tests showed that this soil had a sandy loam texture (63% sand, 21% silt, 16% clay), a mean pH of 6.0 and mean organic matter content of 5.42%. After adjusting for moisture content (determined by oven-drying samples at 105° C for 48 hours), the soil was weighed and packed by hand in 50 mm layers to a total depth of 200 mm at an equivalent dry bulk density of 1.2 g cm^{-3}.

Fresh leaf litter and soil were collected from the top 10 cm of the soil profile beneath sycamore (representing a deciduous broadleaf species) and Scots pine (representing an evergreen coniferous species) woodland plots at an experimental site at Glensaugh in Scotland (56°54′ N, 2°33′ E) during October 2012. The leaf litter was roughly chopped and steam-sterilised at 121° C by placing it in an autoclave. It was then oven-dried at 60° C until its weight reached a constant value. The soil was used to prepare suspensions of soil biota associated with the respective tree species. A ratio of 1 kg of soil to 1 litre of water was mixed, left for an hour to allow the large particles to settle and then the suspension filtered through a 1 mm sieve (adapted from Ayres *et al.*, 2004; Bezemer *et al.*, 2005).

For experiment 1 nine treatments, each with six replicates (*i.e.* a total of 54 microcosms), were created to test the effect of litter type and soil biota on soil hydraulic conductivity as follows:

1) sycamore litter with sycamore soil biota;
2) sycamore litter with Scots pine soil biota;
3) sycamore litter with no soil organisms;
4) Scots pine litter with sycamore soil biota;
5) Scots pine litter with Scots pine soil biota;
6) Scots pine litter with no soil organisms;
7) no litter with sycamore soil biota;
8) no litter with Scots pine soil biota;
9) a control with no litter and no soil organisms.

For experiment 2, the sycamore litter with sycamore soil biota and Scots pine litter with Scots pine soil biota treatments were used to represent standard litter treatments, while treatments with the same combination of litter type and soil biota, but double the quantity of litter, were prepared to represent high litter treatments. This gave a total of four treatments, also replicated six times (*i.e.* 24 microcosms), to test the importance of litter type versus litter quantity:

1) sycamore standard litter treatment;
2) Scots pine standard litter treatment;
3) sycamore high litter treatment;
4) Scots pine high litter treatment.

9 g dry weight of litter was added to standard litter treatments and 18 g to high litter treatments. This equates to 2 times and 4 times an estimated annual litterfall of 540 g m^{-2}, which falls within a range of published values for sycamore and Scots pine (Binkley, 1994; Davis & Trettin, 2006; McKay, 2011). Macrofauna were not included in the soil biota, so half of the litter was mixed into the top layer of sterilised soil before it was packed into the microcosm to simulate mixing by earthworms, while the remainder was spread over the soil surface. Microcosms were then inoculated with soil biota by applying 100 ml of the appropriate suspension and placed on trays with capillary matting beneath them. In order to avoid cross-contamination between treatments inoculated with different soil biota, it was necessary to group microcosms containing the same biota together. Since grouping treatments can introduce a confounding factor of location, all treatments were divided into two equal sets of replicates and placed in two separate blocks within the glasshouse, with the groups in the second block placed in a different order to those in the first.

Maintenance of the experiment

Automated lighting and heating in the glasshouse maintained daylight between 06:00 and 22:00, with a daytime temperature of 18° C and a night time temperature of 14° C. A manual sprinkling system was used to simulate rainfall, with 21 mm of water applied each week. This approximately equates to the annual rainfall of 1100 mm recorded at the Glensaugh experimental site, where litter and soil for the inoculum were collected. For the first five weeks this was delivered in a single weekly application; however, after observing rapid drying of the soil surface by evaporation, this was subsequently divided into twice weekly applications. Capillary matting beneath the microcosms was kept wet to simulate water supply from ground water.

Sampling

At the end of 30 weeks sampling was undertaken to determine K_s, bulk density (ρ_b), SOM and the amount of leaf litter lost from the soil surface. Soil columns were initially placed in a bowl of water and allowed to saturate from below. They were then removed and a fixed-head of water applied to the soil surface using a Mariotte bottle (figure 5.1). Infiltration rate was measured until a steady state was reached and K_s calculated from Darcy's Law. Columns were subsequently allowed to drain and surface litter carefully removed and weighed. ρ_b was determined gravimetrically from a small, undisturbed core (diameter 48.8 mm, depth 50.9 mm) extracted from the soil surface and oven-dried at 105° C for 48 hours. A sub-sample of this soil was then used to determine SOM by loss on ignition (560° C overnight).

Figure 5.1. Photograph showing how infiltration through the soil columns was measured using a Mariotte bottle to maintain a fixed-head of water at the soil surface.

Statistical analysis

Statistical analysis was carried out using the software package R version 3.0.0 (R Core Team, 2013). The Shapiro-Wilks test was used to check for normality of the data and a square root transform applied to the K_s data in order to achieve a Gaussian distribution before testing for treatment differences. The effects of litter type, litter quantity and soil biota were analysed using linear mixed effects models with a random effect of block, followed by the *post hoc* Tukey test. Model simplification was performed where interactions terms were not found to be significant (Crawley, 2013). Relationships between variables were investigated by regression analysis using linear models.

5.3 Results

Experiment 1: The effect of litter and soil biota

The effect of adding litter on K_s was found to depend on tree species. K_s (figure 5.2a) was reduced by sycamore litter ($p < 0.001$) but not Scots pine litter and the two were significantly different from each other ($p < 0.001$). K_s was also reduced by the soil biota ($p < 0.01$), although there was no difference in the effect of soil biota associated with different tree species ($p > 0.05$). ρ_b (figure 5.2b) was found to be significantly reduced by the addition of either litter type ($p < 0.001$), while soil biota had no effect ($p > 0.05$) and there was no difference between tree species ($p > 0.05$). Furthermore, there was no interactive effect of litter type and soil biota on either K_s or ρ_b. In contrast, changes in SOM (figure 5.2c) were found to be driven by complex interactions between leaf litter and soil biota. After thirty weeks, with sterile soil and no litter added, SOM in the control treatment remained unchanged, acting as a benchmark to compare the other treatments. The addition of litter to sterile soil increased SOM, although this was only significant for Scots pine ($p < 0.01$), whereas there was no significant difference in SOM when soil biota were added without leaf litter ($p > 0.05$). The addition of either sycamore or Scots pine litter to soil inoculated with soil biota associated with sycamore significantly increased SOM ($p < 0.01$); however, when litter was added to soil inoculated with soil biota associated with Scots pine, SOM was similar the control treatment ($p > 0.05$). Neither litter or soil biota

CON+CON	no litter and no soil organisms	SP+CON	Scots pine litter with no soil organisms
SYC+CON	sycamore litter with no soil organisms	CON+SP	no litter with Scots pine soil biota
SP+SP	Scots pine litter with Scots pine soil biota	SYC+SP	sycamore litter with Scots pine soil biota
CON+SYC	no litter with sycamore soil biota	SP+SYC	Scots pine litter with sycamore soil biota
SYC+SYC	sycamore litter with sycamore soil biota		

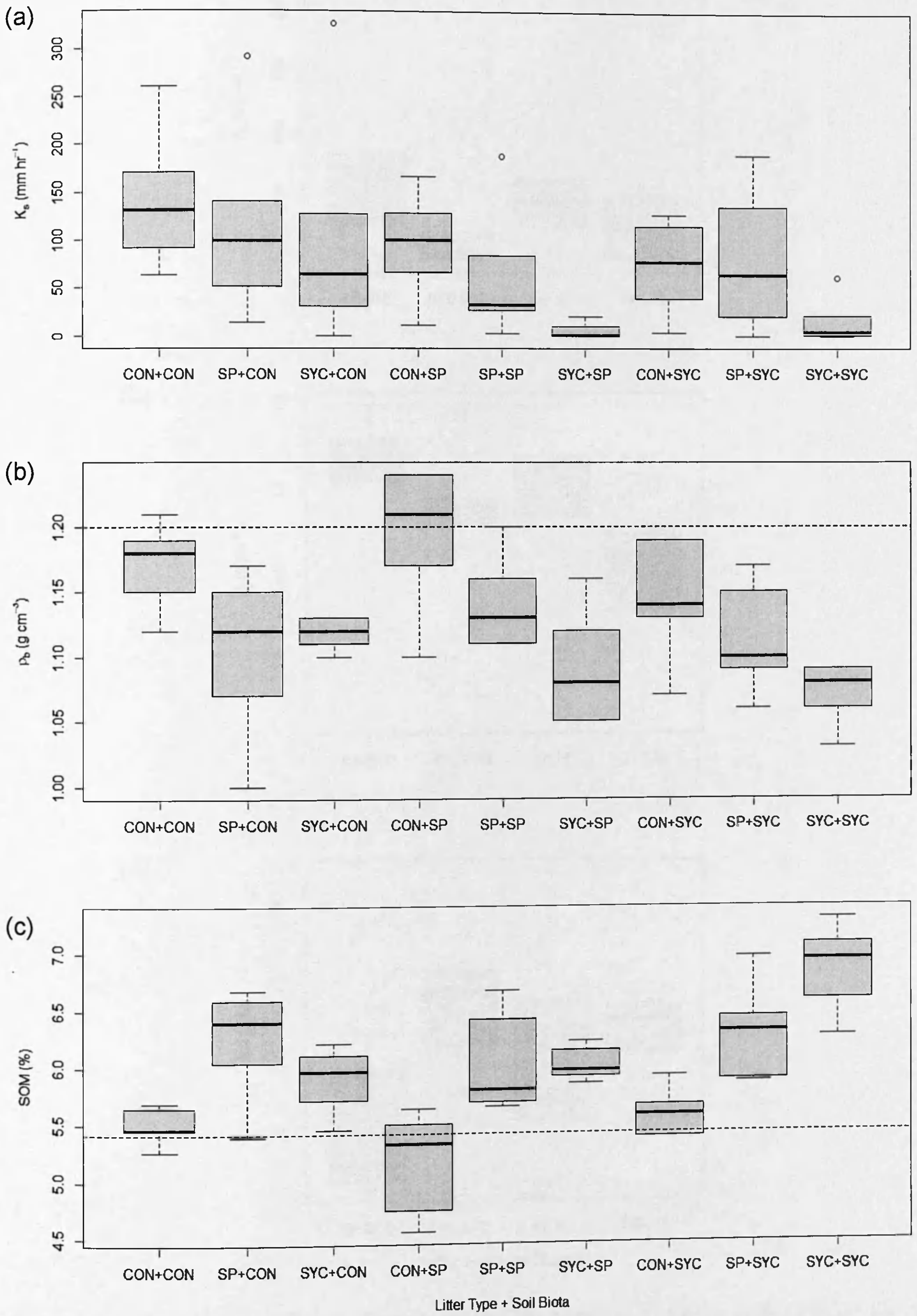


Figure 5.2. Boxplots showing median values and range of (a) K_s, (b) ρ_b and (c) SOM for each of the experimental treatments. Superimposed dashed lines show (b) the original packing density of the microcosms and (c) the mean value of SOM at the start of the experiment.

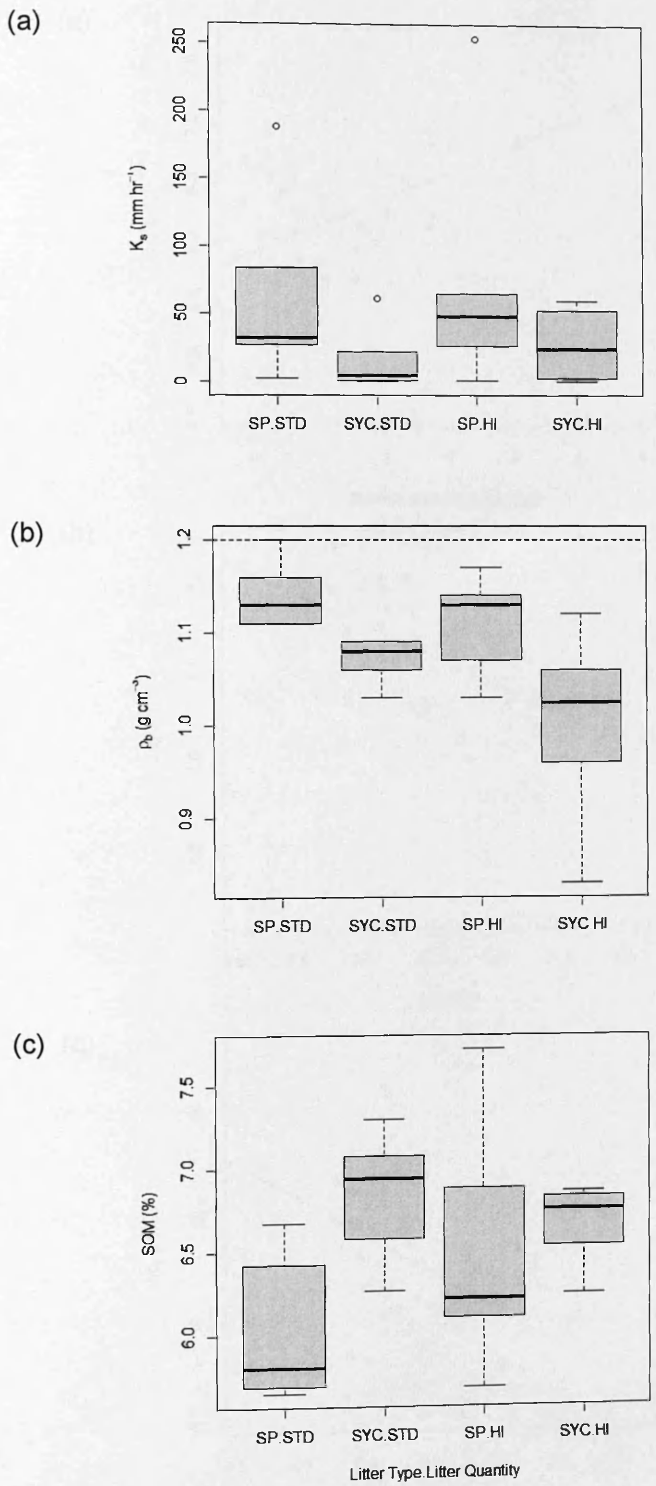


Figure 5.3. Boxplots showing median values and range of (a) K_s, (b) ρ_b and (c) SOM for each of the Scots pine standard litter (SP.STD), sycamore standard litter (SYC.STD), Scots pine high litter (SP.HI) and sycamore high (SYC.HI) litter treatments in experiment 2. The superimposed dashed line shows the original packing density of the microcosms.

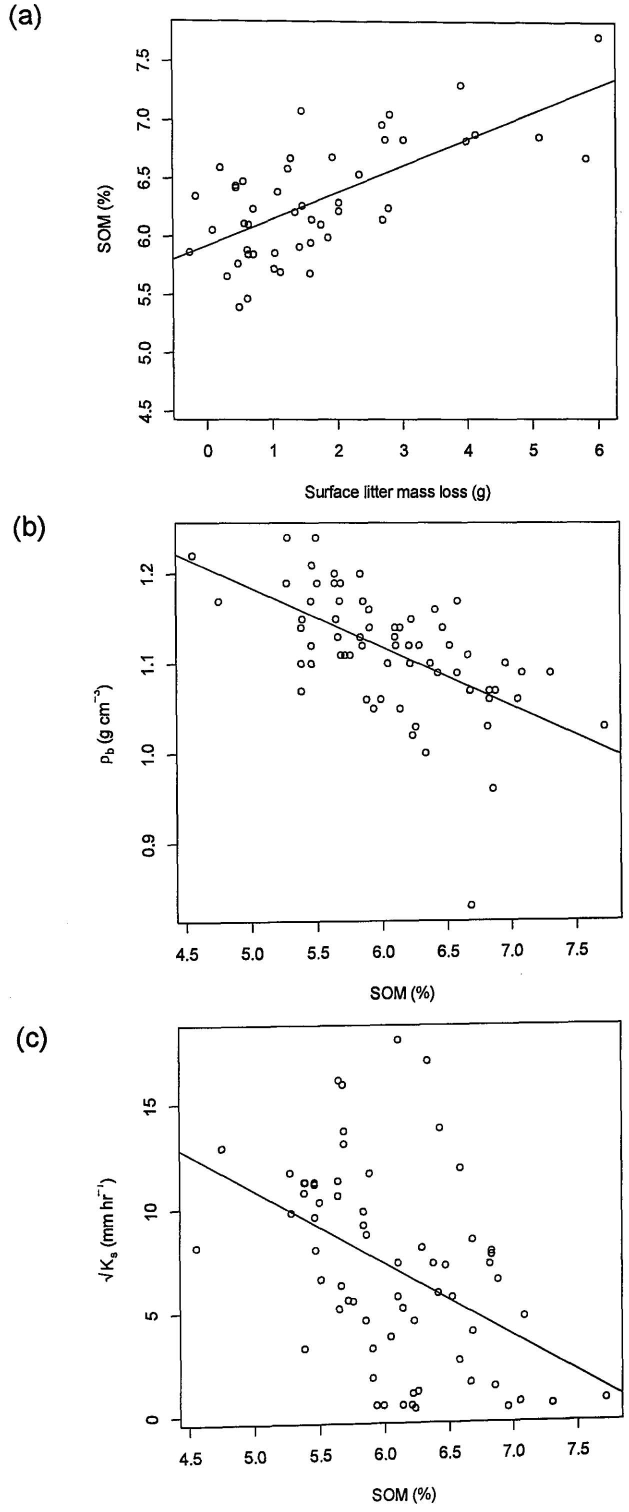


Figure 5.4. Relationships between (a) SOM and mass loss from surface litter; (b) ρ_b and SOM; (c) $\sqrt{K_s}$ and SOM.

alone demonstrated a tree species difference ($p > 0.05$); however, when sycamore litter was added to soil inoculated with sycamore soil biota, SOM was significantly greater ($p < 0.05$) than in any other treatment.

Experiment 2: Litter type vs. litter quantity

Litter type, together with the associated soil biota, was found to have a greater effect than the quantity of litter added (figure 5.3). Doubling the quantity of litter added had no significant effect on SOM or K_s, and ρ_b was only marginally lower ($p = 0.06$), whereas the sycamore treatments significantly increased SOM ($p < 0.01$), decreased ρ_b ($p < 0.001$) and decreased K_s ($p = 0.017$) compared to Scots pine.

Relationships between soil properties

Significant relationships were found between all of the measured soil properties, as well as between the reduction in mass of surface litter and SOM. While difficulty in separating surface litter from soil particles resulted in some of the variation in the mass loss from litter added to the soil surface, mass loss was found to be significantly related to increases in SOM ($r^2 = 0.46$; $p < 0.001$; figure 5.4a). ρ_b was negatively related to SOM ($r^2 = 0.34$; $p < 0.001$; figure 5.4b), as was K_s, although this relationship was non-linear, with the K_s data following a square root distribution while SOM was normally distributed. After applying a square root transform to the K_s data, a highly significant negative relationship between the transformed K_s data and SOM was observed ($r^2 = 0.21$; $p < 0.001$; figure 5.4c). In addition, the transformed K_s data were positively correlated with ρ_b ($r^2 = 0.14$; $p = 0.002$).

5.4 Discussion and conclusions

Litterfall from trees is a major contributor of organic matter to the soil and is thought to contribute to the higher K_s generally observed under forest owing to a positive effect of SOM on soil porosity. Thirty weeks after litter was added to the soil in this study, there were significant differences in SOM, ρ_b and K_s between treatments; however, while significant relationships between SOM, ρ_b and K_s were observed, the results suggest that the influence of litter on K_s is comprised of a number of competing effects.

The effect of litter on K_s

Contrary to expectations, K_s did not increase when leaf litter was added to the soil, with Scots pine litter having no effect and sycamore litter significantly reducing K_s. The hypothesis that K_s will be enhanced by the addition of leaf litter is therefore not supported. Neither is the hypothesis that leaf litter from a coniferous species will increase K_s more than a deciduous species. However, these results do confirm a species effect of leaf litter on K_s, which may contribute to variation in K_s for soils supporting different tree species. In addition, it was hypothesised that higher quantities of leaf litter will increase the effect of that litter on K_s; yet the high litter treatments, with double the amount of litter added, had no significant effect on K_s compared to the standard treatments. The difference in the amount of litter added between standard and high litter treatments was higher than that generally reported for differences between tree species of the same age and similar site conditions (Binkley, 1994), therefore these results suggest that variation in litterfall quantity is unlikely to contribute to variation in K_s for different tree species. Whelan *et al.* (2013), studying the effects of compost addition to a brownfield site, also observed that higher applications of organic matter had little additional effect on K_s, although significant differences were observed for different types of organic material.

The effect of soil biota on K_s

Soil biota were also found to reduce K_s, with a similar effect observed for biota associated with both tree species, supporting the hypothesis that soil biota have a negative effect on K_s. These results contradict those of an earlier study by Allton *et al.* (2007), in which microbial communities had no effect on mean volume of runoff or percolate, suggesting no effect on K_s. This may be because their study was of much shorter duration (two weeks compared with thirty weeks), allowing less time for microbial activity to affect soil properties.

The relationship between SOM, ρ_b and K_s

In common with previous studies (Adams, 1973; Agnese *et al.*, 2011), SOM was found to be negatively correlated with ρ_b, which suggests a positive relationship with porosity; however, although higher porosity usually leads to increased K_s, a weak, but highly significant, negative correlation was observed between SOM and K_s. While several studies support the hypothesis that K_s and SOM are positively related (*e.g.* Lado *et al.*, 2004; Benjamin *et al.*, 2008; Agnese *et al.*, 2011), Nemes *et al.* (2005) reported that a negative relationship between K_s and SOM was predicted for a wide range of soils by several pedotransfer function models based on regression analysis of existing soil databases. A subsequent field study by Wang *et al.* (2009) confirmed this negative relationship for a sandy soil in Nebraska. They suggest that a positive effect of SOM on K_s, by increasing soil aggregation, is counteracted by increased water repellency caused by some organic compounds; although, since water repellency has been shown to break down under saturated conditions (Clothier *et al.*, 2000; Doerr *et al.*, 2000), pre-saturation of the microcosms was expected to eliminate this effect. Nemes *et al.* (2005) propose that enhanced aggregation as a result of increased SOM alters pore characteristics, reducing pore diameters and increasing tortuosity, therefore reducing K_s. A third possibility is that increased microbial activity, stimulated by SOM, leads to bioclogging. Morales *et al.* (2010) suggested that microbial activity may significantly reduce K_s as a result of pore clogging by the extracellular substances that microbes excrete to protect themselves from abrupt hydrating or dehydrating conditions. The fact that soil biota reduced K_s while having no effect on ρ_b in this study is consistent with a reduction in the number of effective flow pathways, without a significant reduction in overall porosity. In contrast, leaf litter reduced ρ_b, suggesting an increase in porosity, but had either no effect or a negative effect on K_s, supporting the idea of competing positive and negative effects of SOM on K_s.

Changes to SOM were driven by interactions between leaf litter and soil biota. Unsurprisingly, without the presence of a soil decomposer community, adding litter to the soil increased SOM. The fact that this was only significant in the case of Scots pine may indicate that the sycamore litter, which is less coarse than the Scots pine needles, had undergone some mechanical breakdown, with subsequent leaching of smaller particles during watering. Without fresh litter, soil biota had no effect on SOM already in the soil, implying that this was the chemically recalcitrant fraction remaining in the late stages of decomposition. Once 70-80% of the original mass is lost, decomposition rates become very slow (Prescott, 2010). Adding both litter and soil biota increased SOM, but this was only significant for soil inoculated with biota associated with sycamore. A significant relationship between mass lost from the surface litter layer and SOM revealed that higher SOM was, in part, related to increased incorporation of surface litter, indicating faster decomposition by the sycamore decomposer community. In addition, sycamore litter increased SOM significantly more than Scots pine litter in the soil inoculated with biota associated with sycamore. There was therefore evidence of HFA for sycamore but not Scots pine, although this did not translate into an effect on K_s. According to Keiser *et al.* (2013), this effect, whereby litter added to a soil derived from a species with more chemically recalcitrant litter decomposes at a similar rate to the litter associated with it, but litter added to a soil associated with a species with more chemically labile litter decomposes at a slower rate, is indicative of greater functional diversity in the decomposer community associated with the more chemically recalcitrant species. Overall faster decomposition associated with sycamore, with a lower C:N and lignin:N ratio (Hobbie *et al.*, 2006), compared to Scots pine is consistent with the pattern of faster decomposition of broadleaved litter compared with coniferous litter controlled by litter quality determined by Zhang *et al.* (2008). This control on decomposition rate determined by litter quality may also explain why doubling the quantity of litter had little effect on SOM and, therefore, K_s.

These experiments were conducted over a relatively short time period. Over longer time-scales, with repeated annual inputs of leaf litter and faster decomposition of sycamore litter relative to Scots pine litter, the influence of litter type may alter. However, contrary to expectation, these experiments have demonstrated a negative influence of tree leaf litter on K_s and, in addition, shown that litter type has a greater effect than litter quantity on K_s.

CHAPTER 6

Summary and Conclusions

The establishment of new forested areas, including riparian tree buffer zones, is increasingly being incorporated into policies for the mitigation of the environmental impacts of diffuse pollution and flooding because the presence of trees is generally associated with enhanced soil hydraulic properties that promote infiltration and reduce surface runoff (Huang *et al.*, 2003; Zimmermann *et al.*, 2006; Chandler & Chappell, 2008; Hassler *et al.*, 2011). However, our understanding of how trees affect soil hydraulic properties is still very limited. The work presented in this thesis attempts to address some of the gaps in our knowledge by investigating how tree species and land use affect the hydraulic properties of forest soils and the key mechanisms involved.

6.1 Key findings

Land use effects

It is generally accepted that soil hydraulic properties are enhanced by the presence of trees; however land use is also known to be a strong driver of soil hydraulic properties. Although studies have shown significantly higher soil hydraulic conductivity under forest compared with other types of vegetation, the intensity of land use in these forests is often much lower than of the land being compared; for example, when relatively undisturbed forest is compared with grassland grazed by livestock (*e.g.* Agnese et al., 2011). While it is important to understand the consequences of land use conversion on soil hydraulic properties, it can be difficult to separate the influence of land use from vegetation effects in these studies. In chapter 2 of this thesis the effect of land use was investigated by comparing hydraulic conductivity at the soil surface in forests grazed by livestock (silvopasture) and ungrazed forests with the hydraulic conductivity of the soil in grazed pasture. Land use was found to have a profound effect on the hydraulic conductivity of forest soils. While saturated hydraulic conductivity in the ungrazed forests was significantly higher than in the pastures, there was no apparent tree effect in the grazed forests. It may be speculated that the higher soil saturated hydraulic conductivity in the relatively undisturbed forest simply reflects the recovery of soil properties in the absence of grazing that has been observed in other studies (*e.g.* Castellano & Valone, 2007); however, a tree species difference in undisturbed forest also observed in this study suggests that the negative effect of the higher intensity land use on forest soils masks the effect of the trees. This has important implications for tree planting schemes aimed at mitigating diffuse pollution and flooding, as the subsequent land use of these newly forested areas will determine the extent of their effectiveness in providing the ecosystem services of water regulation and purification. Such schemes may aim to provide multiple ecosystem services, including recreation; however, it is clear that this could involve a trade-off with the other ecosystem services provided by trees.

Tree species effects

While a high intensity land use, such as livestock grazing, was found to have a greater influence than the trees on the hydraulic properties of forest soils, the work presented in chapter 2 found clear evidence of a tree species effect when the forest remained relatively undisturbed. However, although saturated hydraulic conductivity at the soil surface was an order of magnitude higher under stands of mature Scots pine than under stands of mature sycamore, comparison of soil hydraulic conductivity with estimated rainfall intensities suggested that infiltration excess overland flow in undisturbed forest planted with either species was extremely rare at the site studied. Therefore, despite this large difference in hydraulic properties between species, the effectiveness of tree planting schemes to mitigate diffuse pollution and flooding may depend more on the subsequent land use than on the actual tree species planted, although this does need to be tested for other soil types and tree species.

Mechanisms

Having found an influence of both tree species and land use on soil hydraulic conductivity in the work described in chapter 2, the underlying causes of differences were explored at the same field site in a study of soil flow pathways and soil pore characteristics, described in chapter 3. In this study higher saturated hydraulic conductivity was found to be associated with higher soil macroporosity and greater macropore flow, which is consistent with reports of a significant positive relationship between these properties observed in other studies (Mbagwu, 1995; Luo et al., 2010; Agnese et al., 2011). Differences in meso- and macroporosity (pores > 74 μm diameter) between forest and pasture could be attributed to generally larger diameter pores in the ungrazed forest soils, while differences between tree species could be attributed to a greater density of large pores in the Scots pine forest soils. The key mechanisms influencing soil hydraulic properties are therefore those that affect soil macropore characteristics and these were investigated further within this chapter.

Potential sources of soil macropores include roots, burrowing soil macrofauna and organic matter, which promotes soil aggregation (Tisdall & Oades, 1982). Macropores can also be destroyed by compaction at the soil surface. Within the ungrazed forests root diameter and soil organic matter was found to be higher in the Scots pine forests than in the sycamore forests, while root length density and soil macrofauna populations were lower. This suggests that differences in soil hydraulic properties between tree species in relatively undisturbed forests are most likely to be related to differences in root diameter and soil organic matter. Differences between pasture and forest soils may also be attributed, in part, to differences in root diameter. In fact, measurements of root characteristics in pasture soils suggested that the grass roots were of insufficient diameter to impact on soil macroporosity. However, compaction can also negatively affect soil porosity and visual observation of dye stained flow pathways and computer tomography (CT) images of soil pores suggested that the interaggregate voids, created by soil aggregation, were destroyed by the compaction associated with livestock trampling. It is interesting to note that the size of soil macrofauna populations followed a reverse trend to the soil hydraulic conductivity in this study. Although burrowing soil fauna, such as earthworms, have been shown to have a positive effect on soil structure and soil hydraulic properties (Edwards & Bohlen, 1996), studies have generally compared these properties for different population sizes where land use and vegetation type is the same. From these results it may be inferred that, compared to land use and vegetation influences, burrowing soil fauna contribute very little to soil hydraulic conductivity at the landscape scale; however, from visual observation of dye stained profiles and CT images, earthworm burrows were found to be an important source of macropores in pasture. This suggests that, under intense land use, where interaggregate voids are destroyed by compaction, planting vegetation that promotes larger earthworm populations could improve infiltration by increasing soil macroporosity and therefore reduce the likelihood of surface runoff.

In order to isolate the potential mechanisms responsible for differences in soil hydraulic properties under different tree species, controlled experiments were carried out alongside the field studies. A mesocosm experiment, described in chapter 4, was set up to investigate the separate and combined influence of tree roots and associated soil macrofauna using the same tree species studied at the field-scale. Although trees had a negative impact on the population of mature earthworms, and a significant positive relationship between the number of mature earthworms and steady-state infiltration was observed, differences in steady-state infiltration between mesocosms with and without soil macrofauna excluded was not significant, again suggesting that earthworms contribute very little to overall variation in soil hydraulic properties when the soil is not compacted. The impact of roots was also found to be insignificant; however, the difference between the Scots pine and sycamore trees followed the same trend as the more mature trees studied, suggesting early signs of a root-mediated species effect. Scots pine trees had a significantly greater mean root diameter than sycamore, in line with the more mature trees studied; however, it has been suggested that decayed roots are more important for creating macropore flow than living roots (Perillo *et al.*, 1999) and this short-term experiment may not have allowed sufficient time for root death and decay to occur. Withington *et al.* (2006) found that the root turnover rate of Scots pine was three times faster than the sycamore trees studied. This suggests that root turnover rate, in addition to root diameter, may also contribute to the differences in soil hydraulic conductivity observed between the two species in more mature trees and it may take a number of years for this difference to become significant.

A microcosm experiment, described in chapter 5, was also set up to test the effects of leaf litter on saturated hydraulic conductivity. The quantity of litter added did not significantly affect saturated hydraulic conductivity in this experiment, but a species effect was observed. Significantly higher saturated hydraulic conductivity was associated with Scots pine litter compared with sycamore litter, suggesting that differences in litter type contributes to the species differences in soil hydraulic properties observed for the mature trees studied. However, when compared with the control, contrary to expectation, sycamore leaf litter was found to have a negative effect, despite a significant decrease in bulk density that might be expected to result in

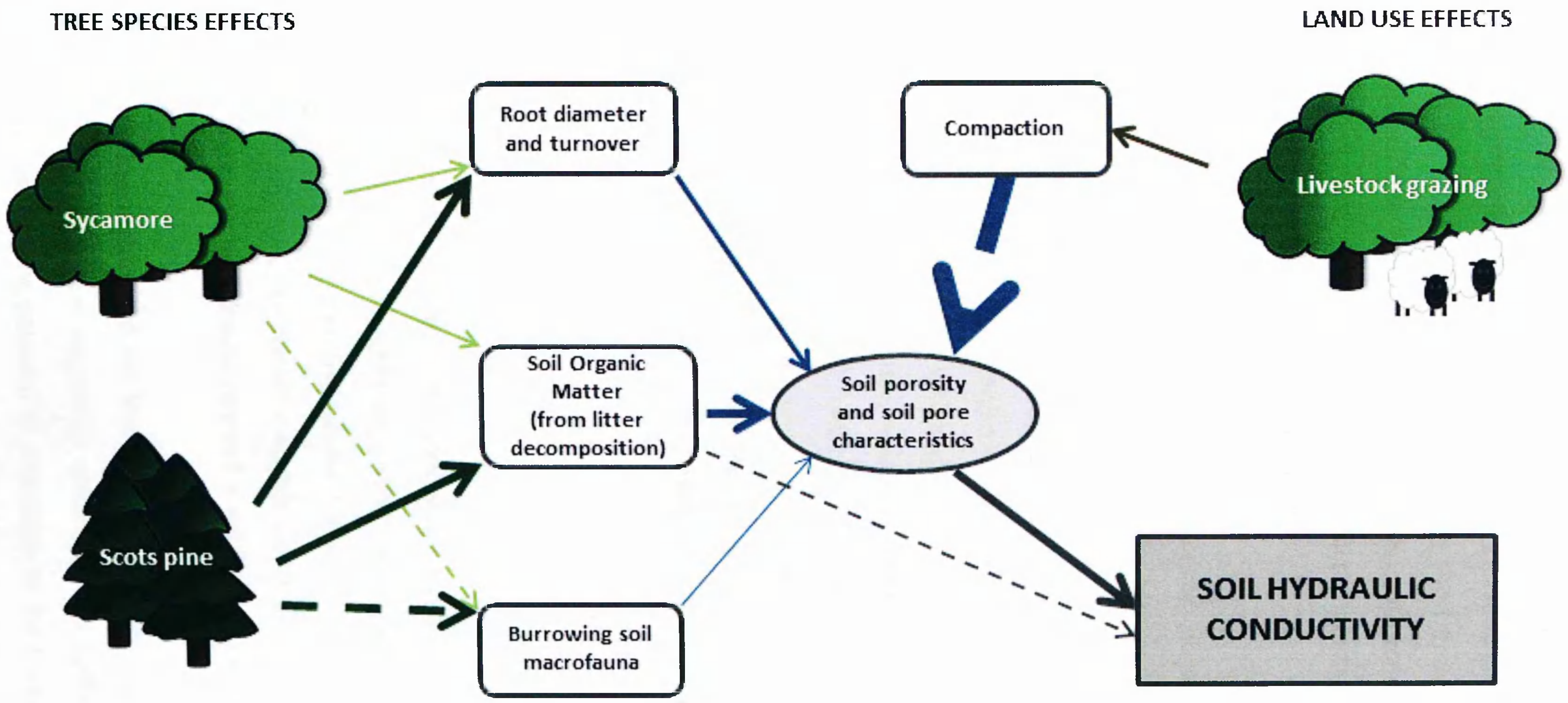


Figure 6.1. Conceptual model showing how tree species and land use influence the hydraulic conductivity of forest soils based on the findings presented in this thesis. The thickness each arrow indicates the relative importance of the effect in relation to other effects, while solid/dashed lines indicate positive/negative effects.

increased hydraulic conductivity. Possible explanations are that increased water repellency, caused by some organic compounds, counteracts the positive effect resulting from increased soil porosity (Wang *et al.*, 2009) or that pores have become clogged by extracellular substances excreted by microbes (Morales *et al.*, 2010). This experiment was also conducted over a relatively short time period and with a single application of litter. Over longer time-scales, and with repeated annual litter inputs, this effect may alter, affected by differences in decomposition rates and the timing of litter fall and this needs to be investigated further.

In conclusion, these findings (and the referenced literature), summarised in a conceptual model (figure 6.1), suggest that the key mechanisms responsible for differences in soil hydraulic properties between tree species are root diameter, root turnover rate and organic matter, acting through changes to soil porosity and pore characteristics. In addition, some organic compounds have a direct negative influence on soil hydraulic conductivity which may also contribute to species differences. Populations of burrowing soil fauna, which are negatively influenced by trees, do not contribute to differences in soil hydraulic conductivity between tree species and appear to contribute very little to soil hydraulic properties overall in relatively undisturbed forests. The effect of tree species is modified by the effect of land use, which can influence soil structure, and consequently soil hydraulic properties, through compaction.

6.2 Future research

The work presented in this thesis has gone some way to address the knowledge gaps identified in the introduction; however, given the ever increasing interest in expanding forest cover to mitigate the environmental impacts of diffuse pollution and flooding, there is a pressing need for further research to clarify the impacts of land use and vegetation cover on soil hydraulic properties and the generation of surface runoff.

This research has highlighted the importance of considering concurrent land use in determining the effects of vegetation cover on soil hydraulic properties. While undisturbed forest has the potential to contribute to the ecosystem services of water regulation and water purification, forests also contribute to cultural ecosystem services, including recreation and ecotourism. Forests are commonly used for recreational purposes, but the impact of these activities, such as walking and mountain biking, may reduce, or even eliminate, the benefits of enhanced infiltration and reduced runoff associated with forest cover. Millward *et al.* (2011), for example, observed an 87% reduction in mean infiltration rates in areas of forested parkland in Toronto used for recreational purposes compared with fenced off areas. It is therefore imperative that we gain a better understanding of the effects of land use combined with forest cover in order to better inform the land management decisions being undertaken to mitigate the impacts of diffuse pollution and flooding.

The recovery of soil properties following a change in land use, without a change in vegetation, also requires further investigation. A number of studies have shown that the soil hydraulic properties of grazed grassland recover following the cessation of grazing (Wheeler *et al.*, 2002; Greenwood *et al.*, 2007; Zimmermann *et al.*, 2010; Hassler *et al.*, 2011), although it may take from a few months to a few years for this to become significant. Improved infiltration may therefore be achieved simply by removing the pressure of grazing, allowing the recovery of the soil under grassland. Over time ungrazed grassland communities tend to undergo natural succession to woodland and it may be more cost effective to allow this natural succession to occur, rather than actively planting trees.

Tree species effects on soil surface hydraulic properties were investigated for this thesis by comparing the effects of sycamore and Scots pine. However, while a clear species effect was observed in this study, some authors have not found an effect when comparing other tree species (*e.g.* Eldridge & Freudenberger, 2005; Sanou *et al.*, 2010; Jost *et al.*, 2012). A broadleaf and a conifer species were chosen for this study because a contrast in root characteristics, associated soil fauna and soil organic matter, previously reported for conifers and broadleaf species (Ovington, 1953; Reich *et al.*, 2005; Withington *et al.*, 2006; Trum *et al.*, 2011), suggested that differences in soil hydraulic properties were most likely to vary between species type. While the underlying mechanisms (*e.g.* root characteristics) investigated as part of this research may provide an indication of how soil hydraulic properties may vary between species, the question of whether species or species type is more important still needs to be answered.

This study revealed a complex collection of interrelated and competing positive and negative mechanisms responsible for changes in soil saturated hydraulic conductivity, that are known to act over different timescales. Although the work presented in this thesis began the process of identifying the relative importance of different mechanisms, there is a need for more work to separate the three key mechanisms related to vegetation, *i.e.* roots, soil fauna and organic matter, from each other and from the effect of compaction related to land use. Furthermore, this work did not consider the other factors that influence soil structural properties, namely climate, topography, soil parent material and time, which can have a direct impact or impact indirectly via their influence on vegetation.

Finally, the work presented in this thesis focused on the effects of tree species and concurrent land use on hydraulic properties at the soil surface. While surface soil hydraulic properties principally affect surface runoff by infiltration excess overland flow, hydraulic properties deeper in the soil will influence surface runoff by saturation excess overland flow. The key mechanisms responsible for affecting hydraulic properties at the soil surface also differ from those deeper in the soil. There is therefore a need for further research of how trees and concurrent land use influence soil hydraulic properties over greater depth.

REFERENCES

Abdel-Magid, A., Trlica, M., & Hart, R. 1987. Soil and Vegetation Responses to Simulated Trampling. *Journal of Range Management* **40**, 303–306.

Adams, W.A. 1973. The Effect of Organic Matter on the Bulk and True Densities of Some Uncultivated Podzolic Soils. *Journal of Soil Science* **24**, 10–17.

Aerts, R. 1997. Climate, Leaf Litter Chemistry and Leaf Litter Decomposition in Terrestrial Ecosystems: A Triangular Relationship. *Oikos* **79**, 439–449.

Agnese, C., Bagarello, V., Baiamonte, G., & Iovino, M. 2011. Comparing Physical Quality of Forest and Pasture Soils in a Sicilian Watershed. *Soil Science Society of America Journal* **75**, 1958–1970.

Alaoui, A., Caduff, U., Gerke, H.H., & Weingartner, R. 2011. Preferential Flow Effects on Infiltration and Runoff in Grassland and Forest Soils. *Vadose Zone Journal* **10**, 367–377.

Alaoui, A., & Goetz, B. 2008. Dye tracer and infiltration experiments to investigate macropore flow. *Geoderma* **144**, 279–286.

Alcamo, J., Bennett, E.M., & Millennium Ecosystem Assessment (Program). 2003. Ecosystems and human well-being: a framework for assessment. Island Press, Washington, DC.

Alegre, J.C., & Cassel, D.K. 1996. Dynamics of soil physical properties under alternative systems to slash-and-burn. *Agriculture, Ecosystems & Environment* **58**, 39–48.

Alegre, J.C., Pashanasi, B., & Lavelle, P. 1996. Dynamics of Soil Physical Properties in Amazonian Agroecosystems Inoculated with Earthworms. *Soil Science Society of America Journal* **60**, 1522.

Allaire-Leung, S.E., Gupta, S.C., & Moncrief, J.F. 2000a. Water and solute movement in soil as influenced by macropore characteristics - 1. Macropore continuity. *Journal of Contaminant Hydrology* **41**, 283–301.

Allaire-Leung, S.E., Gupta, S.C., & Moncrief, J.F. 2000b. Water and solute movement in soil as influenced by macropore characteristics - 2. Macropore tortuosity. *Journal of Contaminant Hydrology* **41**, 303–315.

Allton, K.E., Harris, J.A., Rickson, R.J., & Ritz, K. 2007. The effect of microbial communities on soil hydrological processes: A microcosm study utilising simulated rainfall. *Geoderma* **142**, 11–17.

Al-Maliki, S., & Scullion, J. 2013. Interactions between earthworms and residues of differing quality affecting aggregate stability and microbial dynamics. *Applied Soil Ecology* **64**, 56–62.

Archer, N.A.L., Bonell, M., Coles, N., MacDonald, A.M., Auton, C.A., & Stevenson, R. 2013. Soil characteristics and landcover relationships on soil hydraulic conductivity at a hillslope scale: A view towards local flood management. *Journal of Hydrology* **497**, 208–222.

Arvidsson, J. 1998. Influence of soil texture and organic matter content on bulk density, air content, compression index and crop yield in field and laboratory compression experiments. *Soil and Tillage Research* **49**, 159–170.

Aubertin, G.M. 1971. Nature and extent of macropores in forest soils and their influence on subsurface water movement. U.S. Department of Agriculture, Forest Service, Northeastern Forest Experiment Station.

Ayres, E., Dromph, K.M., & Bardgett, R.D. 2006. Do plant species encourage soil biota that specialise in the rapid decomposition of their litter? *Soil Biology and Biochemistry* **38**, 183–186.

Ayres, E., Heath, J., Possell, M., Black, H.I.J., Kerstiens, G., & Bardgett, R.D. 2004. Tree physiological responses to above-ground herbivory directly modify below-ground processes of soil carbon and nitrogen cycling. *Ecology Letters* 7, 469–479.

Ayres, E., Steltzer, H., Berg, S., & Wall, D.H. 2009a. Soil biota accelerate decomposition in high-elevation forests by specializing in the breakdown of litter produced by the plant species above them. *Journal of Ecology* **97**, 901–912.

Ayres, E., Steltzer, H., Simmons, B.L., Simpson, R.T., Steinweg, J.M., Wallenstein, M.D., Mellor, N., Parton, W.J., Moore, J.C., & Wall, D.H. 2009b. Home-field advantage accelerates leaf litter decomposition in forests. *Soil Biology and Biochemistry* **41**, 606–610.

Bardgett, R.D. 2005. The biology of soil: a community and ecosystem approach. Oxford University Press, New York.

Barley, K.P. 1954. Effects of Root Growth and Decay on the Permeability of A Synthetic Sandy Loam. *Soil Science* **78**, 205–210.

Bartens, J., Day, S.D., Harris, J.R., Dove, J.E., & Wynn, T.M. 2008. Can Urban Tree Roots Improve Infiltration through Compacted Subsoils for Stormwater Management? *Journal of Environmental Quality* **37**, 2048–2057.

Benjamin, J.G., Mikha, M.M., & Vigil, M.F. 2008. Organic Carbon Effects on Soil Physical and Hydraulic Properties in a Semiarid Climate. *Soil Science Society of America Journal* **72**, 1357–1362.

Bens, O., Wahl, N.A., Fischer, H., & Huttl, R.F. 2007. Water infiltration and hydraulic conductivity in sandy cambisols: impacts of forest transformation on soil hydrological properties. *European Journal of Forest Research* **126**, 101–109.

Beven, K., & Germann, P. 1982. Macropores and Water-Flow in Soils. *Water Resources Research* **18**, 1311–1325.

Bezemer, T.M., De Deyn, G.B., Bossinga, T.M., Van Dam, N.M., Harvey, J.A., & Van der Putten, W.H. 2005. Soil community composition drives aboveground plant–herbivore–parasitoid interactions. *Ecology Letters* **8**, 652–661.

Binkley, D. 1994. The influence of tree species on forest soils: processes and patterns. *In* Mead, D.J., Cornforth, I.S. (eds.), Proceedings of the Trees and Soil Workshop. Lincoln University Press, Canterbury, Lincoln University.

Black, K.E., Harbron, C.G., Franklin, M., Atkinson, D., & Hooker, J.E. 1998. Differences in root longevity of some tree species. *Tree Physiology* **18**, 259–264.

Bonell, M., Purandara, B.K., Venkatesh, B., Krishnaswamy, J., Acharya, H.A.K., Singh, U.V., Jayakumar, R., & Chappell, N. 2010. The impact of forest use and reforestation on soil hydraulic conductivity in the Western Ghats of India: Implications for surface and sub-surface hydrology. *Journal of Hydrology* **391**, 47–62.

Borin, M., Vianello, M., Morari, F., & Zanin, G. 2005. Effectiveness of buffer strips in removing pollutants in runoff from a cultivated field in North-East Italy. *Agriculture, Ecosystems & Environment* **105**, 101–114.

Bormann, H., & Klaassen, K. 2008. Seasonal and land use dependent variability of soil hydraulic and soil hydrological properties of two Northern German soils. *Geoderma* **145**, 295–302.

Bouwer, H. 1986. Intake Rate: Cylinder Infiltrometer. *In* Klute, A. (ed.), SSSA Book Series. Soil Science Society of America, American Society of Agronomy.

Boyle, T.P., & Fairchild, J.F. 1997. The role of mesocosm studies in ecological risk analysis. *Ecological Applications* 7, 1099–1102.

Boyle, M., Frankenberger, W.T., & Stolzy, L.H. 1989. The Influence of Organic Matter on Soil Aggregation and Water Infiltration. *Journal of Production Agriculture* **2**, 290–299.

Brady, N.C., & Weil, R.R. 2002. The nature and properties of soils. 13th edition. Prentice Hall, Upper Saddle River, NJ.

Bronick, C.J., & Lal, R. 2005. Soil structure and management: a review. *Geoderma* **124**, 3–22.

Buczko, U., Bens, O., & Huttl, R.F. 2006. Water infiltration and hydrophobicity in forest soils of a pine-beech transformation chronosequence. *Journal of Hydrology* **331**, 383–395.

Burch, G.J., Bath, R.K., Moore, I.D., & O'Loughlin, E.M. 1987. Comparative hydrological behaviour of forested and cleared catchments in southeastern Australia. *Journal of Hydrology* **90**, 19–42.

Capowiez, Y., Sammartino, S., & Michel, E. 2011. Using X-ray tomography to quantify earthworm bioturbation non-destructively in repacked soil cores. *Geoderma* **162**, 124–131.

Capuliak, J., Pichler, V., Fluhler, H., Pichlerova, M., & Homolak, M. 2010. Beech Forest Density Control on the Dominant Water Flow Types in Andic Soils. *Vadose Zone Journal* **9**, 747–756.

Carminati, A., & Vetterlein, D. 2012. Plasticity of rhizosphere hydraulic properties as a key for efficient utilization of scarce resources. *Annals of Botany*, mcs262.

Castellano, M.J., & Valone, T.J. 2007. Livestock, soil compaction and water infiltration rate: Evaluating a potential desertification recovery mechanism. *Journal of Arid Environments* **71**, 97–108.

Chandler, K.R., & Chappell, N.A. 2008. Influence of individual oak (Quercus robur) trees on saturated hydraulic conductivity. *Forest Ecology and Management* **256**, 1222–1229.

Chaney, K., & Swift, R.S. 1984. The Influence of Organic-Matter on Aggregate Stability in Some British Soils. *Journal of Soil Science* **35**, 223–230.

Chappell, N.A., & Franks, S.W. 1996. Property distributions and flow structure in the Slapton Wood Catchment. *Field Studies* **8**, 559–575.

Churchland, C., Grayston, S.J., & Bengtson, P. 2013. Spatial variability of soil fungal and bacterial abundance: Consequences for carbon turnover along a transition from a forested to clear-cut site. *Soil Biology and Biochemistry* **63**, 5–13.

Clothier, B.E., Vogeler, I., & Magesan, G.N. 2000. The breakdown of water repellency and solute transport through a hydrophobic soil. *Journal of Hydrology* **231**, 255–264.

Cornwell, W.K., Cornelissen, J.H.C., Amatangelo, K., Dorrepaal, E., Eviner, V.T., Godoy, O., Hobbie, S.E., Hoorens, B., Kurokawa, H., Pérez-Harguindeguy, N., Quested, H.M., Santiago, L.S., Wardle, D.A., Wright, I.J., Aerts, R., Allison, S.D., Van Bodegom, P., Brovkin, V., Chatain, A., Callaghan, T.V., Díaz, S., Garnier, E., Gurvich, D.E., Kazakou, E., Klein, J.A., Read, J., Reich, P.B., Soudzilovskaia, N.A., Vaieretti, M.V., & Westoby, M. 2008. Plant species traits are the predominant control on litter decomposition rates within biomes worldwide. *Ecology Letters* **11**, 1065–1071.

Cournane, F.C., McDowell, R., Littlejohn, R., & Condron, L. 2011. Effects of cattle, sheep and deer grazing on soil physical quality and losses of phosphorus and suspended sediment losses in surface runoff. *Agriculture, Ecosystems & Environment* **140**, 264–272.

Crawley, M.J. 2013. The R book. Wiley.

Davis, A.A., & Trettin, C.C. 2006. Sycamore and sweetgum plantation productivity on former agricultural land in South Carolina. *Biomass and Bioenergy* **30**, 769–777.

Defra. 2007. A strategy for England's trees, Woods and Forests. Defra.

Derouard, L., Tondoh, J., Vilcosqui, L., & Lavelle, P. 1997. Effects of earthworm introduction on soil processes and plant growth. *Soil Biology and Biochemistry* **29**, 541–545.

Devitt, D.A., & Smith, S.D. 2002. Root channel macropores enhance downward movement of water in a Mojave Desert ecosystem. *Journal of Arid Environments* **50**, 99–108.

Dev Sharma, R., Sarkar, R., & Dutta, S. 2013. Run-off generation from fields with different land use and land covers under extreme storm events. *Current Science (00113891)* **104**, 1046–1053.

Diekmann, M. 2003. Species indicator values as an important tool in applied plant ecology – a review. *Basic and Applied Ecology* **4**, 493–506.

Doerr, S.H., Shakesby, R.A., & Walsh, R.P.D. 2000. Soil water repellency: its causes, characteristics and hydro-geomorphological significance. *Earth-Science Reviews* **51**, 33–65.

Doube, M., Kłosowski, M.M., Arganda-Carreras, I., Cordelières, F.P., Dougherty, R.P., Jackson, J.S., Schmid, B., Hutchinson, J.R., & Shefelbine, S.J. 2010. BoneJ: Free and extensible bone image analysis in ImageJ. *Bone* **47**, 1076–1079.

Drewry, J.J., & Paton, R.J. 2005. Effects of sheep treading on soil physical properties and pasture yield of newly sown pastures. *New Zealand Journal of Agricultural Research* **48**, 39–46.

EC. 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy.

Edwards, C.A., & Bohlen, P.J. 1996. Biology and ecology of earthworms. Chapman & Hall, London.

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

Eldridge, D.J. 1993. Effect of Ants on Sandy Soils in Semiarid Eastern Australia - Local-Distribution of Nest Entrances and Their Effect on Infiltration of Water. *Australian Journal of Soil Research* **31**, 509–518.

Eldridge, D.J., & Freudenberger, D. 2005. Ecosystem wicks: Woodland trees enhance water infiltration in a fragmented agricultural landscape in eastern Australia. *Austral Ecology* **30**, 336–347.

Ellis, J.B. 2013. Sustainable surface water management and green infrastructure in UK urban catchment planning. *Journal of Environmental Planning and Management* **56**, 24–41.

Ellis, J.B., & Mitchell, G. 2006. Urban diffuse pollution: key data information approaches for the Water Framework Directive. *Water and Environment Journal* **20**, 19–26.

Environment Agency. 2008. Best Farming Practices. Environment Agency.

Environmental Change Network. 2012. Dataset accessed 2012-08-02 at http://data.ecn.ac.uk - data citation code: 'ECN:KC8/12'.

Ertsen, A.C.D., Alkemade, J.R.M., & Wassen, M.J. 1998. Calibrating Ellenberg indicator values for moisture, acidity, nutrient availability and salinity in the Netherlands. *Plant Ecology* **135**, 113–124.

EU. 2007. Directive 2007/60/EC of the European Parliament and of the Council of 23 October 2007 on the assessment and management of flood risk.

FAO. 2006. Guidelines for soil description. 4th ed. Food and Agriculture Organization of the United Nations, Rome.

Flury, M., & Flühler, H. 1994. Brilliant Blue FCF as a Dye Tracer for Solute Transport Studies—A Toxicological Overview. *Journal of Environmental Quality* **23**, 1108–1112.

Fogg. 2005. A review of "soft engineering" techniques for on-farm bioremediation of diffuse and point sources of pollution. - ES0132.

Forest Service. 2006. Northern Ireland Forestry: a strategy for sustainability and growth. DARDNI.

Germer, S., Neill, C., Krusche, A.V., & Elsenbeer, H. 2010. Influence of land-use change on near-surface hydrological processes: Undisturbed forest to pasture. *Journal of Hydrology* **380**, 473–480.

Godsey, S., & Elsenbeer, H. 2002. The soil hydrologic response to forest regrowth: a case study from southwestern Amazonia. *Hydrological Processes* **16**, 1519–1522.

Gonzalez-Sosa, E., Braud, I., Dehotin, J., Lassabatère, L., Angulo-Jaramillo, R., Lagouy, M., Branger, F., Jacqueminet, C., Kermadi, S., & Michel, K. 2010. Impact of land use on the hydraulic properties of the topsoil in a small French catchment. *Hydrological Processes* **24**, 2382–2399.

Greacen, E., & Sands, R. 1980. Compaction of forest soils. A review. *Soil Research* **18**, 163–189.

Greenwood, K.L., MacLeod, D.A., Scott, J.M., & Hutchinson, K.J. 2007. Changes to soil physical properties after grazing exclusion. *Soil Use and Management* **14**, 19–24.

Hagedorn, F., & Bundt, M. 2002. The age of preferential flow paths. *Geoderma* **108**, 119–132.

Hale, C.M., Frelich, L.E., Reich, P.B., & Pastor, J. 2005. Effects of European Earthworm Invasion on Soil Characteristics in Northern Hardwood Forests of Minnesota, USA. *Ecosystems* **8**, 911–927.

Hapca, S.M., Houston, A.N., Otten, W., & Baveye, P.C. 2013. New Local Thresholding Method for Soil Images by Minimizing Grayscale Intra-Class Variance. *Vadose Zone Journal* **12**Available at https://www-soils-org.ezproxy.lancs.ac.uk/publications/vzj/abstracts/12/3/vzj2012.0172?search-result=1 (verified 25 August 2013).

Hassler, S.K., Zimmermann, B., van Breugel, M., Hall, J.S., & Elsenbeer, H. 2011. Recovery of saturated hydraulic conductivity under secondary succession on former pasture in the humid tropics. *Forest Ecology and Management* **261**, 1634–1642.

Heiskanen, J., & Mäkitalo, K. 2002. Soil water-retention characteristics of Scots pine and Norway spruce forest sites in Finnish Lapland. *Forest Ecology and Management* **162**, 137–152.

Hill, M.O., Mountford, J.O., Roy, D.B., & Bunce, R.G.H. 1999. Ellenberg's indicator values for British plants. Centre for Ecology and Hydrology, NERC, Abbots Ripton, Huntingdon.

Hillel, D. 1998. Environmental Soil Physics. Academic Press, San Diego, CA.

Hobbie, S.E., Reich, P.B., Oleksyn, J., Ogdahl, M., Zytkowiak, R., Hale, C., & Karolewski, P. 2006. Tree species effects on decomposition and forest floor dynamics in a common garden. *Ecology* **87**, 2288–2297.

Hoogerkamp, M., Rogaar, H., & Eijsackers, H.J.P. 1983. Effect of earthworms on grassland on recently reclaimed polder soils in the Netherlands. p. 85–105. *In* Satchell, J.E. (ed.), Earthworm Ecology. Springer Netherlands.

Houlbrooke, D.J., Drewry, J.J., Monaghan, R.M., Paton, R.J., Smith, L.C., & Littlejohn, R.P. 2009. Grazing strategies to protect soil physical properties and maximise pasture yield on a Southland dairy farm. *New Zealand Journal of Agricultural Research* **52**, 323–336.

Huang, M., Zhang, L., & Gallichand, J. 2003. Runoff responses to afforestation in a watershed of the Loess Plateau, China. *Hydrological Processes* **17**, 2599–2609.

Humann, M., Schuler, G., Muller, C., Schneider, R., Johst, M., & Caspari, T. 2011. Identification of runoff processes - The impact of different forest types and soil properties on runoff formation and floods. *Journal of Hydrology* **409**, 637–649.

IUSS Working Group WRB. 2007. World Reference Base for Soil Resources 2006, first update 2007.

Jarvis, N.J. 2007. A review of non-equilibrium water flow and solute transport in soil macropores: principles, controlling factors and consequences for water quality. *European Journal of Soil Science* **58**, 523–546.

Joffre, R., Vacher, J., Llanos, C. de los, & Long, G. 1988. The dehesa: an agrosilvopastoral system of the Mediterranean region with special reference to the Sierra Morena area of Spain. *Agroforestry Systems* **6**, 71–96.

Johnson-Maynard, J.L., Graham, R.C., Wu, L., & Shouse, P.J. 2002. Modification of soil structural and hydraulic properties after 50 years of imposed chaparral and pine vegetation. *Geoderma* **110**, 227–240.

Jordan, A., Martinez-Zavala, L., & Bellinfante, N. 2008. Heterogeneity in soil hydrological response from different land cover types in southern Spain. *Catena* **74**, 137–143.

Jost, G., Schume, H., Hager, H., Markart, G., & Kohl, B. 2012. A hillslope scale comparison of tree species influence on soil moisture dynamics and runoff processes during intense rainfall. *Journal of Hydrology* **420–421**, 112–124.

Jusoff, K. 1989. Physical Soil-properties Associated with Recreational Use of a Forested Reserve Area in Malaysia. *Environmental Conservation* **16**, 339–342.

Kalliokoski, T., Nygren, P., & Sievanen, R. 2008. Coarse root architecture of three boreal tree species growing in mixed stands. Available at http://www.metla.fi/silvafennica/abs/sa42/sa422189.htm (verified 17 January 2013).

Kandeler, E., Kampichler, C., Joergensen, R.G., & Mölter, K. 1999. Effects of mesofauna in a spruce forest on soil microbial communities and N cycling in field mesocosms. *Soil Biology and Biochemistry* **31**, 1783–1792.

Kasel, S., Singh, S., Sanders, G.J., & Bennett, L.T. 2011. Species-specific effects of native trees on soil organic carbon in biodiverse plantings across north-central Victoria, Australia. *Geoderma* **161**, 95–106.

Keiser, A.D., Knoepp, J.D., & Bradford, M.A. 2013. Microbial communities may modify how litter quality affects potential decomposition rates as tree species migrate. *Plant and Soil* **372**, 167–176.

Keith, A.M., Brooker, R.W., Osler, G.H.R., Chapman, S.J., Burslem, D.F.R.P., & van der Wal, R. 2009. Strong impacts of belowground tree inputs on soil nematode trophic composition. *Soil Biology & Biochemistry* **41**, 1060–1065.

Keith, A.M., Rowe, R.L., Parmar, K., Perks, M.P., Mackie, E., Dondini, M., & McNamara, N.P. 2014. Implications of land use change to Short Rotation Forestry in Great Britain for soil and biomass carbon. *GCB Bioenergy*Available at http://onlinelibrary.wiley.com/doi/10.1111/gcbb.12168/abstract (verified 29 January 2014).

Knowles, R.L. 1991. New Zealand experience with silvopastoral systems: A review. *Forest Ecology and Management* **45**, 251–267.

Kohler-Milleret, R., Bayon, R.-C.L., Chenu, C., Gobat, J.-M., & Boivin, P. 2013. Impact of two root systems, earthworms and mycorrhizae on the physical properties of an unstable silt loam Luvisol and plant production. *Plant and Soil* **370**, 251–265.

Kulli, B., Gysi, M., & Flühler, H. 2003. Visualizing soil compaction based on flow pattern analysis. *Soil and Tillage Research* **70**, 29–40.

Kumar, S., Anderson, S.H., Udawatta, R.P., & Kallenbach, R.L. 2012. Water infiltration influenced by agroforestry and grass buffers for a grazed pasture system. *Agroforestry Systems* **84**, 325–335.

Lado, M., Paz, A., & Ben-Hur, M. 2004. Organic matter and aggregate-size interactions in saturated hydraulic conductivity. *Soil Science Society of America Journal* **68**, 234–242.

Lange, B., Luescher, P., & Germann, P.F. 2009. Significance of tree roots for preferential infiltration in stagnic soils. *Hydrol. Earth Syst. Sci.* **13**, 1809–1821.

Le Bissonnais, Y. 1996. Aggregate stability and assessment of soil crustability and erodibility .1. Theory and methodology. *European Journal of Soil Science* **47**, 425–437.

Lee, K.E., & Foster, R.C. 1991. Soil Fauna and Soil Structure. *Australian Journal of Soil Research* **29**, 745–775.

Léonard, J., Perrier, E., & Rajot, J.L. 2004. Biological macropores effect on runoff and infiltration: a combined experimental and modelling approach. *Agriculture, Ecosystems & Environment* **104**, 277–285.

Lewis, J., & Sjostrom, J. 2010. Optimizing the experimental design of soil columns in saturated and unsaturated transport experiments. *Journal of Contaminant Hydrology* **115**, 1–13.

Lorimer, M.S., & Douglas, L.A. 1995. Effect of Management Practice on Properties of a Victorian Red-Brown Earth .1. Soil Physical-Properties. *Australian Journal of Soil Research* **33**, 851–857.

Lowrance, R., Altier, L.S., Newbold, J.D., Schnabel, R.R., Groffman, P.M., Denver, J.M., Correll, D.L., Gilliam, J.W., Robinson, J.L., Brinsfield, R.B., Staver, K.W., Lucas, W., & Todd, A.H. 1997. Water Quality Functions of Riparian Forest Buffers in Chesapeake Bay Watersheds. *Environmental Management* **21**, 687–712.

Luo, L.F., Lin, H., & Schmidt, J. 2010. Quantitative Relationships between Soil Macropore Characteristics and Preferential Flow and Transport. *Soil Science Society of America Journal* **74**, 1929–1937.

Martens, D.A., & Frankenberger, W.T. 1992. Modification of Infiltration Rates in an Organic-Amended Irrigated Soil. *Agronomy Journal* **84**, 707–717.

Mbagwu, J.S.C. 1995. Saturated hydraulic conductivity in relation to physical properties of soils in the Nsukka Plains, southeastern Nigeria. *Geoderma* **68**, 51–66.

McKay, H. 2011. Short Rotation Forestry: review of growth and environmental impacts. Forest Research, Surrey.

McKenzie, N., Coughlan, K., & Cresswell, H. 2002. Soil Physical Measurement and Interpretation for Land Evaluation.

MEA. 2005. Ecosystems and human well-being: Synthesis. Island Press, Washington, DC.

Messing, I., Alriksson, A., & Johansson, W. 1997. Soil physical properties of afforested and arable land. *Soil Use and Management* **13**, 209–217.

Milcu, A., & Manning, P. 2011. All size classes of soil fauna and litter quality control the acceleration of litter decay in its home environment. *Oikos* **120**, 1366–1370.

Milleret, R., Le Bayon, R.-C., Lamy, F., Gobat, J.-M., & Boivin, P. 2009. Impact of roots, mycorrhizas and earthworms on soil physical properties as assessed by shrinkage analysis. *Journal of Hydrology* **373**, 499–507.

Millward, A.A., Paudel, K., & Briggs, S.E. 2011. Naturalization as a strategy for improving soil physical characteristics in a forested urban park. *Urban Ecosystems* **14**, 261–278.

Mishra, A., & Sharma, S.D. 2010. Influence of forest tree species on reclamation of semiarid sodic soils. *Soil Use and Management* **26**, 445–454.

Mitchell, A.R., Ellsworth, T.R., & Meek, B.D. 1995. Effect of root systems on preferential flow in swelling soil. *Communications in Soil Science and Plant Analysis* **26**, 2655–2666.

Morales, V.L., Parlange, J.Y., & Steenhuis, T.S. 2010. Are preferential flow paths perpetuated by microbial activity in the soil matrix? A review. *Journal of Hydrology* **393**, 29–36.

Mountford, J.O., & Chapman, J.M. 1993. Water Regime Requirements of British Wetland Vegetation: Using the Moisture Classification of Ellenberg and Londo. *Journal of Environmental Management* **38**, 275–288.

Muys, B., Lust, N., & Granval, P. 1992. Effects of grassland afforestation with different tree species on earthworm communities, litter decomposition and nutrient status. *Soil Biology and Biochemistry* **24**, 1459–1466.

Neirynck, J., Mirtcheva, S., Sioen, G., & Lust, N. 2000. Impact of Tilia platyphyllos Scop., Fraxinus excelsior L., Acer pseudoplatanus L., Quercus robur L. and Fagus sylvatica L. on earthworm biomass and physico-chemical properties of a loamy topsoil. *Forest Ecology and Management* **133**, 275–286.

Nemes, A., Rawls, W.J., & Pachepsky, Y.A. 2005. Influence of Organic Matter on the Estimation of Saturated Hydraulic Conductivity. *Soil Science Society of America Journal* **69**, 1330–1337.

Nie, Z.N., Mackay, A.D., Valentine, I., Barker, D.J., & Hodgson, J. 1997. Influence of pastoral fallow on plant root growth and soil physical and chemical characteristics in a hill pasture. *Plant and Soil* **197**, 201–208.

Noguchi, S., Tsuboyama, Y., Sidle, R.C., & Hosoda, I. 1999. Morphological Characteristics of Macropores and the Distribution of Preferential Flow Pathways in a Forested Slope Segment. *Soil Science Society of America Journal* **63**, 1413.

Odum, E.P. 1984. The Mesocosm. *BioScience* **34**, 558–562.

Ovington, J.D. 1953. Studies of the Development of Woodland Conditions under Different Trees .1. Soils pH. *Journal of Ecology* **41**, 13–34.

Palmer, M.A., Filoso, S., & Fanelli, R.M. 2013. From ecosystems to ecosystem services: Stream restoration as ecological engineering. *Ecological Engineering* Available at http://dx.doi.org/10.1016/j.ecoleng.2013.07.059.

Perillo, C., Gupta, S.., Nater, E.., & Moncrief, J.. 1999. Prevalence and initiation of preferential flow paths in a sandy loam with argillic horizon. *Geoderma* **89**, 307–331.

Perkins, K.S., Nimmo, J.R., & Medeiros, A.C. 2012. Effects of native forest restoration on soil hydraulic properties, Auwahi, Maui, Hawaiian Islands. *Geophysical Research Letters* **39**, L05045.

Petersen, A.E. & Dixon, R.M. 1971. Water movement in large soil pores. *Res. Rep. 75,* pp. 1-8, Res. Div. Coll. Agr. Life Sci., Univ. Wisc., Madison

Pietola, L., Horn, R., & Yli-Halla, M. 2005. Effects of trampling by cattle on the hydraulic and mechanical properties of soil. *Soil and Tillage Research* **82**, 99–108.

Prescott, C.E. 2010. Litter decomposition: what controls it and how can we alter it to sequester more carbon in forest soils? *Biogeochemistry* **101**, 133–149.

Pritchett, W.L., & Fisher, R.F. 1987. Properties and management of forest soils. 2nd edition. Wiley, New York.

R Core Team. 2013. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Raper, R.L. 2005. Agricultural traffic impacts on soil. *Journal of Terramechanics* **42**, 259–280.

Rasband, W.S. 2002. ImageJ. National Institutes of Health, USA.

Regent Instruments Inc., Canada. WinRHIZO. Regent Instruments Inc., Canada.

Reich, P.B., Oleksyn, J., Modrzynski, J., Mrozinski, P., Hobbie, S.E., Eissenstat, D.M., Chorover, J., Chadwick, O.A., Hale, C.M., & Tjoelker, M.G. 2005. Linking litter calcium, earthworms and soil properties: a common garden test with 14 tree species. *Ecology Letters* **8**, 811–818.

Reynolds, W.D., & Elrick, D.E. 1990. Ponded Infiltration from a Single Ring .1. Analysis of Steady Flow. *Soil Science Society of America Journal* **54**, 1233–1241.

Reynolds, W.D., & Elrick, D.E. 1991. Determination of Hydraulic Conductivity Using a Tension Infiltrometer. *Soil Science Society of America Journal* **55**, 633–639.

Reynolds, W.D., & Elrick, D.E. 2002. 3.4.3.2.b Pressure Infiltrometer. *In* Dane, J.H., Topp, G.C. (eds.), Methods of soil analysis: Part 4 -Physical methods. SSSA Book Series. Soil Science Society of America.

Reynolds, W.D., Elrick, D.E., & Youngs, E.G. 2002. 3.4.3.2 Ring or Cylinder Infiltrometers (Vadose Zone). *In* Dane, J.H., Topp, G.C. (eds.), Methods of soil analysis: Physical methods. SSSA Book Series. Soil Science Society of America.

Rogasik, H., Schrader, S., Onasch, I., Kiesel, J., & Gerke, H.H. 2014. Micro-scale dry bulk density variation around earthworm (Lumbricus terrestris L.) burrows based on X-ray computed tomography. *Geoderma* **213**, 471–477.

Rowell, D.L. 1994. Soil Science: methods and applications. Longman Scientific & Technical, Harlow, England [etc.].

Sanou, J., Zougmore, R., Bayala, J., & Teklehaimanot, Z. 2010. Soil infiltrability and water content as affected by Baobab (Adansonia digitata L.) and Nere (Parkia biglobosa (Jacq.) Benth.) trees in farmed parklands of West Africa. *Soil Use and Management* **26**, 75–81.

Schaffers, A.P., & Sýkora, K.V. 2000. Reliability of Ellenberg indicator values for moisture, nitrogen and soil reaction: a comparison with field measurements. *Journal of Vegetation Science* **11**, 225–244.

Scharenbroch, B.C., & Johnston, D.P. 2011. A microcosm study of the common night crawler earthworm (Lumbricus terrestris) and physical, chemical and biological properties of a designed urban soil. *Urban Ecosystems* **14**, 119–134.

Scheu, S., Albers, D., Alphei, J., Buryn, R., Klages, U., Migge, S., Platner, C., & Salamon, J.-A. 2003. The soil fauna community in pure and mixed stands of beech and spruce of different age: trophic structure and structuring forces. *Oikos* **101**, 225–238.

Schutz, K., Nagel, P., Dill, A., & Scheu, S. 2008. Structure and functioning of earthworm communities in woodland flooding systems used for drinking water production. *Applied Soil Ecology* **39**, 342–351.

Schwärzel, K., Ebermann, S., & Schalling, N. 2012. Evidence of double-funneling effect of beech trees by visualization of flow pathways using dye tracer. *Journal of Hydrology* **470–471**, 184–192.

Scottish Executive. 2006. The Scottish Forestry Strategy. Forestry Commission Scotland.

Seto, K.C., Fragkias, M., Güneralp, B., & Reilly, M.K. 2011. A Meta-Analysis of Global Urban Land Expansion. *PLoS ONE* **6**, e23777.

Sharrow, S. 2007. Soil compaction by grazing livestock in silvopastures as evidenced by changes in soil physical properties. *Agroforestry Systems* **71**, 215–223.

Shaw, E.M. 1994. Hydrology in practice. Chapman & Hall, London; New York.

Sheldrick, R., & Auclair, D. 2000. Origins of agroforestry and recent histrory in the UK. *In* Hislop, M., Claridge, J. (eds.), Agroforestry in the UK. Forestry Commission, Edinburgh.

Shougrakpam, S., Sarkar, R., & Dutta, S. 2010. An experimental investigation to characterise soil macroporosity under different land use and land covers of northeast India. *Journal of Earth System Science* **119**, 655–674.

Six, J., Feller, C., Denef, K., Ogle, S.M., Sa, J.C.D., & Albrecht, A. 2002. Soil organic matter, biota and aggregation in temperate and tropical soils - Effects of no-tillage. *Agronomie* **22**, 755–775.

Six, J., Paustian, K., Elliott, E.T., & Combrink, C. 2000. Soil structure and organic matter: I. Distribution of aggregate-size classes and aggregate-associated carbon. *Soil Science Society of America Journal* **64**, 681–689.

Spurgeon, D.J., Keith, A.M., Schmidt, O., Lammertsma, D.R., & Faber, J.H. 2013. Land-use and land-management change: relationships with earthworm and fungi communities and soil structural properties. *BMC Ecology* **13**, 46.

St John, M.G., Orwin, K.H., & Dickie, I.A. 2011. No "home" versus "away" effects of decomposition found in a grassland-forest reciprocal litter transplant study. *Soil Biology & Biochemistry* **43**, 1482–1489.

Stoate, C., Boatman, N.D., Borralho, R.J., Carvalho, C.R., Snoo, G.R. de, & Eden, P. 2001. Ecological impacts of arable intensification in Europe. *Journal of Environmental Management* **63**, 337–365.

Talsma, T., & Hallam, P.M. 1980. Hydraulic Conductivity Measurement of Forest Catchments. *Australian Journal of Soil Research* **18**, 139–148.

Tang, G., & Li, K. 2013. Tree species controls on soil carbon sequestration and carbon stability following 20 years of afforestation in a valley-type savanna. *Forest Ecology and Management* **291**, 13–19.

Tang, J., Mo, Y., Zhang, J., & Zhang, R. 2011. Influence of biological aggregating agents associated with microbial population on soil aggregate stability. *Applied Soil Ecology* **47**, 153–159.

Teuben, A., & Verhoef, H.A. 1992. Relevance of Micro- and Mesocosm Experiments for Studying Soil Ecosystem Processes. *Soil Biology & Biochemistry* **24**, 1179–1183.

Thomas, P. 2000. Trees: their natural history. Cambridge University Press, Cambridge, U.K.; New York.

Thomas, P., & Packham, J.R. 2007. Ecology of woodlands and forests: description, dynamics and diversity. Cambridge University Press, Cambridge; New York.

Tisdall, J.M., & Oades, J.M. 1982. Organic-Matter and Water-Stable Aggregates in Soils. *Journal of Soil Science* **33**, 141–163.

Toriyama, J., Ohta, S., Araki, M., Kosugi, K., Nobuhiro, T., Kabeya, N., Shimizu, A., Tamai, K., Kanzaki, M., & Chann, S. 2011. Soil pore characteristics of evergreen and deciduous forests of the tropical monsoon region in Cambodia. *Hydrological Processes* **25**, 714–726.

Trum, F., Titeux, H., Ranger, J., & Delvaux, B. 2011. Influence of tree species on carbon and nitrogen transformation patterns in forest floor profiles. *Annals of Forest Science* **68**, 837–847.

Udawatta, R.P., & Anderson, S.H. 2008. CT-measured pore characteristics of surface and subsurface soils influenced by agroforestry and grass buffers. *Geoderma* **145**, 381–389.

Udawatta, R.P., Gantzer, C.J., Anderson, S.H., & Garrett, H.E. 2008. Agroforestry and grass buffer effects on pore characteristics measured by high-resolution x-ray computed tomography. *Soil Science Society of America Journal* **72**, 295–304.

Vesterdal, L., Schmidt, I.K., Callesen, I., Nilsson, L.O., & Gundersen, P. 2008. Carbon and nitrogen in forest floor and mineral soil under six common European tree species. *Forest Ecology and Management* **255**, 35–48.

Vivanco, L., & Austin, A.T. 2008. Tree species identity alters forest litter decomposition through long-term plant and soil interactions in Patagonia, Argentina. *Journal of Ecology* **96**, 727–736.

Volume Graphics. VG Studio Max. Volume Graphics, Heidelberg, Germany.

Wahl, N.A., Bens, O., Schafer, B., & Huttl, R.F. 2003. Impact of changes in land-use management on soil hydraulic properties: hydraulic conductivity, water repellency and water retention. *Physics and Chemistry of the Earth* **28**, 1377–1387.

Wang, T., Wedin, D., & Zlotnik, V.A. 2009. Field evidence of a negative correlation between saturated hydraulic conductivity and soil carbon in a sandy soil. *Water Resources Research* **45**, n/a–n/a.

Watson, K.W., & Luxmoore, R.J. 1986. Estimating Macroporosity in a Forest Watershed by Use of a Tension Infiltrometer. *Soil Science Society of America Journal* **50**, 578–582.

Welsh Assembly Government. 2009. Woodlands for Wales. Forestry Commission Wales.

Wheeler, M.A., Trlica, M.J., Frasier, G.W., & Reeder, J.D. 2002. Seasonal grazing affects soil physical properties of a montane riparian community. *Journal of Range Management* **55**, 49–56.

Whelan, A., Kechavarzi, C., Coulon, F., Sakrabani, R., & Lord, R. 2013. Influence of compost amendments on the hydraulic functioning of brownfield soils. *Soil Use and Management* **29**, 260–270.

Willatt, S., & Pullar, D. 1984. Changes in soil physical properties under grazed pastures. *Australian Journal of Soil Research* **22**, 343–348.

Withington, J.M., Reich, P.B., Oleksyn, J., & Eissenstat, D.M. 2006. Comparisons of Structure and Life Span in Roots and Leaves among Temperate Trees. *Ecological Monographs* **76**, 381–397.

Wood, H.B. 1977. Hydrologic Differences between Selected Forested and Agricultural Soils in Hawaii. *Soil Science Society of America Journal* **41**, 132–136.

XTEK Systems Ltd. 2004. CT-Pro.

Yeates, G.W., & Boag, B. 1995. Relation between macrofaunal populations and soil physical properties in two agroforestry trials. *Acta Zoologica Fennica* **196**, 275–277.

Young, A. 1997. Agroforestry for soil management. CAB International in association with the International Centre for Research in Agroforestry, New York.

Zhang, D., Hui, D., Luo, Y., & Zhou, G. 2008. Rates of litter decomposition in terrestrial ecosystems: global patterns and controlling factors. *Journal of Plant Ecology* **1**, 85–93.

Ziemer, R.R. 1968. Soil Moisture Depletion Patterns Around Scattered Trees. U.S. Department of Agriculture, Forest Service, Pacific Southwest Forest & Range Experiment Station.

Zimmermann, B., Elsenbeer, H., & De Moraes, J.M. 2006. The influence of land-use changes on soil hydraulic properties: Implications for runoff generation. *Forest Ecology and Management* **222**, 29–38.

Zimmermann, B., Papritz, A., & Elsenbeer, H. 2010. Asymmetric response to disturbance and recovery: Changes of soil permeability under forest-pasture-forest transitions. *Geoderma* **159**, 209–215.

Zuur, A.F., Ieno, E.N., & Elphick, C.S. 2010. A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution* **1**, 3–14.

APPENDIX

Determination of aggregate stability following the Le Bissonais (1996) method

A number of methods have been developed to test aggregate stability; however each tends to emphasise a particular mechanism of aggregate breakdown. Since different soils may be more prone to breakdown under a particular mechanism, the Le Bissonais (1996) method combines three tests that simulate the different mechanisms involved.

Field sampling and preparation

1. Take all samples at the same time of the year and avoid extreme conditions where the soil is frozen, very wet or very dry. Optimal conditions are when the soil can be broken apart by hand.
2. Transfer samples from field to laboratory in rigid boxes to avoid crushing the aggregates and immediately air dry.
3. Place a 5 mm sieve above a 2 mm sieve and force the air-dried sample through the 5 mm sieve to collect aggregates in the 2 – 5 mm range.
4. Just before treatment, place the samples in the oven at 40°C for 24 hours so that they are at constant matric potential.

Treatment 1: Fast wetting

Aggregates are immersed in water to simulate rapid wetting, such as may occur during heavy rain storms. Immersion is the simplest way to check stability and can be used as a simple field test.

1. Add 50 ml of de-ionised water to a 250 ml beaker and gently immerse 5 g of aggregates for 10 minutes.
2. Use a pipette to suck the water out of the beaker.
3. Transfer the sample to a 50 μm sieve that has previously been immersed in ethanol and then follow the instructions for fragment size distribution measurement (below).

Treatment 2: Slow wetting

Slow wetting with controlled tension corresponds to a field condition of wetting under gentle rain.

1. Place a 110 mm diameter filter paper on a tension table set to a matric potential of -0.3 kPa and allow it to wet through.
2. Place 5 g of aggregates on the filter paper and leave for 30 minutes.
3. Transfer the sample to a 50 μm sieve that has previously been immersed in ethanol and then follow the instructions for fragment size distribution measurement (below).

Treatment 3: Mechanical breakdown by shaking after pre-wetting

This treatment includes pre-wetting the aggregates in ethanol so that wet mechanical cohesion is tested independently of slaking. By pre-wetting in ethanol, air is removed from the aggregates before energy is applied.

1. Add 50 ml of ethanol to a 250 ml beaker and gently immerse 5 g of aggregates for 10 minutes.
2. Use a pipette to suck the ethanol out of the beaker.
3. Transfer the sample to a 250 ml conical flask filled with 50 ml of de-ionised water.
4. Add a further 150 ml of de-ionised water to make up the water content to 200 ml.
5. Place a stopper in the top of the flask and then agitate the flask end over end 20 times.
6. Leave for 30 minutes to allow sedimentation of the coarse fragments and then use a pipette to suck off the excess water.
7. Transfer the sample to a 50 μm sieve that has previously been immersed in ethanol and then follow the instructions for fragment size distribution measurement (below).

Fragment size distribution measurement

This step is divided into two operations and is intended to measure the result of the breakdown occurring for each of the treatments while trying to minimise any further breakdown.

The first operation involves wet sieving in ethanol and should be undertaken immediately after each of the other treatments. The second operation involves dry sieving and can only be undertaken after the ethanol has been allowed to evaporate and the sample oven dried.

Wet sieving:

1. In a fume cupboard, immerse the sieve with the sample on it into a bowl filled with ethanol.
2. Gently agitate the sieve 5 times using a circular motion to separate the > 50 µm fraction from the < 50 µm fraction.
3. Transfer the sample to a foil dish and leave in the fume cupboard overnight to allow the ethanol remaining on the sample to evaporate.

Dry sieving:

1. Place the air-dry sample in an oven at 105°C for 24 hours.
2. Transfer the oven-dried sample to the top of a sieve stack (2000, 1000, 500, 200, 100, 50 µm in size).
3. Gently sieve by hand.
4. Weigh the fractions left on each sieve.

Calculating fragment size distribution and mean weight diameter:

1. Calculate the mass percentage of each size fraction, assuming that the difference between the initial mass and the sum of the six weighed fractions is the mass of the < 50 µm fraction.

2. The aggregate stability for each breakdown mechanism can be expressed using the resulting fragment size distribution (FSD) in the seven size classes or by calculating the mean weight diameter (MWD), which is the sum of the mass fraction of soil remaining on each sieve multiplied by the mean aperture of the sieve mesh (MWDs range from 25 µm to 3.5 mm). MWDs may be classified as follows:

Class	**MWD value (mm)**	**Stability**	**Crustability**
1	< 0.4	Very unstable	Systematic crust formation
2	0.4 – 0.8	Unstable	Crusting frequent
3	0.8 – 1.3	Medium	Crusting moderate
4	1.3 – 2.0	Stable	Crusting rare
5	> 2.0	Very stable	No crusting

The three tests generally give similar MWDs for very stable or very unstable soils; however it is possible that only one treatment will give a good discrimination for a specific group of soils because of threshold effects. Since each test corresponds to specific conditions, it may be desirable to use the results of a single treatment or a combination of all three.